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Fluridone Ecological Risk Assessment, Final Report

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Fluridone
Ecological Risk Assessment

Final Report

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Executive Summary

The Bureau of Land Management (BLM), United States Department of the Interior (USDI), is proposing a program to treat vegetation on up to six million acres of public lands annually in 17 western states in the continental United States (U.S.) and Alaska. As part of this program, the BLM is proposing the use of ten herbicide active ingredients (a.i.) to control invasive plants and noxious weeds on approximately one million of the 6 million acres proposed for treatment. The BLM and its contractor, ENSR, are preparing a Vegetation Treatments Programmatic Environmental Impact Statement (EIS) to evaluate this and other proposed vegetation treatment methods and alternatives on lands managed by the BLM in the western continental US and Alaska. In support of the EIS, this Ecological Risk Assessment (ERA) evaluates the potential risks to the environment that would result from the use of the herbicide fluridone, including risks to rare, threatened, and endangered (RTE) plant and animal species.

One of the BLM's highest priorities is to promote ecosystem health, and one of the greatest obstacles to achieving this goal is the rapid expansion of invasive plants (including noxious weeds and other plants not native to the region) across public lands. These invasive plants can dominate and often cause permanent damage to natural plant communities. If not eradicated or controlled, invasive plants will jeopardize the health of public lands and the activities that occur on them. Herbicides are one method employed by the BLM to control these plants.

Herbicide Description

Fluridone is a selective systemic herbicide that inhibits carotene production in leaves, which causes the breakdown of chlorophyll—preventing the plant from synthesizing food. This herbicide comes in two formulations: liquid and pellet. Fluridone is used by the BLM for vegetation control in their Aquatic program. Application is carried out through both aerial and ground dispersal. Aerial dispersal is executed through the use of a plane or helicopter. Ground applications are executed on foot or horseback with backpack sprayers, or from all terrain vehicles or trucks equipped with spot or boom/broadcast sprayers. The BLM applies fluridone at different rates depending on the waterbody category (i.e., Ponds, Whole Lake/Reservoir, Partial Lakes/Reservoir, or Canals). In order for the risk assessment simulations to span the concentration range of applied herbicide in typical and maximum cases, the lowest typical application rate (Whole Lake/Reservoir) was selected for use as the typical rate and the highest maximum application rate (Partial Lake/Reservoir) was selected for use as the maximum application rate. The lowest typical application of fluridone is 0.15 pounds (lbs) a.i. per acre (a.i./ac). The maximum application rate is 1.3 lbs a.i./ac.

Ecological Risk Assessment Guildelines

The main objectives of this ERA were to evaluate the potential ecological risks from fluridone to the health and welfare of plants and animals and their habitats and to provide risk managers with a range of generic risk estimates that vary as a function of site conditions. The categories and guidelines listed below were designed to help the BLM determine which of the proposed alternatives evaluated in the EIS should be used on BLM-managed lands.

- Exposure pathway evaluation – The effects of fluridone on several ecological receptor groups (i.e., terrestrial animals, non-target terrestrial plants, fish and aquatic invertebrates, and non-target aquatic plants) via particular exposure pathways were evaluated. The resulting exposure scenarios included the following:
  - direct contact with the herbicide or a contaminated waterbody;
  - indirect contact with contaminated foliage;
  - ingestion of contaminated food items;
  - off-site drift of spray to terrestrial areas; and
  - accidental spills to waterbodies.

- Definition of data evaluated in the ERA – Herbicide concentrations used in the ERA were based on typical and maximum application rates provided by the BLM. These application rates were used to predict herbicide concentrations in various environmental media (e.g., soils, water). Some of these calculations required the computer model AgDRIFT®, which was used to estimate off-site herbicide transport due to spray drift, and an additional sensitivity model designed to determine how pond and stream volumes affect exposure concentrations

- Identification of risk characterization endpoints – Endpoints used in the ERA included acute mortality; adverse direct effects on growth, reproduction, or other ecologically important sublethal processes; and adverse indirect effects on the survival, growth, or reproduction of salmonid fish. Each of these endpoints was associated with measures of effect such as the no observable adverse effect level (NOAEL) and the median lethal effect dose and median lethal concentration (LD_{50} and LC_{50}).

- Development of a conceptual model – The purpose of the conceptual model is to display working hypotheses about how fluridone might pose hazards to ecosystems and ecological receptors. This is shown via a diagram of the possible exposure pathways and the receptors evaluated for each exposure pathway.

In the analysis phase of the ERA, estimated exposure concentrations (EECs) were identified for the various receptor groups in each of the applicable exposure scenarios via exposure modeling. Risk quotients (RQs) were then calculated by dividing the EECs by herbicide- and receptor-specific or exposure media-specific Toxicity Reference Values (TRVs) selected from the available literature. These RQs were compared to Levels of Concern (LOCs) established by the United States Environmental Protection Agency (USEPA) Office of Pesticide Programs (OPP) for specific risk presumption categories (i.e., acute high risk, acute high risk potentially mitigated through restricted use, acute high risk to endangered species, and chronic high risk).

Uncertainty

Uncertainty is introduced into the herbicide ERA through the selection of surrogates to represent a broad range of species on BLM-managed lands, the use of mixtures of fluridone with other herbicides (tank mixtures) or other potentially toxic ingredients (i.e., degradates, inert ingredients, and adjuvants), and the estimation of effects via exposure concentration models. The uncertainty inherent in screening level ERAs is especially problematic for the evaluation of risks to RTE species, which are afforded higher levels of protection through government regulations and policies. To attempt to minimize the chances of underestimating risk to RTE and other species, the lowest toxicity levels found in the literature were selected as TRVs; uncertainty factors were incorporated into these TRVs; allometric scaling was used to develop dose values; model assumptions were designed to conservatively estimate herbicide exposure; and indirect as well as direct effects on species of concern were evaluated.

Herbicide Effects

Literature Review

According to the Ecological Incident Information System (EIIS) database run by the USEPA OPP, fluridone has been associated with only one reported "ecological incident" involving damage or mortality to non-target flora. It was listed as probable that direct contact of fluridone was responsible.

A review of the available ecotoxicological literature was conducted in order to evaluate the potential for fluridone to negatively directly or indirectly affect non-target taxa. This review was also used to identify or derive TRVs for use in the ERA. The sources identified in this review indicate that fluridone has low toxicity to most terrestrial species. Studies conducted with mammals found that acute exposure to fluridone does not commonly 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

(*Apis* spp.). While no quantitative data were found to evaluate fluridone's effects on terrestrial plants, qualitative results indicate that the sensitivity of terrestrial plants is variable. Some plant species (e.g., grasses and sedges) were more sensitive than others (e.g., willow).

Fluridone is an herbicide used to control aquatic plants. In the available literature, aquatic plants were not affected by concentrations up to 1 milligrams (mg) a.i./liter (L) (typical herbicide application rates used in the direct spray scenarios in this ERA resulted in a pond concentration of 0.017 mg a.i./L and a stream concentration of 0.084 mg a.i./L). Acute and chronic toxicity tests indicate that fluridone causes toxicity to fish species at concentrations of 10 mg/L, with some adverse effect concentrations approaching 1 mg/L. Acute toxicity concentrations for aquatic invertebrates reached 1.3 mg/L. No data were found to evaluate the toxicity of fluridone to amphibians.

Ecological Risk Assessment Results

Based on the ERA conducted for fluridone, there is the potential for risk to selected ecological receptors from exposure to herbicides under specific conditions on BLM-managed lands. The following bullets summarize the risk assessment findings for fluridone under each evaluated exposure scenario:

- Direct Spray – No risks were predicted for terrestrial wildlife (i.e., insects, birds, or mammals). Risks to terrestrial plants could not be evaluated as a result of a lack of toxicity information; however, one ecological incident report suggests the potential for risk to terrestrial plants. No risks to non-target aquatic plants are predicted when waterbodies are accidentally (streams) or intentionally (ponds) sprayed, but risks to fish or aquatic invertebrates may occur when waterbodies are accidentally or intentionally sprayed.

- Off-Site Drift to Non-Target Terrestrial Plants – Risks to terrestrial plants could not be evaluated because of a lack of toxicity information; however, product literature and one ecological incident report suggest the potential for risk.

- Accidental Spill to Pond – Risk to fish, aquatic invertebrates, and non-target aquatic plants may occur when herbicides are spilled directly into the pond.

Based on the results of the ERA, it is unlikely that RTE species would be harmed by appropriate use (see following section) of the herbicide fluridone on BLM-managed lands.

Recommendations

The following recommendations are designed to reduce potential unintended impacts to the environment from the application of fluridone:

- Select adjuvants carefully (none are currently ingredients in fluridone-containing Sonar products) since these have the potential to increase the level of toxicity above that predicted for the a.i. alone. This is especially important for application scenarios that already predict potential risk from the a.i. itself.

- Review, understand, and conform to "Environmental Hazards" section on 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 the environment.

- Avoid accidental direct spray on the stream to reduce the most significant potential impacts.

- Because the effects of normal herbicide application on terrestrial plants are uncertain, limit fluridone use in areas where RTE plants are near application areas. Avoid accidental direct spray and off-site drift to terrestrial plants to reduce potential impacts observed in a previous ecological incident report (Section 2.3). Limit fluridone application in wind, and monitor effects on adjacent terrestrial vegetation.

- Use the typical application rate in the pond to reduce risk to fish and aquatic invertebrates.

The results from this ERA assist the evaluation of proposed alternatives in the EIS and contribute to the development of a Biological Assessment (BA), specifically addressing the potential impacts to proposed and listed RTE species on western BLM treatment lands. Furthermore, this ERA will inform BLM field offices on the proper application of fluridone to ensure that impacts to plants and animals and their habitat are minimized to the extent practical.

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LIST OF ACRONYMS, ABBREVIATIONS, AND SYMBOLS

ac	-	acres
a.i.	-	active ingredient
BA	-	Biological Assessment
BCF	-	Bioconcentration Factor
BLM	-	Bureau of Land Management
BO	-	Biological Opinion
BW	-	Body Weight
°C	-	Degrees Celsius
CBI	-	Confidential Business Information
cm	-	centimeter
cms	-	cubic meters per second
CWE	-	Cumulative Watershed Effect
DPR	-	Department of Pesticide Registration
EC_{25}	-	Concentration causing 25% inhibition of a process (Effect Concentration)
EC_{50}	-	Concentration causing 50% inhibition of a process (Median Effective Concentration)
EEC	-	Estimated Exposure Concentration
EIS	-	Environmental Impact Statement
EIIS	-	Ecological Incident Information System
EFED	-	Environmental Fate and Effects Division
ERA	-	Ecological Risk Assessment
ESA	-	Endangered Species Act
FIFRA	-	Federal Insecticide, Fungicide, and Rodenticide Act
FOIA	-	Freedom of Information Act
ft	-	feet
g	-	grams
gal	-	gallon
GLEAMS	-	Groundwater Loading Effects of Agricultural Management Systems
HHRA	-	Human Health Risk Assessment
HSDB	-	Hazardous Substances Data Bank
IPM	-	Integrated Pest Management
IRIS	-	Integrated Risk Information System
ISO	-	International Organization for Standardization
IUPAC	-	International Union of Pure and Applied Chemistry
Kd	-	Partition coefficient
kg	-	Kilogram
K_{oc}	-	Organic carbon-water partition coefficient
K_{ow}	-	Octanol-water partition coefficient
L	-	Liters
lb(s)	-	pound(s)
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(s)	-	Level(s) of Concern
Log	-	Common logarithm (base 10)
m	-	meters
mg	-	milligrams

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

mg/kg	-	milligrams per kilogram
mg/L	-	milligrams per liter
mmHg	-	millimeters of mercury
MSDS	-	Material Safety Data Sheet
MW	-	Molecular Weight
NMFS	-	National Marine Fisheries Service
NOAA	-	National Oceanic and Atmospheric Administration
NOAEL	-	No Observed Adverse Effect Level
OPP	-	Office of Pesticide Programs
OPPTS	-	Office of Pollution Prevention and Toxic Substances
ORNL	-	Oak Ridge National Laboratory
ppm	-	parts per million
RQ	-	Risk Quotient
RTE	-	Rare, Threatened, and Endangered
RTEC	-	Registry of Toxic Effects of Chemical Substances
SDTF	-	Spray Drift Task Force
TOXNET	-	National Library of Medicines Toxicology Data Network
TP	-	Transformation Product
TRV	-	Toxicity Reference Value
TSCA		Toxic Substances Control Act
US	-	United States
USDA	-	United States Department of Agriculture
USDI	-	United States Department of Interior
USEPA	-	United States Environmental Protection Agency
USFWS	-	United States Fish and Wildlife Service
µg	-	micrograms
>	-	greater than
<	-	less than
=	-	equal to

1.0 INTRODUCTION

The Bureau of Land Management (BLM), United States Department of the Interior (USDI), is proposing a program to treat vegetation on up to six million acres of public lands annually in 17 western states in the continental United States (U.S.) and Alaska. The primary objectives of the proposed program include fuels management, weed control, and fish and wildlife habitat restoration. Vegetation would be managed using five primary vegetation treatment methods - mechanical, manual, biological, chemical, and prescribed fire.

The BLM and its contractor, ENSR, are preparing a *Vegetation Treatments Programmatic Environmental Impact Statement* (EIS) to evaluate proposed vegetation treatment methods and alternatives on lands managed by the BLM in the western continental US and Alaska (ENSR 2004a). As part of the EIS, several ERAs and a Human Health Risk Assessment (HHRA; ENSR 2004b) were conducted on several herbicides used, or proposed for use, by the BLM. These risk assessments evaluate potential risks to the environment and human health from exposure to these herbicides both during and after treatment of public lands. For the ERAs, the herbicide a.i. evaluated were tebuthiuron, diuron, bromacil, chlorsulfuron, sulfometuron-methyl, diflufenzopyr, Overdrive® (a mix of dicamba and diflufenzopyr), imazapic, diquat, and fluridone. The HHRA evaluated the risks to humans from only six a.i. (sulfometuron-methyl, imazapic, diflufenzopyr, dicamba, diquat, and fluridone) because the other a.i. were already quantitatively evaluated in previous EISs (e.g., BLM 1991). [Note that in the HHRA, Overdrive® was evaluated as its two separate components, dicamba and diflufenzopyr, as these two a.i. have different toxicological endpoints, indicating that their effects on human health are not additive.] The purpose of this document is to summarize results of the ERA for the herbicide fluridone.

Updated risk assessment methods were developed for both the HHRA and ERA and are described in a separate document, *Vegetation Treatments Programmatic EIS Ecological Risk Assessment Methodology* (hereafter referred to as the "Methods Document;" ENSR 2004c). The methods document provides, in detail, specific information and assumptions used this ERA.

1.1 Objectives of the Ecological Risk Assessment

The purpose of the ERA is to evaluate the ecological risks of ten herbicides on the health and welfare of plants and animals and their habitats, including threatened and endangered species. This analysis will be used by the BLM, in conjunction with analyses of other treatment effects on plants and animals, and effects of treatments on other resources, to determine which of the proposed treatment alternatives evaluated in the EIS should be used by the BLM. The BLM Field Offices will also utilize this ERA 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. The US Fish and Wildlife Service (USFWS) and National Oceanic and Atmospheric Administration Fisheries Service (NOAA Fisheries), in their preparation of a Biological Opinion (BO), will also use the information provided by the ERA to assess the potential impact of vegetation treatment actions on fish and wildlife and their critical habitats.

This ERA, which provides specific information regarding the use of the terrestrial herbicide fluridone, contains the following sections:

Section 1: Introduction

Section 2: BLM Herbicide Program Description – This section contains information regarding herbicide formulation, mode of action, and specific BLM herbicide use, which includes application rates and methods of dispersal. This section also contains a summary of incident reports documented with the United States Environmental Protection Agency (USEPA).

Section 3: Herbicide Toxicology, Physical-Chemical Properties, and Environmental Fate – This section contains a summary of scientific literature pertaining to the toxicology and environmental fate of fluridone in terrestrial and aquatic environments, and discusses how its physical-chemical properties are used in the risk assessment.

Section 4: Ecological Risk Assessment – This section describes the exposure pathways and scenarios and the assessment endpoints, including potential measured effects. It provides quantitative estimates of risks for several risk pathways and receptors.

Section 5: Sensitivity Analysis – This section describes the sensitivity of each of three models used for the ERA to specific input parameters. The importance of these conditions to exposure concentration estimates is discussed.

Section 6: Rare, Threatened, and Endangered Species (RTE) – This section identifies RTE species potentially directly and/or indirectly affected by the herbicide program. It also describes how the ERA can be used to evaluate potential risks to RTE species.

Section 7: Uncertainty in the Ecological Risk Assessment – This section describes data gaps and assumptions made during the risk assessment process and how uncertainty should be considered in interpreting results.

Section 8: Summary – This section provides a synopsis of the ecological receptor groups, application rates, and modes of exposure. This section also provides a summary of the factors that most influence exposure concentrations with general recommendations for risk reduction.

2.0 BLM HERBICIDE PROGRAM DESCRIPTION

2.1 Problem Description

One of the BLM's highest priorities is to promote ecosystem health, and one of the greatest obstacles to achieving this goal is the rapid expansion of weeds across public lands. These invasive plants can dominate and often cause permanent damage to natural plant communities. If not eradicated or controlled, noxious weeds will jeopardize the health of public lands and the myriad of activities that occur on them. The BLM's ability to respond effectively to the challenge of noxious weeds depends on the adequacy of the agency's resources.

Millions of acres of once healthy, productive rangelands, forestlands and riparian areas have been overrun by noxious or invasive weeds. Noxious weeds are any plant designated by a federal, state, or county government as injurious to public health, agriculture, recreation, wildlife, or property (Sheley et al. 1999). Invasive plants include not only noxious weeds, but also other plants that are not native to the region. The BLM considers plants invasive if they have been introduced into an environment where they did not evolve. Invasive plants usually have no natural enemies to limit their reproduction and spread (Westbrooks 1998). They invade recreation areas, BLM-managed public lands, National Parks, State Parks, roadsides, streambanks, federal, state, and private lands. Invasive weeds can:

- destroy wildlife habitat, reduce opportunities for hunting, fishing, camping and other recreational activities;
- displace RTE species and other species critical to ecosystem functioning (e..g, riparian plants);
- reduce plant and animal diversity;
- invade following wildland and prescribed fire (potentially into previously unaffected areas), limiting regeneration and establishment of native species and rapidly increasing acreage of infested land;
- increase fuel loads and decrease the length of fire cycles and/or increase the intensity of fires;
- disrupt waterfowl and neo-tropical migratory bird flight patterns and nesting habitats; and
- cost millions of dollars in treatment and loss of productivity to private land owners.

The BLM uses an Integrated Pest Management (IPM) approach to manage invasive plants. Management techniques may be biological, mechanical, chemical, or cultural. Many herbicides are currently used by the BLM under their chemical control program. This report considers the impact to ecological receptors (animals and plants) from the use of the herbicide fluridone for the management of aquatic vegetation on BLM lands.

2.2 Herbicide Description

The herbicide-specific use-criteria discussed in this document were obtained from the product label as registered with the USEPA as it applies to the BLM use. Fluridone application rates and methods discussed in this section are based on past and predicted BLM herbicide use and are in accordance with product labels approved by the USEPA. The BLM should be aware of all state-specific label requirements and restrictions. In addition, new USEPA approved herbicide labels may be issued after publication of this report, and BLM land managers should be aware of all newly approved federal, state, and local restrictions on herbicide use when planning vegetation management programs.

Fluridone is a selective systemic herbicide that inhibits carotene production in leaves, which causes the breakdown of chlorophyll—preventing the plant from synthesizing food. This herbicide comes in two formulations: liquid and granule.

Fluridone is being proposed for use in the BLM's Aquatic Vegetation Management program. The majority of application occurs in inland freshwater habitats; diquat is rarely used in marine or estuarine habitats. Applications will be carried out through both aerial and ground application methods. Aerial applications will be made using a fixed-wing airplane or a helicopter. Ground applications will be made on foot, horseback, boat, or using an ATV or truck mounted sprayer applying as a spot or broadcast application. Boat applications will use either a handgun, which will be used to make spot treatments, or a boom, which will be used to make broadcast applications onto the surface of the water or to inject the herbicide under the water surface. The BLM is proposing a typical application rate of 1.0 lbs (lbs) a.i./ac, and the maximum application rate will be 1.3 lbs a.i./ac. Details regarding expected fluridone usage by BLM are provided in Table 2-1 at the end of this section.

2.3 Herbicide Incident Reports

An "ecological incident" occurs when non-target flora or fauna is killed or damaged due to application of a pesticide. When ecological incidents are reported to a state agency or other proper authority, they are investigated and an ecological incident report is generated. The Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) requires product registrants to report adverse effects of their product to the USEPA.

The USEPA OPP manages a database, the EIIS, which contains much of the information in the ecological incident reports. As part of this risk assessment, USEPA was requested to provide all available incident reports in the EIIS that listed fluridone as a potential source of the observed ecological damage.

The USEPA EIIS contained one incident report involving fluridone. Fluridone was listed as the "probable" cause of damage to tomato (*Lycopersicon esculentum*) plants due to direct contact. The type of herbicide use (e.g., registered use, accidental, misuse) and severity of the impact was not specified. There were no other pesticides implicated in this incident report.

TABLE 2-1
BLM Fluridone Use Statistics

Program	Scenario	Vehicle	Method	Used?	Application Rate: Typical (lbs a.i./ac)	Application Rate: Maximum (lbs a.i./ac)
Rangeland				No		
Public-Domain Forest Land				No		
Energy & Mineral Sites				No		
Rights-of-way				No		
Recreation				No		
Aquatic	Aerial	Plane	Fixed Wing	Yes	0.15	1.3
		Helicopter	Rotary	Yes	0.15	1.3
	Ground	Human	Backpack	Yes	0.15	1.3
			Horseback	Yes	0.15	1.3
		ATV	Spot	Yes	0.15	1.3
			Boom/Broadcast	Yes	0.15	1.3
		Truck	Spot	Yes	0.15	1.3
			Boom/Broadcast	Yes	0.15	1.3

The BLM applies fluridone at different typical and maximum rates for four different water bodies: Ponds, Whole Lake/Reservoir, Partial Lakes/Reservoir, and Canals. The lowest typical application rate (Whole Lake/Reservoir) was selected for use as the typical rate and the highest maximum application rate (Partial Lake/Reservoir) was selected for use as the maximum application rate. Application rates are dependent on water depth, which is assumed to be 1 meter (3.28 feet).

3.0 HERBICIDE TOXICOLOGY, PHYSICAL-CHEMICAL PROPERTIES, AND ENVIRONMENTAL FATE

This section summarizes available herbicide toxicology information, describes how this information was obtained, and provides a basis for the LOC values selected for this risk assessment. Fluridone's physical-chemical properties and environmental fate are also discussed.

3.1 Herbicide Toxicology

A review of the available ecotoxicological literature was conducted in order to evaluate the potential for fluridone to negatively effect the environment and to derive TRVs for use in the ERA (provided in italics in sections 3.1.2 and 3.1.3). The process for the literature review and the TRV derivation is provided in the Methods Document (ENSR 2004c). This review generally included a review of published manuscripts and registration documents, information obtained through a Freedom of Information Act (FOIA) request to EPA, electronic databases (e.g., EPA pesticide ecotoxicology database, EPA's on-line ECOTOX database), and other internet sources. This review included both freshwater and marine/estuarine data, although the focus of the review was on the freshwater habitats more likely to occur on BLM lands.

Endpoints for aquatic receptors and terrestrial plants were reported based on exposure concentrations (mg/L and lbs/ac, respectively). Dose-based endpoints (e.g., LD_{50}s) were used for birds and mammals. When possible, dose-based endpoints were obtained directly from the literature. When dosages were not reported, dietary concentration data were converted to dose-based values (e.g., LC_{50} to LD_{50}) following the methodology recommended in USEPA risk assessment guidelines (Sample et al. 1996). Acute TRVs were derived first to provide an upper boundary for the remaining TRVs; chronic TRVs were always equivalent to, or less than (<), the acute TRV. The chronic TRV was established as the highest NOAEL value that was less than both the chronic lowest observed adverse effect level (LOAEL) and the acute TRV. When acute or chronic toxicity data was unavailable, TRVs were extrapolated from other relevant data using an uncertainty factor of 3, as described in the Methods Document (ENSR 2004c).

This section reviews the available information identified for fluridone and presents the TRVs selected for this risk assessment (Table 3-1). Appendix A presents a summary of the fluridone data identified during the literature review. Toxicity data are presented in the units used in the reviewed study. In most cases this applies to the a.i. itself (e.g., fluridone); however, some data correspond to a specific product or applied mixture (e.g., Sonar) containing the a.i. under consideration, and potentially other ingredients (e.g., other a.i. or inert ingredients). This topic, and others related to the availability of toxicity data, is discussed in Section 7.1 of the Uncertainty section. The review of the toxicity data did not focus on the potential toxic effects of inert ingredients (inerts), adjuvants, surfactants, and degradates. Section 7.3 of the Uncertainty section discusses the potential impacts of these constituents in a qualitative manner.

3.1.1 Overview

According to USEPA ecotoxicity classifications presented in registration materials,[1] fluridone has low toxicity to most terrestrial species. Studies conducted with mammals found that acute exposure to fluridone commonly does not cause adverse effects, even to mammals that were exposed to fluridone for longer periods of time or during

[1] Available at http://www.epa.gov/oppefed1/ecorisk_ders/toera_analysis_eco.htm#Ecotox

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 classified as 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).

Fluridone is an herbicide used to control aquatic plants. In the available literature, aquatic plants were not affected by concentrations up to 1 mg/liter (L) (Anderson 1991). Acute and chronic toxicity tests indicate that fluridone causes toxicity to fish species at concentrations < 10 mg/L, and some adverse effect concentrations approach 1 mg/L (Hamelink et al. 1986). 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 (Hamelink et al. 1986), which is equal to the maximum application rate.

3.1.2 Toxicity to Terrestrial Organisms

3.1.2.1 Mammals

Oral toxicity studies conducted in small mammals demonstrated that acute exposure to fluridone typically does not cause adverse effects, even at relatively high dose levels (greater than [>] 10,000 mg a.i./kilogram (kg) body weight (BW) (USEPA 1979). Similarly, acute dermal exposure studies found no adverse effects to rabbits (*Leporidae* spp.) exposed to 5,000 mg a.i./kg BW of fluridone (Eli Lilly 2003). Adverse effects were demonstrated during studies of longer duration. In subchronic oral gavage studies, rabbits exhibited signs of maternal and fetal toxicity (decreased maternal weight, abortions) when dosed with 300 mg a.i./kg BW-day of fluridone during pregnancy (Integrated Risk Information System [IRIS] 2003, MRID 00103302). In this same study, no adverse effects were noted at 125 mg a.i./kg BW-day.

The effects of dietary exposure to fluridone were evaluated in several long-term feeding trails. Rats (*Rattus* spp.) fed fluridone for two years at dietary concentrations as high as 650 parts per million (ppm; equivalent to 25 mg a.i./kg BW-day) exhibited adverse effects, such as decreased BWs and damage to kidneys, testes, and eyes. In this same study, no adverse effects were observed at concentrations of 200 ppm (equivalent to 8 mg a.i./kg BW-day) (IRIS 2003, MRID 00135208).

Based on these findings, the oral LD_{50} (the dose that causes the mortality of 50 percent of the organisms tested; >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.

For large mammals, a one-year feeding trial showed systemic effects (weight loss, increased liver weight, and alkaline phosphatase) in beagle dogs (*Canis familiaris*) fed 150 mg a.i./kg BW-day, while no adverse effects were observed in dogs fed 75 mg a.i./kg BW-day (CA EPA 2000).

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.

3.1.2.2 Birds

Information related to avian exposure to fluridone suggests that acute oral exposure to fluridone is practically non-toxic to birds. The LD_{50} value (the dose that causes the mortality of 50 percent of the organisms tested) was > 2,000 mg/kg BW for bobwhite quail (*Colinus virginianus*) orally administered technical grade fluridone at 95 to 97% a.i.

(USEPA 2003b). In dietary studies, the LC_{50} for bobwhite quail was reported to be > 4,350 ppm of fluridone (equivalent to a dose of 2,627 mg a.i./kg BW-day) (USEPA 1978). For mallards (*Anas platyrhynchos*), the dietary LC_{50} value for fluridone was > 4,540 ppm (equivalent to 454 mg a.i./kg BW-day) for acute exposures (USEPA 1978). In these dietary tests, the test organism was presented with the dosed food for 5 days, with 3 days of additional observations after the dosed food was removed. The endpoint reported for this assay is generally an LC_{50} representing mg a.i./ kg food. For this ERA, the concentration based value was converted to a dose-based value following the methodology presented in the Methods Document (ENSR 2004c). Then the dose-based value was multiplied by the number of days of exposure (generally 5) to result in an LD_{50} value representing the full herbicide exposure over the course of the test. This resulted in LD_{50} values of >13,135 mg a.i./kg BW and >2,270 mg a.i./kg BW for the bobwhite quail and mallard, respectively. Although this study did not provide information regarding % a.i., it was conducted with technical grade fluridone which is generally 95 to 97% a.i.

Similarly, birds fed high concentrations of fluridone in their diets for longer periods of time also showed no adverse effects. Bobwhite quail exposed to 1,000 ppm of fluridone (equivalent to 604 mg a.i./kg BW-day) via the diet for an entire generation did not exhibit signs of systemic or reproductive adverse effects (USEPA 2003b, ACC070932). Similarly, mallards fed 1,000 ppm fluridone (equivalent to 100 mg a.i./kg BW-day) in their diets for an entire generation did not show signs of adverse effects (USEPA 2003b, ACC070932).

Based on these findings, 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.

3.1.2.3 Terrestrial Invertebrates

A standard acute contact toxicity bioassay in honeybees is required for the USEPA pesticide registration process. In this study, fluridone was directly applied to the bee's thorax and mortality was assessed during a 48-hr period. The USEPA reports a NOAEL of 362.58 micrograms (μg)/bee using a 33.3% a.i. technical fluridone product (USEPA 2003b, ACC070932).

In a manufacturer's user's guide (Eli Lilly and Company 2003), data were presented indicating that no mortality has been observed in toxicity tests with earthworms exposed to concentrations as high as 102.6 ppm. This value could not be confirmed by any other source of information reviewed for this document.

Since an LD_{50} was not established in the literature, the NOAEL was multiplied by an uncertainty factor of 3, resulting in a 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. This uncertainty factor was selected based on a review of the application of uncertainty factors (Chapman et al. 1998), and the use of uncertainty factors for this assessment is described in the Methods Document (ENSR 2004c).

3.1.2.4 Terrestrial Plants

Fluridone is sold commercially as Sonar and is primarily used to control aquatic weeds. 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 (*Salix* spp.) and cypress (*Cupressus* spp.), were considered "tolerant," while the tolerance of members of the evening primrose (*Oenothera* and *Camissonia* spp.) and acanthus families (Acanthaceae) was classified as "intermediate".

3.1.3 Toxicity to Aquatic Organisms

3.1.3.1 Fish

In acute toxicity tests, the 96-hour LC_{50} value (i.e., concentration that cause 50% mortality) for rainbow trout (*Oncorhynchus mykiss*) was found to be as low as 4.2 mg/L (Hamelink et al. 1986). Acute toxicity tests conducted on warmwater fish species (bluegill sunfish [*Lepomis macrochirus*], fathead minnow [*Pimephales promelas*], and channel catfish [*Ictalurus punctatus*]) documented 96-hour LC_{50} values as low as 8.2 mg/L (Hamelink et al. 1986; USEPA 2003b, MRID 40098001). Chronic, life-cycle tests on fathead minnow showed adverse effects at fluridone concentrations of 0.96 mg/L, and no adverse effects at concentrations of 0.48 mg /L (Hamelink et al. 1986, USEPA, 2003b, ACC 070934). As a consequence, fluridone is considered to be moderately toxic to fish species. Most studies reviewed, and all studies selected, for TRV derivation for fish were based on products containing at least 97% fluridone.

The lower of the cold- 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.

3.1.3.2 Amphibians

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

3.1.3.3 Aquatic Invertebrates

The toxicity of fluridone was evaluated with several freshwater aquatic invertebrates, including water fleas (e.g., *Daphnia magna*), scuds (*Hyallela* spp.), crayfish (e.g. Astacidae), and chironomids. Acute toxicity was observed in aquatic invertebrates exposure to fluridone concentrations as low as 1.3 mg/L (Hamelink et al. 1986; USEPA 2003b, MRID 40098001). This result is listed for several different studies with % a.i. ranging from 41% to 98% fluridone. Based on the available information, crayfish appear to be less sensitive than other aquatic invertebrates, with LC_{50}s above 16.9 mg a.i./L (Hamelink et al. 1986). NOAELs for several species were derived from chronic or short-term chronic studies. The 21 day reproduction NOAEL for *D. magna* is 0.2 mg/L and the chronic NOAELs for *Gammarus pseudolimnaeus* (60 day growth endpoint) and *Chironomus plumosus* (30 day emergence endpoint) is 0.6 mg/L using a technical grade fluridone at 98 to 99% a.i. (Hamelink et al. 1986).

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.

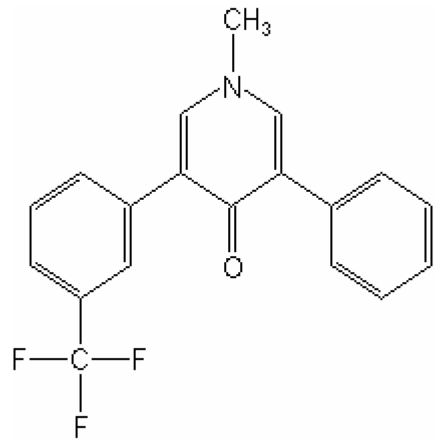
3.1.3.4 Aquatic Plants

Standard toxicity tests were conducted on aquatic plants. The duration of the studies ranged from 37 days to 15 months (McCowen et al. 1979; Anderson 1981; Farone & McNabb 1993; Netherland et al. 1997; Madsen et al. 2002). Study endpoints evaluated included species diversity and growth, measured as biomass and length. Studies failed to detect adverse effects to aquatic macrophytes with fluridone concentrations as high as 1 mg/L (Anderson. 1991). No information was provided regarding the % fluridone contained in the tested product, although it is identified as fluridone [1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]-4(1H)-pyridinone applied at 9.3 liters (L) per hectare (Anderson. 1991).

The NOAEL was set at 1 mg/L. Since no Median Effective Concentration [EC_{50}] values were identified in the reviewed literature, the NOAEL was multiplied by an uncertainty factor of 3 to estimate an EC_{50} of 3 mg./L.

3.2 Herbicide Physical-Chemical Properties

The chemical formula for fluridone is 1-methyl-3-phenyl-5-(α,α,α-trifluoro-m-tolyl)-4-pyridone. At low pH values, some of the fluridone molecules will exist as cations (pKa = 1.7) (Reinert 1989). The chemical structure of fluridone is shown below:



Fluridone Chemical Structure

The physical-chemical properties and degradation rates critical to fluridone's environmental fate are listed in Table 3-2, which presents the range of values encountered in the literature for these parameters. To complete Table 3-2, available USEPA literature on fluridone was obtained either from the Internet or through a FOIA request. Herbicide information that had not been cleared of Confidential Business Information (CBI) was not provided by USEPA as part of the FOIA documents. Additional sources, both on-line and in print, were consulted for information about the herbicide:

- The British Crop Protection Council and The Royal Society of Chemistry. 1994. The Pesticide Manual Incorporating the Agrochemicals Handbook. Tenth Edition. Surrey and Cambridge, United Kingdom.

- California Department of Pesticide Registration (DPR.). 2003. USEPA/OPP Pesticide Related Database. Updated weekly. Available at: http://www.cdpr.ca.gov/docs/epa/epamenu.htm.

- Compendium of Pesticide Common Names. 2003. A website listing all International Organization for Standardization (ISO)-approved names of chemical pesticides. Available at: http://www.hclrss.demon.co.uk.

- Hazardous Substances Data Bank (HSDB). 2002. A toxicology data file on the National Library of Medicines Toxicology Data Network (TOXNET). Available at: http://toxnet.nlm.nih.gov.

- Hornsby, A., R. Wauchope, and A. Herner. 1996. Pesticide Properties in the Environment. P. Howard (ed.). Springer-Verlag, New York.

- Mackay, D., S. Wan-Ying, and M. Kuo-ching. 1997. Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Volume III. Pesticides Lewis Publishers, Chelsea, Minnesota.

- Montgomery, J.H. (ed.). 1997. Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals. Volume V. Pesticide Chemicals. Lewis Publishers, Boca Raton, Florida.

- Tomlin, C (ed.). 1994. The Agrochemicals Desk Reference 2nd Edition. Lewis Publishers, Boca Raton, Florida.

In addition, information was also obtained from the product label for the herbicide Sonar A.S. (SePRO 2002a), the Handbook of Environmental Degradation Rates (Howard et al. 1991), and a fact sheet prepared by Washington State's Department of Health (WA Dept of Health 2000). Relevant papers from the scientific literature were also reviewed. These papers were obtained as part of the literature review to define ecological toxicity endpoints. Values for the foliar half-life and for the foliar washoff coefficient were not found during the review of chemical-physical properties. Thus, as conservative estimates, a foliar half-life of 365 days (no herbicide degradation occurs while on foliage) and a foliar washoff fraction of 1 (all herbicide washes off plant during the first rain) were used in risk assessment calculations. The half-life in pond water was estimated using the physical-chemical properties listed in Table 3-2 and the information reviewed concerning the environmental fate of fluridone in aquatic systems. Values for foliar half-life and foliar washoff fraction were obtained from a database included in the Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) computer model (United States Department of Agriculture [USDA] 1999). Residue rates were obtained from the Kenaga nomogram, as updated (Fletcher et al. 1994). Values selected for use in risk assessment calculations are shown in bold in Table 3-2, presented at the end of this section.

3.3 Herbicide Environmental Fate

The Pesticide Manual reports that biodegradation is the primary fluridone loss mechanism from soils (The British Crop Protection Council and The Royal Society of Chemistry 1994). Soil biodegradation half-lives from 44 days to 192 days have been reported (Howard et al. 1991). The K_{oc}, or organic carbon-water partitioning coefficient, measures the affinity of a chemical to organic carbon relative to water. The higher the K_{oc}, the less soluble in water and the higher affinity for organic carbon, an important constituent of soil particles. Therefore, the higher the K_{oc}, the less mobile the chemical. All but one of the K_{oc} values reviewed ranged from 270 to 6400, indicating fluridone has moderate to no mobility in soils (Table 3-2; Swann et al. 1986). Fluridone sorption increases with clay content, organic matter content, cation exchange capacity, surface area, and decreasing pH (Table 3-2; Weber et al. 1986; Reinert 1989). Protonation at low pH values leads to increased sorption due to cation exchange (Reinert 1989). Fluridone is stable to hydrolysis (USEPA 1986). Based on its Henry's Law constant (the ratio of the chemical's equilibrium distribution between the gas and liquid phases) and vapor pressure, fluridone might volatilize slowly from wet soil surfaces, but volatilization from dry soils would not be expected (Lyman et al. 1990; Mackay et al. 1997; HSDB 2002; Table 3-2). Field half-lives ranging from 21 days to five years have been reported (Table 3-2).

In aquatic systems, photodegradation and biodegradation are important loss pathways for fluridone (The British Crop Protection Council and The Royal Society of Chemistry 1994). As in terrestrial systems, fluridone is stable to hydrolysis and based on the Henry's law constant would volatilize slowly from water bodies (USEPA 1986; Lyman et al. 1990; Mackay et al. 1997; HSDB 2002; Table 3-2). Also, based on reported K_{oc} values, fluridone would be expected to sorb to suspended solids and sediments in aquatic systems (Tomlin 1994). Desorption from sediments followed by photolysis is reported to be a major loss mechanism from aquatic systems (Tomlin 1994). Biodegradation may also remove fluridone from aquatic systems (WA Department of Health 2000). Based on a bioconcentration factor (BCF) of 3.01, fluridone would have little tendency to bioaccumulate in fish (Table 3-2; WA Department of Health 2000). Aquatic dissipation half-lives from 4 to 7 days to 9 months (anaerobic sediments) have been reported (Table 3-2).

TABLE 3-1
Selected Toxicity Reference Values for Fluridone

Receptor	Selected TRV	Units	Duration	Endpoint	Species	Notes
RECEPTORS INCLUDED IN 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	1.3	mg/L	48 h	LC_{50}	midge (*Chironomus*)	multiple studies; 41% - 98% a.i.
Fish	4.25	mg/L	96 h	LC_{50}	rainbow trout	98 – 99% a.i. product
Aquatic plants and algae	3	mg/L	37 d	EC_{50}	American pondweed	extrapolated from NOAEL; no % a.i. listed
Aquatic invertebrates	0.6	mg/L	30 d	NOAEL	midge (*Chironomus*)	98 – 99% a.i. product
Fish	0.48	mg/L	life cycle	NOAEL	fathead minnow	extrapolated from LOAEL; swimming speed
Aquatic plants and algae	1	mg/L	37 d	NOAEL	American pondweed	biomass

TABLE 3-1 (Cont.)
Selected Toxicity Reference Values for Fluridone

Receptor	Selected TRV	Units	Duration	Endpoint	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}

Notes:
Toxicity endpoints for terrestrial animals
LD_{50} - to address acute exposure.
NOAEL - to address chronic exposure.
Toxicity endpoints for terrestrial plants
EC_{25} - to address direct spray, drift, and dust impacts on typical species.
EC_{05} or NOAEL - to address direct spray, drift, and dust impacts on threatened or endangered 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 EC50).
NOAEL - to address chronic exposure.
Value for fish is the lower of the warmwater and coldwater values.

Piscivorous bird TRV = Large bird chronic TRV.
Fish TRV = lower of coldwater and warm water fish TRVs.
Durations:
h - hours
d - days
w - weeks
m - months
y - years
NR – Not reported
Units represent those presented in the reviewed study

TABLE 3-2
Physical-Chemical Properties of Fluridone

Parameter	Value
Herbicide family	Unclassified herbicide (Compendium of Pesticide Common Names 2003).
Mode of action	Inhibits carotene production, which leads to chlorophyll breakdown. (SePRO 2002a).
Chemical Abstract Service number	59756-60-4 (Mackay et al. 1997).
Office of Pesticide Programs chemical code	112900 (DPR 2003).
Chemical name (International Union of Pure and Applied Chemistry [IUPAC])	1-methyl-3-phenyl-5-(α,α,α-trifluoro-m-tolyl)-4-pyridone (Tomlin 1994).
Empirical formula	$C_{19}H_{14}F_3NO$ (Mackay et al. 1997).
Molecular weight (MW)	**329.3** (Tomlin 1994).
Appearance, ambient conditions	White to tan crystalline solid (technical product) (Tomlin 1994).
Acid / Base properties	1.7 (pKa) (Reinert 1989).
Vapor pressure (millimeters of mercury [mmHg] at 25ºC)	< $1x10^{-7}$ (Weber et al. 1986); 9.8 x 10^{-8} (Mackay et al. 1997; Tomlin 1994); 1 x 10^{-7} (Hornsby 1996).
Water solubility (mg/L at 25ºC)	12 (Reinert 1989); 12 (pH 7) (Mackay et al. 1997; Tomlin 1994); 10 (Hornsby et al. 1996).
Log Octanol-water partition coefficient (Log(K_{OW}), unitless)	**1.87** (pH 7, 25ºC) (Tomlin 1994; USEPA 1982); 2.98 (Mackay et al. 1997).
Henry's law constant (atm-m^3/mole)	3.52 x 10^{-6} (Mackay et al. 1997).
Soil / Organic matter sorption coefficient (Kd / K_{oc})	880 (K_{oc}). K_{oc} values from 70 to 2700 obtained for three soils. Kd (Freundlich) / K_{oc} for three soils: 29 / 2700 (Stockton clay, pH 6, organic matter 1.8%, clay 60%, cation exchange capacity 44), 8.6 / 370 (Yolo sandy clay loam, pH 7, organic matter 4.0%, clay 21%, cation exchange capacity 21), and 2.7 / 270 (Hesperia fine sandy loam, pH 7.3, organic matter 1.7%, clay 8.5%, cation exchange capacity 8.5) (Reinert 1989). Freundlich Kd values of 2.6-38 measured on 13 soils (Weber et al. 1986). All values are log(K_{oc}): 2.544-3.04, 1.60 (soil), 2.97-3.39 (pond sediment), 3.36, 2.95 (lake and river sediment), 3.00 (Mackay et al. 1997). For 5 soils, 3-16 (Kd), 350-1100 (K_{oc}) (Tomlin 1994); 1000 (K_{oc}) (Hornsby et al. 1996).
Bioconcentration factor (BCF)	175 samples, 10 fish species: Whole fish BCF for fluridone = **3.01** (West et al. 1983; USEPA 1982).
Field dissipation half-life	6 months to 5 years (USEPA 1982); Ranging from 46-365 days observed for fluridone applied at 1 or 10 μg ai/g soil on sandy loam, sandy clay loam, and peaty loam soils at three different moisture contents (1/4 field capacity, 1/2 field capacity, field capacity, and wet-dry cycling) and two temperature regimes (10ºC and 18-24ºC). Longest half-life generally found for driest condition (Malik 1990); 21 days (Hornsby et al. 1996).
Soil dissipation half-life(1)	Estimated 103 and 27 days (based on dissipation rates of 0.0067 and 0.025 1/day) (Mackay et al. 1997); In a silt-loam > 343 days (pH 7.3, organic matter 2.6%) (Tomlin 1994); Soil aerobic of 44-192 days based on soil die-away test data and field study soil persistence (Howard et al. 1991).
Aquatic dissipation half-life	Fluridone concentration decreased logarithmically with time after Sonar 4AS treatment (liquid) in two NC ponds at 1.0 lb ai/ac and 2.0 lbs ai/ac. Estimated time to reach zero concentration, 64 and 69 days. No observed decrease in a VA pond treated with Sonar 5P, a pelleted formulation, for 53 days (1.0 lb ai/ac). Authors speculate that shading in pond receiving Sonar 5P reduced loss due to photolysis (Langeland and Warner 1986). Half-lives ranging from 4-7 days reported for fluridone in Canadian fish ponds applied at 70, 700, and 5000 μg ai/L (Muir et al. 1980). 5-60 days (av. 20) in 13 ponds treated with SONAR AS: Pond locations FL, TX, TN, CA, WV, IN, MO, MI, NY, and Manotick, Canada. Ponds treated with SONAR 5P (pelleted) reached max fluridone concentration ~ 14 days.

TABLE 3-3 (Cont.)
Physical-Chemical Properties of Fluridone

Parameter	Value
Aquatic dissipation half-life (continued)	after treatment and then fluridone levels declined at a rate similar to ponds treated with SONAR AS. Lake half-lives less than 1 week due to dispersion and dilution as well as degradation and/or adsorption (West et al. 1983); Hydrosoil degradation product only observed in laboratory experiments. In aquatic systems, no degradate observed. Believed desorption followed by photolysis responsible for loss from sediments. In ponds treated with SONAR AS, hydrosoil concentrations reached max after ~ 1 month. In SONAR AP treated ponds, hydrosoil concentrations reached a max within 14 days after treatment. Average half-life for declining phase of fluridone in hydrosoils of SONAR AS treated ponds was 3 months. No fluridone found in treated lake sediments. (West et al. 1983); 21 days in surface water (Mackay et al. 1997); In water (anaerobic) 9 months, (aerobic) about 20 days. (Tomlin 1994; USEPA 1982); Surface water 12-36 days based upon estimated photolysis in water, ground water 88-383 days based upon estimated unacclimated aqueous aerobic biodegradation (Howard et al. 1991).
Hydrolysis half-life	Stable to hydrolysis (USEPA 1986); Stable to hydrolysis, pH = 3 to 9. (Tomlin 1994); > 113 days for 1 μg/ml to hydrolyze in pond water at 4ºC (Mackay et al. 1997).
Photodegradation half-life in water	26 - 55 hours (pH 3 to 9, different fluridone concentrations, pond water, distilled water, no oxygen water) (USEPA 1982); ~ 23 hours in distilled water under > 290 nm light, ~6 hours for 5 ug/ml to degrade in nonsterile pond water under sunlight, ~27 days for 85% of 10 μg/ml to degrade in distilled water and for 85% of 10 ug/ml to degrade in lake water at pH 8.4 both under sunlight (Mackay et al. 1997); 12-36 days based upon measured rate constant for summer sunlight photolysis in distilled water (12 days) and adjusted for relative winter sunlight intensity (36 days) (Howard et al. 1991).
Photodegradation half-life in soil	Not available.
Soil biodegradation half-life	Soil aerobic of 44-192 days based on soil die-away test data and field study soil persistence (Howard et al. 1991).
Aquatic biodegradation half-life	In aquatic systems: 20 days (aerobic), 9 months (anaerobic), 90 days (hydrosoil) (USEPA 1986).
Other degradation rates / half-lives	In hydrosoil > 1 year after initial application and 20 weeks in a retreated pond (Muir et al. 1980).
Foliar half-life	not available.[(2)]
Residue Rate for grass [(3)]	**197 ppm** (maximum) and **36 ppm** (typical) per lb a.i./ac
Residue Rate for vegetation [(4)]	**296 ppm** (maximum) and **35 ppm** (typical)
Residue Rate for insects [(5)]	**350 ppm** (maximum) and **45 ppm** (typical)
Residue Rate for berries [(6)]	**40.7 ppm** (maximum) and **5.4 ppm** (typical)

Notes:
Values presented in bold were used in risk assessment calculations.
(1) Some studies listed in this category may have been performed under field conditions, but insufficient information was provided in the source material to make this determination.
(2) A foliar half-life was not found during our literature review and the available information concerning fluridone's environmental fate did not suggest a value that could be used as a reasonable surrogate. As a conservative estimate, the foliar half-life of fluridone was set at **365 days** for use in risk assessment calculations; that is, fluridone degradation is zero on the time scale of the simulation.
(3) Residue rates selected are the high and mean values for long grass. Fletcher et al. (1994).
(4) Residue rates selected are the high and mean values for leaves and leafy crops. Fletcher et al. (1994).
(5) Residue rates selected are the high and mean values for forage such as legumes. Fletcher et al. (1994).
(6) Residue rates selected are the high and mean values for fruit (includes both woody and herbaceous). Fletcher et al. (1994).

4.0 ECOLOGICAL RISK ASSESSMENT

This section presents a screening-level evaluation of the risks to ecological receptors from potential exposure to the herbicide fluridone. The general approach and analytical methods for conducting the fluridone ERA were based on the USEPA's Guidelines for ERA (hereafter referred to as the "Guidelines;" USEPA 1998).

The ERA is a structured evaluation of all currently available scientific data (exposure chemistry, fate and transport, toxicity, etc.) that leads to quantitative estimates of risk from environmental stressors to non-human organisms and ecosystems. The current Guidelines for conducting ERAs include three primary phases: problem formulation, analysis, and risk characterization. These phases are discussed in detail in the Methods Document (ENSR 2004c) and briefly in the following sub-sections.

4.1 Problem Formulation

Problem formulation is the initial step of the standard ERA process and provides the basis for decisions regarding the scope and objectives of the evaluation. The problem formulation phase for fluridone assessment included:

- definition of risk assessment objectives;
- ecological characterization;
- exposure pathway evaluation;
- definition of data evaluated in the ERA;
- identification of risk characterization endpoints; and
- development of the conceptual model.

4.1.1 Definition of Risk Assessment Objectives

The primary objective of this ERA was to evaluate the potential ecological risks from fluridone to the health and welfare of plants and animals and their habitats. This analysis is part of the process used by the BLM to determine which of the proposed treatment alternatives evaluated in the EIS should be used on BLM-managed lands.

An additional goal of this process was to provide risk managers with a tool that develops a range of generic risk estimates that vary as a function of site conditions. This tool primarily consists of Excel spreadsheets (presented in the Ecological Risk Assessment Worksheets; Appendix B), which may be used to calculate exposure concentrations and evaluate potential risks in the risk assessment. A number of the variables included in the worksheets can be modified by BLM land managers for future evaluations.

4.1.2 Ecological Characterization

As described in Section 2.2, fluridone is used by the BLM for vegetation control in Aquatic program. The proposed BLM program involves the general use and application of herbicides on public lands in 17 western states in the continental US and Alaska. These applications have the potential to affect organisms in a wide variety of ecological habitats that could include: deserts and prairie land, and many others. It is not feasible to characterize all of the potential affected habitats within this report; however, this ERA was designed to address generic receptors, including RTE species (see Section 6.0) that could occur within a variety of habitats.

4.1.3 Exposure Pathway Evaluation

The following ecological receptor groups were evaluated:

- terrestrial animals;
- non-target terrestrial plants; and
- aquatic species (fish, invertebrates, and non-target aquatic plants).

These groups of receptor species were selected for evaluation because they: (1) are potentially exposed to herbicides within BLM management areas (directly or indirectly); (2) are likely to play key roles in site ecosystems; (3) have complex life cycles; (4) represent a range of trophic levels; and (5) are surrogates for other species likely to be found on BLM-managed lands.

The exposure scenarios considered in the ERA were primarily organized by potential exposure pathways. In general, the exposure scenarios describe how a particular receptor group may be exposed to the herbicide as a result of a particular exposure pathway. These exposure scenarios were developed to address potential acute and chronic impacts to receptors under a variety of exposure conditions that may occur within BLM-managed lands. Fluridone is an aquatic herbicide; therefore, as discussed in detail in the Methods Document (ENSR 2004c), the following exposure scenarios were considered:

- direct contact with the herbicide or a contaminated waterbody;
- indirect contact with contaminated foliage;
- ingestion of contaminated food items;
- off-site drift of spray to terrestrial areas; and
- accidental spills to waterbodies.

Two generic waterbodies were considered in this ERA: 1) a small pond (1/4 acre pond of 1 meter [m] depth, resulting in a volume of 1,011,715 L) and 2) a small stream representative of Pacific Northwest low-order streams that provide habitat for critical life-stages of anadromous salmonids. The stream size was established at 2 m wide and 0.2 m deep with a mean water velocity of approximately 0.3 meters per second, resulting in a base flow discharge of 0.12 cubic meters per second (cms).

4.1.4 Definition of Data Evaluated in the ERA

Herbicide concentrations used in the ERA were based on typical and maximum application rates provided by the BLM (Table 2-1). These application rates were used to predict herbicide concentrations in various environmental media (e.g., soils, water). For the aquatic herbicides these calculations were fairly straightforward and generally required only simple algebraic calculations (e.g., water concentrations from direct aerial spray). However, off-site herbicide transport due to spray drift was modeled using the AgDRIFT® computer model. AgDRIFT® Version 2.0.05 (SDTF 2002) 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).

4.1.5 Identification of Risk Characterization Endpoints

Assessment endpoints and associated measures of effect were selected to evaluate whether populations of ecological receptors are potentially at risk from exposure to proposed BLM applications of fluridone. The selection process is discussed in detail in Methods Document (ENSR 2004c), and the selected endpoints are presented below (impacts to RTE species are discussed in more detail in Section 6.0).

<u>Assessment Endpoint 1: Acute mortality to mammals, birds, invertebrates, non-target plants</u>

- **Measures of Effect** included median lethal effect concentrations (e.g., LD_{50} and LC_{50}) from acute toxicity tests on target organisms or suitable surrogates. To add conservatism to the RTE assessment, lowest available germination NOAELs were used to evaluate non-target RTE plants, and LOCs for RTE species were lower than for typical species.

Assessment Endpoint 2: Acute mortality to fish, aquatic invertebrates, and aquatic plants

- **Measures of Effect** included median lethal effect concentrations (e.g., LC_{50} and EC_{50}) from acute toxicity tests on target organisms or suitable surrogates (e.g., data from other coldwater fish to represent threatened and endangered salmonids). As with terrestrial species, lowest available germination NOAELs were used to evaluate non-target RTE plants, and LOCs for RTE species were lower than for typical species.

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 observable adverse effect level (NOAEL) for both terrestrial and aquatic organisms. Depending on data available for a given herbicide, chronic endpoints reflect either individual impacts (e.g., 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 indications of change of parr [freshwater stage salmonids] to adulthood), thermoregulation (i.e., ability to maintain body temperature), and migratory behavior, if such data were available.

Assessment Endpoint 4: Adverse indirect effects on the survival, growth, or reproduction of salmonid fish

- **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 fluridone on salmonids and their habitat, only qualitative estimates of indirect effects were possible. 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). Similar approaches are already being applied by USEPA OPP for Endangered Species Effects Determinations and Consultations (http://www.epa.gov/oppfead1/endanger/effects).

4.1.6 Development of the Conceptual Model

The fluridone conceptual model (Figure 4-1) is presented as a series of working hypotheses about how fluridone might pose hazards to the ecosystem and ecological receptors. The conceptual model indicates the possible exposure pathways for the herbicide as well as the types receptors that were evaluated for each exposure pathway. Figure 4-2 presents the trophic levels and receptor groups evaluated in the ERA.

The conceptual model for herbicide application on BLM lands is designed to display potential herbicide exposure through several pathways, although all pathways may not exist for all locations. The exposure pathways and ecological receptor groups considered in the conceptual model are also described in Section 4.1.3.

The aquatic herbicide conceptual model (Figure 4-1) presents essentially three mechanisms for the release of an herbicide into the environment: direct spray (either accidental or during normal applications), drift, and accidental spills. These release mechanisms may occur as the aquatic herbicide is applied to the intended pond area from a boat or from the shoreline. The aquatic herbicide considered in this risk assessment is not applied to streams.

As indicated in the conceptual model figure, accidental direct spray of terrestrial receptors may occur when the aquatic herbicide is being applied from a boat. This may result in herbicide exposure for wildlife or non-target terrestrial plants if they are directly sprayed during the application. Terrestrial wildlife may also be exposed to the herbicide by brushing against sprayed vegetation or by ingesting contaminated food items.

Direct spray of non-target receptors may also occur during shoreline applications of the aquatic herbicide. Herbicides may be applied to either a pond (normal application) or a stream (accidental application) resulting in exposure of aquatic plants, fish, and aquatic invertebrates to impacted water. Piscivorous birds may also be impacted by ingesting contaminated fish from an exposed pond.

During normal application of aquatic herbicides, it is possible for a portion of the herbicide to drift outside of the treatment area and deposit onto non-target terrestrial receptors. This may occur during terrestrial or aerial applications and may result in exposure of non-target terrestrial plants to the aquatic herbicide.

Accidental spills may also occur during normal herbicide applications. Spills represent the worst-case transport mechanism for herbicide exposure. An accidental spill to a waterbody would result in exposure for aquatic plants, fish, and aquatic invertebrates to impacted water.

4.2 Analysis Phase

The analysis phase of an ERA consists of two principal steps: the characterization of exposure and the characterization of ecological effects. The exposure characterization described the source, fate, and distribution of the herbicides in various environmental media. All EECs are presented in Appendix B. The ecological effects characterization consisted of compiling exposure-response relationships from all available toxicity studies on the herbicide.

4.2.1 Characterization of Exposure

The BLM uses herbicides in the Aquatics program with several different application methods (e.g., boat, plane, helicopter). In order to assess the potential ecological impacts of these herbicide uses, a variety of exposure scenarios were considered. These scenarios, which were selected based on actual BLM herbicide usage under a variety of conditions, are described in Section 4.1.3.

When considering the exposure scenarios and the associated predicted concentrations, it is important to recall that the frequency and duration of the various scenarios are not equal. For example, exposures associated with accidental spills will be very rare, while ingestion of contaminated vegetation may be more common. Similarly, direct spray events will be short-lived while ingestion of fish from a contaminated pond may occur over weeks or months following application. The ERA has generally treated these differences in a conservative manner (i.e., potential risks are presented despite their likely rarity and/or transience). Thus, tables and figures summarizing RQs may present both relatively common and very rare exposure scenarios. Additional perspective on the frequency and duration of exposures are provided in the narrative below.

As described in Section 4.1.3, the following ecological receptor groups were selected to address the potential risks due to unintended exposure to fluridone: terrestrial animals, terrestrial plants, and aquatic species. A set of generic terrestrial animal receptors, listed below, were selected to cover a variety of species and feeding guilds that might be found on BLM-managed lands. Unless otherwise noted, receptor BWs were selected from the *Wildlife Exposure Factors Handbook* (USEPA 1993a). This list includes surrogate species, although not all of these surrogate species will be present within each actual application area:

- A pollinating insect with a BW of 0.093 grams (g). The honeybee (*Apis mellifera*) was selected as the surrogate species to represent pollinating insects. This BW was based on the estimated weight of receptors required for testing in 40CFR158.590.

- A small mammal with a BW of 20 g that feeds on fruit (e.g., berries). The deer mouse (*Peromyscus maniculatus*) was selected as the surrogate species to represent small mammalian omnivores consuming berries.

- A large mammal with a BW of 70 kg that feeds on plants. The mule deer (*Odocolieus hemionus*) was selected as the surrogate species to represent large mammalian herbivores, including wild horses and burros (Hurt and Grossenheider 1976).

- A large mammal with a BW of 12 kg that feeds on small mammals. The coyote (*Canis latrans*) was selected as the surrogate species to represent large mammalian carnivores (Hurt and Grossenheider 1976).

- A small bird with a BW of 80 g that feeds on insects. The American robin (*Turdus migratorius*) was selected as the surrogate species to represent small avian insectivores.

- A large bird with a BW of approximately 3.5 kg that feeds on vegetation. The Canada goose (*Branta canadensis*) was selected as the surrogate species to represent large avian herbivores.

- A large bird with a BW of approximately 5 kg that feeds on fish. The Northern subspecies of the bald eagle (*Haliaeetus leucocephalus alascanus*) was selected as the surrogate species to represent large avian piscivores (Brown and Amadon 1968[2]).

Potential impacts to non-target terrestrial plants could not be evaluated quantitatively for fluridone due to a lack of terrestrial plant toxicity data. Aquatic exposure pathways were evaluated using fish, aquatic invertebrates, and non-target aquatic plants in a pond or stream habitat (as defined in Section 4.1.3). Rainbow trout and walleyes (*Stizostedion vitreum*) were surrogates for fish, the water flea and water scud were surrogates for aquatic invertebrates, and non target aquatic plants and algae were represented by giant duckweed (*Spirodela polyrhiza*).

Section 3.0 of the Methods Document (ENSR 2004b) presents the details of the exposure scenarios considered in the risk assessments. The following sub-sections describe the scenarios that were evaluated for fluridone.

4.2.1.1 Direct Spray

Plant and wildlife species may be unintentionally impacted during normal application of an aquatic herbicide as a result of a 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 (direct spray of waterbody) or outside of the application area (consumption of terrestrial prey items accidentally sprayed by aquatic herbicide). Generally, impacts outside of the intended application area are accidental exposures and are not typical of BLM application practices. The following direct spray scenarios were evaluated:

<u>Exposure Scenarios Within the Application Area</u>

- Direct Spray to Pond (normal application)
- Consumption of Fish From Contaminated Pond

<u>Exposure Scenarios Outside the Application Area</u>

- Accidental Direct Spray of Terrestrial Wildlife
- Accidental Direct Spray of Non-Target Terrestrial Plants
- Indirect Contact With Foliage After Accidental Direct Spray

[2] As cited on the Virginia Tech Conservation Management Institute Endangered Species Information System website (http://fwie.fw.vt.edu/WWW/esis/).

- Ingestion of Prey Items Contaminated by Accidental Direct Spray
- Accidental Direct Spray Over Stream (fluridone is not indicated for use in streams)

4.2.1.2 Off-site Drift

During normal application of aquatic herbicides, it is possible for a portion of the herbicide to drift outside of the treatment area and deposit onto non-target terrestrial receptors. To simulate off-site herbicide transport as spray drift, AgDRIFT® software was used to evaluate a number of possible scenarios. Based on actual BLM uses of fluridone, ground applications were modeled using a low- or high-placed boom and aerial application was modeled from both a helicopter and a plane over non-forested land. Ground applications were modeled using either a high boom (spray boom height set at 50 inches above the ground) or a low boom (spray boom height set at 20 inches above the ground). Deposition rates vary by the height of the application (the higher the application height, the greater the off-target drift). Drift deposition was modeled at 25, 100, and 900 feet (ft) from the application area for ground applications and 100, 300, and 900 ft from the application area for aerial applications. The AgDRIFT® model determined the fraction of the application rate that is deposited off-site without considering herbicide degradation. Impacts to off-site terrestrial plants were evaluated based on deposition modeled by AgDRIFT.®

4.2.1.3 Accidental Spill to Pond

To represent worst-case potential impacts to the pond, two spill scenarios were considered. These consist of 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 1/4 acre, 1 meter deep pond.

4.2.2 Effects Characterization

The ecological effects characterization phase entailed a compilation and analysis of the stressor-response relationships and any other evidence of adverse impacts from exposure to each herbicide. For the most part, available data consisted of the toxicity studies conducted in support of USEPA pesticide registration described in Section 3.1. TRVs selected for use in the ERA are presented in Table 3-1. Appendix A presents the full set of toxicity information identified for fluridone.

In order to address potential risks to ecological receptors, RQs were calculated by dividing the EEC for each of the previously described scenarios by the appropriate TRV presented in Table 3-1. An RQ was calculated by dividing the EEC for a particular scenario by 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 to LOCs established by the USEPA OPP to assess potential risk to non-target organisms. Table 4-1 presents the LOCs established for this assessment. 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 through a restricted use designation.
- **Acute endangered species** – the potential for acute risk to endangered species is high.
- **Chronic risk -** the potential for chronic risk is high.

Additional uncertainty factors may also be applied to the standard LOCs to reflect uncertainties inherent in extrapolating from surrogate species toxicity data to obtain RQs (see Sections 6.3 and 7.0 for a discussion of uncertainty). A "chronic endangered 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 the conservative two-fold difference in contaminant sensitivity between RTE and surrogate test fishes (Sappington et al. 2001). Risk quotients predicted for acute scenarios (e.g., direct spray, accidental spill) were compared to the three acute LOCs, and the RQs predicted for chronic scenarios (e.g., long term ingestion) were compared to the two chronic LOCs. If all RQs were less than the most conservative LOC for a particular receptor, comparisons against other, more elevated LOCs were not necessary.

The RQ approach used in this ERA provides a conservative measure of the potential for risk based on a "snapshot" of environmental conditions (i.e., rainfall, slope) and receptor assumptions (i.e., BW, ingestion rates). Sections 6.3 and 7.0 discuss several of the uncertainties inherent in the RQ methodology.

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 (set at 1) for all scenarios. The plant toxicity endpoints were selected to provide extra protection to the RTE species. In the direct spray and spray drift scenarios, the selected toxicity endpoints were an effect concentration (EC_{25}) for "typical" species and a NOAEL for RTE species. Potential impacts to non-target terrestrial plants from fluridone could not be evaluated quantitatively due to a lack of terrestrial plant toxicity data.

The evaluation of RTE terrestrial wildlife and aquatic species is 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 LOC was lowered for RTE species.

4.3 Risk Characterization

The ecological risk characterization integrates the results of the exposure and effects phases (i.e., risk analysis), and provides comprehensive estimates of actual or potential risks to ecological receptors. Risk quotients are summarized in Tables 4-2 to 4-3 and presented graphically in Figures 4-3 to 4-6. The results are discussed below for each of the evaluated exposure scenarios.

Box plots are used to graphically display the range of RQs obtained from evaluating each receptor and exposure scenario combination (Figures 4-3 to 4-6). These plots illustrate how RQ data are distributed about the mean and their relative relationships with LOCs. Outliers (data points outside the 90^{th} or 10^{th} percentile) were not discarded in this ERA; all RQ data presented in these plots were included in the risk assessment.

4.3.1 Direct Spray

As described in Section 4.2.1, potential impacts from direct spray were evaluated for exposure that could occur within the aquatic application area (direct spray of pond during normal application, consumption of fish from contaminated pond) and outside the intended application area (accidental direct spray of terrestrial wildlife and non-target terrestrial plants, indirect contact with foliage, ingestion of contaminated prey items, accidental direct spray over stream). Table 4-2 presents the RQs for the following scenarios: direct spray of terrestrial wildlife, indirect contact with foliage after direct spray, ingestion of contaminated prey items by terrestrial wildlife, direct spray of non-target terrestrial plants, and direct spray over a pond or stream. Figures 4-3 to 4-6 present graphic representations of the range of RQs and associated LOCs.

4.3.1.1 Terrestrial Wildlife

Acute RQs for terrestrial animals (Figure 4-3) were below the most conservative LOC of 0.1 (acute endangered species) for all scenarios. Only one chronic exposure scenario exceeded the terrestrial animal chronic LOC. 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 under most conditions.

4.3.1.2 Non-target Plants – Terrestrial and Aquatic

No toxicity data was identified for non-target terrestrial plant species; therefore, a quantitative evaluation is not possible. However, the ecological incident report described in Section 2.3 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. The incident report and the user's guide both indicate that fluridone may cause negative impacts to terrestrial plants (e.g., tomatoes, grasses, sedges), but that shoreline plants are more tolerant. It is these more tolerant shoreline plants that are more likely to come in contact with fluridone during normal pond applications. The Sonar labels (SePRO 2002a,b,c; 2003) warn against using treated water for irrigation purposes for seven to thirty days after treatment. Even at the low fluridone concentrations used to treat milfoil, some terrestrial plants may be sensitive to fluridone if they are watered with treated lake water.

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. According to the Sonar user's guide (Eli Lilly and Company 2003), many native aquatic plants are tolerant to fluridone and show little or no impact following treatment. However, the target nuisance species, hydrilla, Eurasian watermilfoil, and curlyleaf pondweed, are highly susceptible to this herbicide.

4.3.1.3 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 endangered species (0.05). However, this value is below the acute high risk LOC, suggesting minimal risk to non-endangered species.

Accidental direct spray of fluridone over the stream results in elevated acute and chronic RQs (Figure 4-5 and 4-6). 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 endangered 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 endangered species (0.5) and the LOC for chronic risk (1).

These results indicate there is potential for risk to aquatic species, especially endangered species, in a stream sprayed with fluridone. It may be noted that these spray scenarios are very conservative because they are instantaneous concentrations and do not consider flow, adsorption to particles, or degradation that may occur over time. In addition, this scenario is not likely to occur as fluridone is reserved for use in ponds.

4.3.1.4 Piscivorous Birds

Risk to piscivorous birds (Figure 4-3) was assessed by evaluating impacts from consumption of fish from a pond impacted by normal application of fluridone. 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.

4.3.2 Off-site Drift to Non-target Terrestrial Plants

As described in Section 4.2.1, AgDRIFT® software was used to evaluate a number of possible scenarios in which a portion of the applied herbicide drifts outside of the treatment area and deposits onto non-target receptors. Ground applications of fluridone were modeled using both a low- and high-placed boom (spray boom height set at 20 and 50 inches above the ground, respectively), and aerial applications were modeled from both a helicopter and a plane over non-forested lands. Drift deposition was modeled at 25, 100, and 900 ft from the application area for ground applications and 100, 300, and 900 ft from the application aerial applications area.

As described previously, no toxicity data was identified for non-target terrestrial plant species, therefore a quantitative evaluation of this scenario is not possible. However, the ecological incident report described in Section 2.3 suggests that impacts to terrestrial plants are possible due to unintended contact with fluridone. As described in Section 4.3.1.2, the Sonar user's guide (Eli Lilly and Company 2003) and labels (SePRO, 2002a,b,c; SePRO 2003) indicate the potential for impact to non-target terrestrial plants.

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. Table 4-3 presents the soil deposition predicted as a result of off-site drift compared to herbicide concentrations resulting from the typical and maximum application rates considered in the direct spray scenarios (Section 4.3.1.2). 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. In general, off-site drift modeled using the typical application rate was < 10% of the typical application rate used in the direct spray scenario. Off-site drift modeled using the maximum application rate was < 1% of the maximum application rate used in the direct spray scenario. This table indicates the significant reduction in deposition and associated risks that occurs with off-site drift relative to direct accidental spray. It may be noted that a significantly greater proportion of the herbicide is deposited due to drift from aerial applications than from ground applications.

4.3.3 Accidental Spill to Pond

As described in Section 4.2.1, two spill scenarios were considered. These consist of a truck and a helicopter spilling entire loads (200 gal spill and 140 gal spill, respectively) of herbicide mixed for the maximum application rate into the 1/4 acre, 1 meter deep pond. The herbicide concentration in the pond was the instantaneous concentration at the moment of the spill. The volume of the pond was determined and the volume of herbicide in the truck and helicopter, respectively, were mixed into the pond volume.

Risk quotients for the truck spill scenario (Table 4-2) were 1.10 for fish, 3.58 for aquatic invertebrates (Figure 4-5 and 4-6), and 1.56 for non-target aquatic plants (Figure 4-4). 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. 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 risks to fish, aquatic invertebrates, and non-target aquatic plants were indicated for the truck and helicopter spills mixed for the maximum application rate.

4.3.4 Potential Risk to Salmonids from Indirect Effects

In addition to direct effects of herbicides on salmonids and other fish species in stream habitats (i.e., mortality due to herbicide concentrations in surface water), reduction in vegetative cover or food supply may indirectly impact individuals or populations. No literature studies were identified that explicitly evaluated the indirect effects of fluridone to salmonids and their habitat; therefore, only qualitative estimates of indirect effects are possible. These estimates were accomplished by discussing predicted impacts to prey items and vegetative cover in the accidental direct spray over the stream scenario evaluated above. The only stream evaluation conducted for this risk assessment was the accidental direct spray scenario, since fluridone is not proposed for use in streams. An evaluation of impacts to non-target terrestrial plants was also included as part of the discussion of vegetative cover within the riparian zone. Prey items for salmonids and other potential RTE species may include other fish species, aquatic invertebrates, or aquatic plants. Additional discussion of RTE species is provided in Section 6.0.

4.3.4.1 Qualitative Evaluation of Impacts to Prey

Fish species were evaluated directly in the ERA using acute and chronic TRVs based on the most sensitive warm- or coldwater species identified during the literature search. Several laboratory studies with salmonids (rainbow trout) were identified in the literature and considered in the selection of the fish TRVs (Appendix A). The chronic fish TRV was based on a warm-water species, the fathead minnow. The acute fish TRV was based the rainbow trout, a salmonid. The inclusion of salmonid data in the TRV derivation reduced the uncertainties inherent in assessing potential indirect impacts to salmonids.

Aquatic invertebrates were also evaluated directly using acute and chronic TRVs based on the most sensitive aquatic invertebrate species. RQs in excess of the acute LOCs for fish and aquatic invertebrates were observed for the accidental direct spray scenario. However, this is an extremely conservative scenario in which it is assumed that a stream is accidentally directly sprayed by an aquatic herbicide intended for a pond. This is unlikely to occur as a result of BLM practices and represents a worst-case scenario. In addition, stream flow would be likely to dilute the herbicide concentration and reduce potential impacts, but no reduction in herbicide concentration is calculated as a result of stream flow.

The only stream evaluation conducted for this risk assessment was an accidental direct spray scenario and may overestimate risk to aquatic stream receptors. However, this conservative evaluation predicts that fish and 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.

4.3.4.2 Qualitative Evaluation of Impacts to Vegetative Cover

A qualitative evaluation of indirect impacts to salmonids due to destruction of riparian vegetation and reduction of available cover was made by considering impacts to terrestrial and aquatic plants. 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.

Although terrestrial plants were not specifically evaluated in the stream scenarios of the ERA, a reduction in riparian cover has the potential to indirectly impact salmonids within the stream. However, terrestrial plant TRVs were not available for this evaluation. A review of incident reports and the manufacturer’s user’s guide (Eli Lilly and Company 2003) indicate that shoreline plant species are generally tolerant of fluridone exposures. However, the user’s guide (Eli Lilly and Company 2003) and labels (SePRO, 2002a,b,c; SePRO 2003) do indicate the potential for impact to non-target terrestrial plants. Therefore, it is uncertain whether or not a reduction in riparian cover is likely.

4.3.4.3 Conclusions

This qualitative evaluation indicates that salmonids may be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates). However, this evaluation is based on worst-case accidental exposure scenarios that are not likely to occur as a result of BLM management practices. Reducing the application rate and avoidance of accidental application on non-target areas would reduce the likelihood of these impacts. A reduction in aquatic vegetative cover was not predicted. Based on a lack of toxicity data, it is unknown whether a reduction in terrestrial plant cover would occur.

In addition, the effects of aquatic herbicides in water are expected to be relatively transient and stream flow is likely to reduce herbicide concentrations over time. Only very persistent pesticides would be expected to have effects beyond the year of their application. An OPP report on the impacts of a terrestrial herbicide on salmonids indicated that if a listed salmonid was not present during the year of application, there would likely be no concern (Turner 2003). Therefore, it is expected that potential adverse impacts to food and aquatic cover would not be occur beyond the season of application.

TABLE 4-1
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 Endangered 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 Endangered 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 Endangered Species	EEC/LC_{50} or EC_{50}	0.05
	Chronic Risk	EEC/NOAEL	1
	Chronic Risk, Endangered Species	EEC/NOAEL	0.5
Plants [3]			
Terrestrial Plants	Acute High Risk	EEC/EC_{25}	1
	Acute Endangered Species	EEC/NOAEL	1
Aquatic Plants	Acute High Risk	EEC/EC_{50}	1
	Acute Endangered Species	EEC/NOAEL	1

[1] Estimated Environmental Concentration (EEC) is in mg $_{prey}$/kg $_{body\ weight}$ for acute scenarios and mg $_{prey}$/kg $_{body\ weight}$/day for chronic scenarios.
[2] EEC is in mg/L.
[3] EEC is in lbs/ac.

TABLE 4-2
Risk Quotients for Direct Spray and Spill Scenarios

Terrestrial Animals	Typical Application Rate	Maximum Application Rate
Direct Spray of Terrestrial Wildlife		
Small mammal - 100% absorption	1.95E-04	1.69E-03
Pollinating insect - 100% absorption	2.03E-03	1.76E-02
Small mammal - 1st order dermal adsorption	5.54E-06	4.80E-05
Indirect Contact With Foliage After Direct Spray		
Small mammal - 100% absorption	1.95E-05	1.69E-04
Pollinating insect - 100% absorption	2.03E-04	1.76E-03
Small mammal - 1st order dermal adsorption	5.54E-07	4.80E-06
Ingestion of Prey Items Contaminated by Direct Spray		
Small mammalian herbivore - acute exposure	2.90E-05	1.89E-03
Small mammalian herbivore - chronic exposure	3.40E-02	**2.22E+00**
Large mammalian herbivore - acute exposure	1.86E-04	8.81E-03
Large mammalian herbivore - chronic exposure	9.27E-03	4.40E-01
Small avian insectivore - acute exposure	2.33E-04	1.57E-02
Small avian insectivore - chronic exposure	4.66E-03	3.14E-01
Large avian herbivore - acute exposure	5.67E-04	4.16E-02
Large avian herbivore - chronic exposure	1.18E-02	8.68E-01
Large mammalian carnivore - acute exposure	1.21E-04	1.05E-03
Large mammalian carnivore - chronic exposure	1.87E-04	1.62E-03

Semi-Aquatic Wildlife	Typical Application Rate	Maximum Application Rate
Ingestion of Prey Items Contaminated by Normal Application to Pond		
Avian piscivore – chronic exposure	4.00E-05	3.47E-04

TABLE 4-2 (Cont.)
Risk Quotients for Direct Spray and Spill Scenarios

Terrestrial Plants	Typical Species		Rare, Threatened, and Endangered Species	
	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate
Direct Spray of Non-Target Terrestrial Plants				
Accidental direct spray	NC	NC	NC	NC

Aquatic Species	Fish		Aquatic Invertebrates		Non-Target Aquatic Plants	
	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate
Direct Spray Over Pond – Normal Application						
Acute	3.96E-03	3.43E-02	1.29E-02	**1.12E-01**	5.60E-03	4.86E-02
Chronic	3.36E-02	2.91E-01	2.80E-02	2.43E-01	1.68E-02	1.46E-01
Direct Spray Over Stream – Accidental Spray						
Acute	1.98E-02	**1.71E-01**	**6.47E-02**	**5.60E-01**	2.80E-02	2.43E-01
Chronic	1.68E-01	**1.46E+00**	1.40E-01	**1.21E+00**	8.41E-02	7.29E-01
Accidental spill						
Truck spill into pond	--	**1.10E+00**	--	**3.59E+00**	--	**1.55E+00**
Helicopter spill into pond	--	**3.84E+00**	--	**1.26E+01**	--	**5.44E+00**

NC - Not calculated. RQs could not be calculated due to a lack of terrestrial plant toxicity testing. Only a qualitative evaluation was possible.
Shading and boldface indicates terrestrial animal acute RQs greater than 0.1 (LOC for acute risk to endangered species - most conservative).
Shading and boldface indicates terrestrial animal chronic RQs greater than 1 (LOC for chronic risk).
Shading and boldface indicates plant RQs greater than 1 (LOC for all plant risks).
Shading and boldface indicates acute RQs greater than 0.05 for fish and invertebrates (LOC for acute risk to endangered species - most conservative).
Shading and boldface indicates chronic RQs greater than 0.5 for fish and invertebrates (LOC for chronic risk to endangered species).
RTE – Rare, threatened, and endangered.
-- indicates the scenario was not evaluated

TABLE 4-3
Comparison of Soil Deposition Due to Off-Site Drift and Direct Spray

Mode of Application	Application Height or Type	Distance From Receptor (ft)	Soil Deposition: Typical Application Rate lbs a.i./ac	Typical Application Rate %	Maximum Application Rate lbs a.i./ac	Maximum Application Rate %
OFF-SITE DRIFT (modeled in AgDRIFT)						
Plane	Non-Forested	100	3.57E-02	[23.8]	1.13E-02	[0.87]
Plane	Non-Forested	300	1.78E-02	[11.9]	5.94E-03	[0.46]
Plane	Non-Forested	900	5.92E-03	[3.94]	2.80E-03	[0.22]
Helicopter	Non-Forested	100	3.57E-02	[23.8]	9.42E-03	[0.72]
Helicopter	Non-Forested	300	8.92E-03	[5.95]	4.62E-03	[0.36]
Helicopter	Non-Forested	900	4.75E-03	[3.16]	2.01E-03	[0.15]
Ground	Low Boom	25	5.15E-03	[3.43]	9.13E-04	[0.07]
Ground	Low Boom	100	1.82E-03	[1.21]	5.01E-04	[0.039]
Ground	Low Boom	900	2.79E-04	[0.19]	9.67E-05	[0.007]
Ground	High Boom	25	8.51E-03	[5.67]	1.47E-03	[0.11]
Ground	High Boom	100	2.86E-03	[1.91]	7.73E-04	[0.059]
Ground	High Boom	900	3.58E-04	[0.24]	1.23E-04	[0.009]
DIRECT SPRAY						
			1.50E-01		1.30E+00	

Value in brackets indicates percentage of the direct spray application rate that is deposited due to off-site drift.

FIGURE 4-1. Conceptual Model for Aquatic Herbicides.

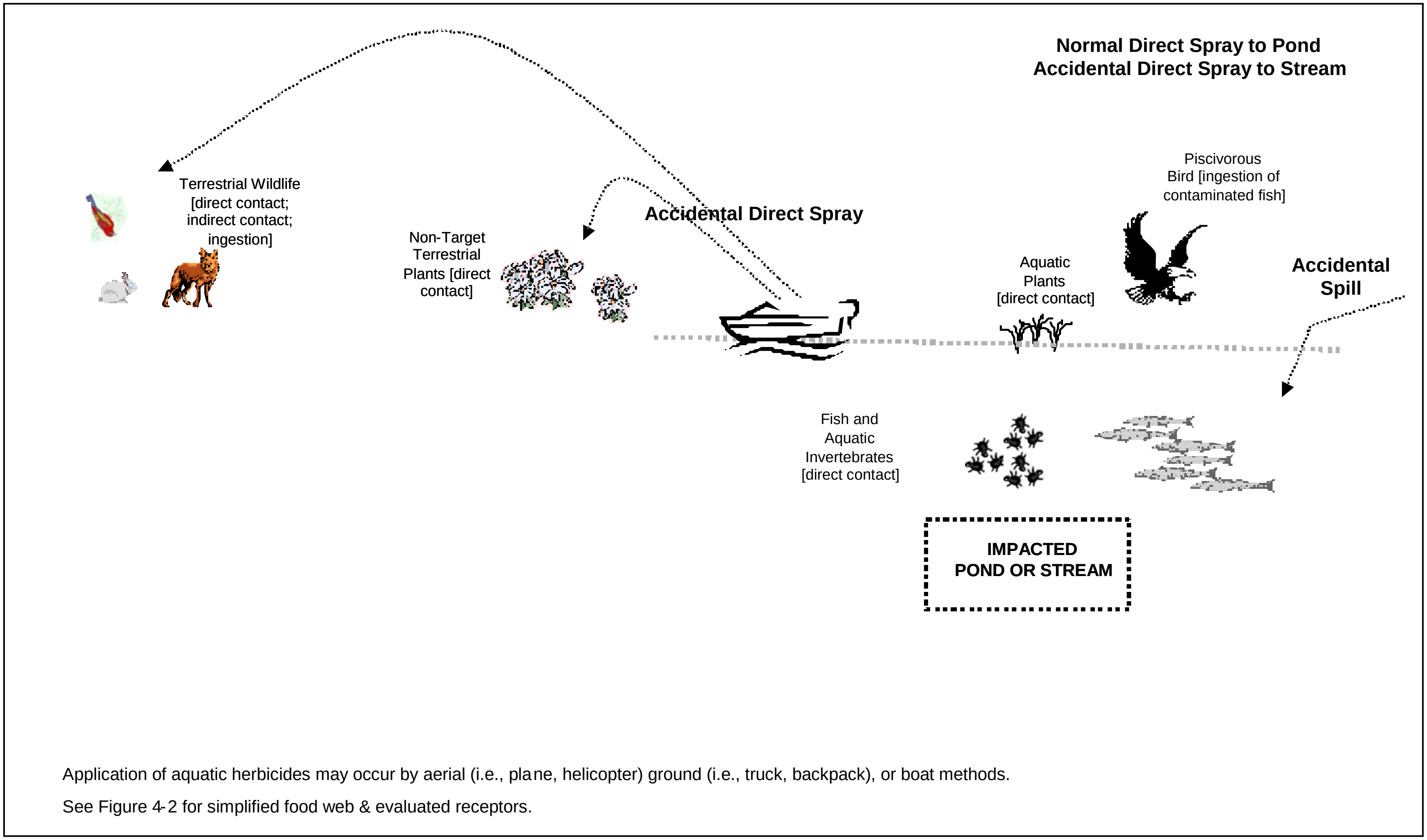
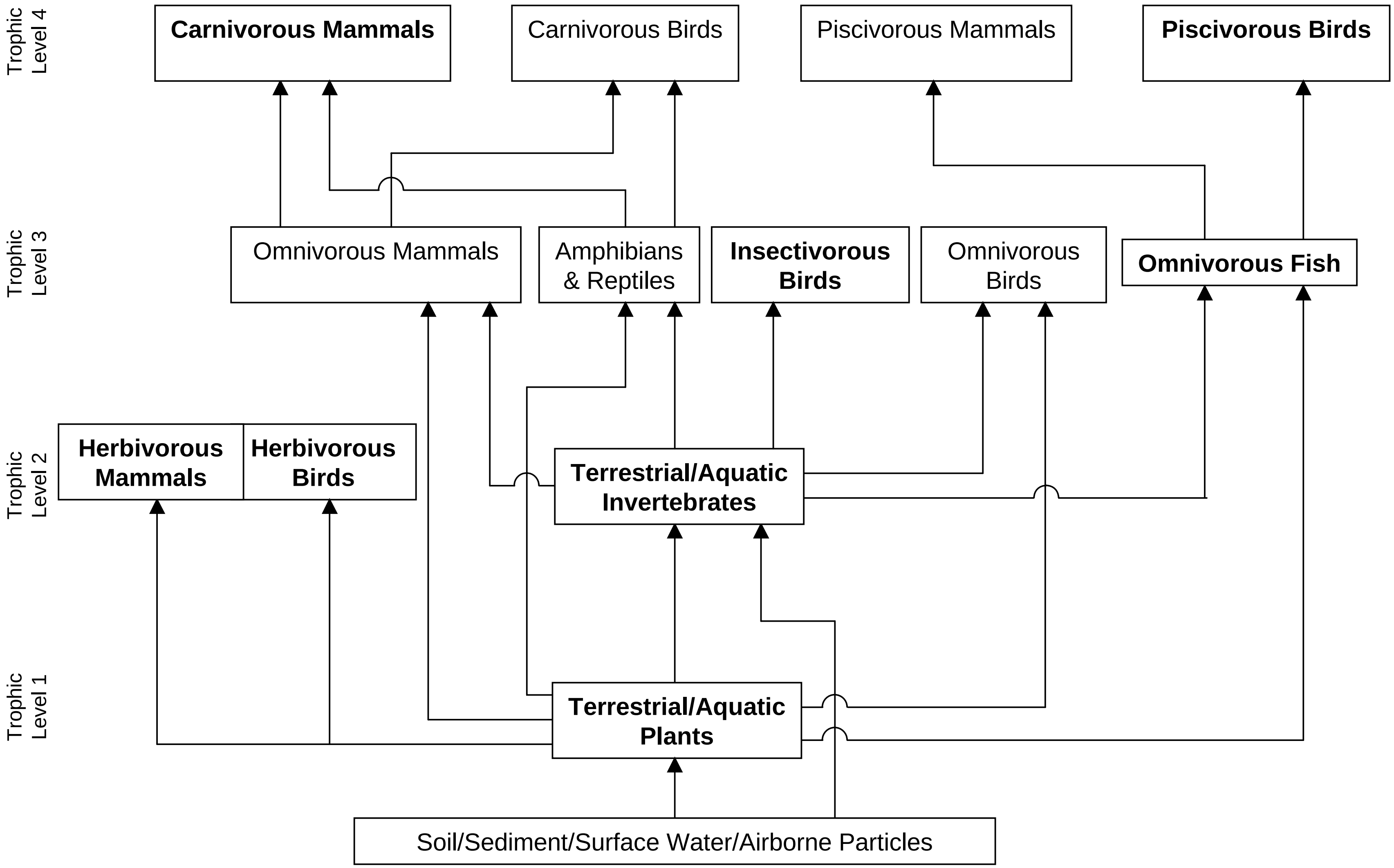


FIGURE 4-2. Simplified Food Web.



Receptors in **bold** type quantitatively assessed in the BLM herbicide ERAs.

Figure 4-3. Direct Spray - Risk Quotients for Terrestrial Animals & Semi-Aquatic Wildlife.

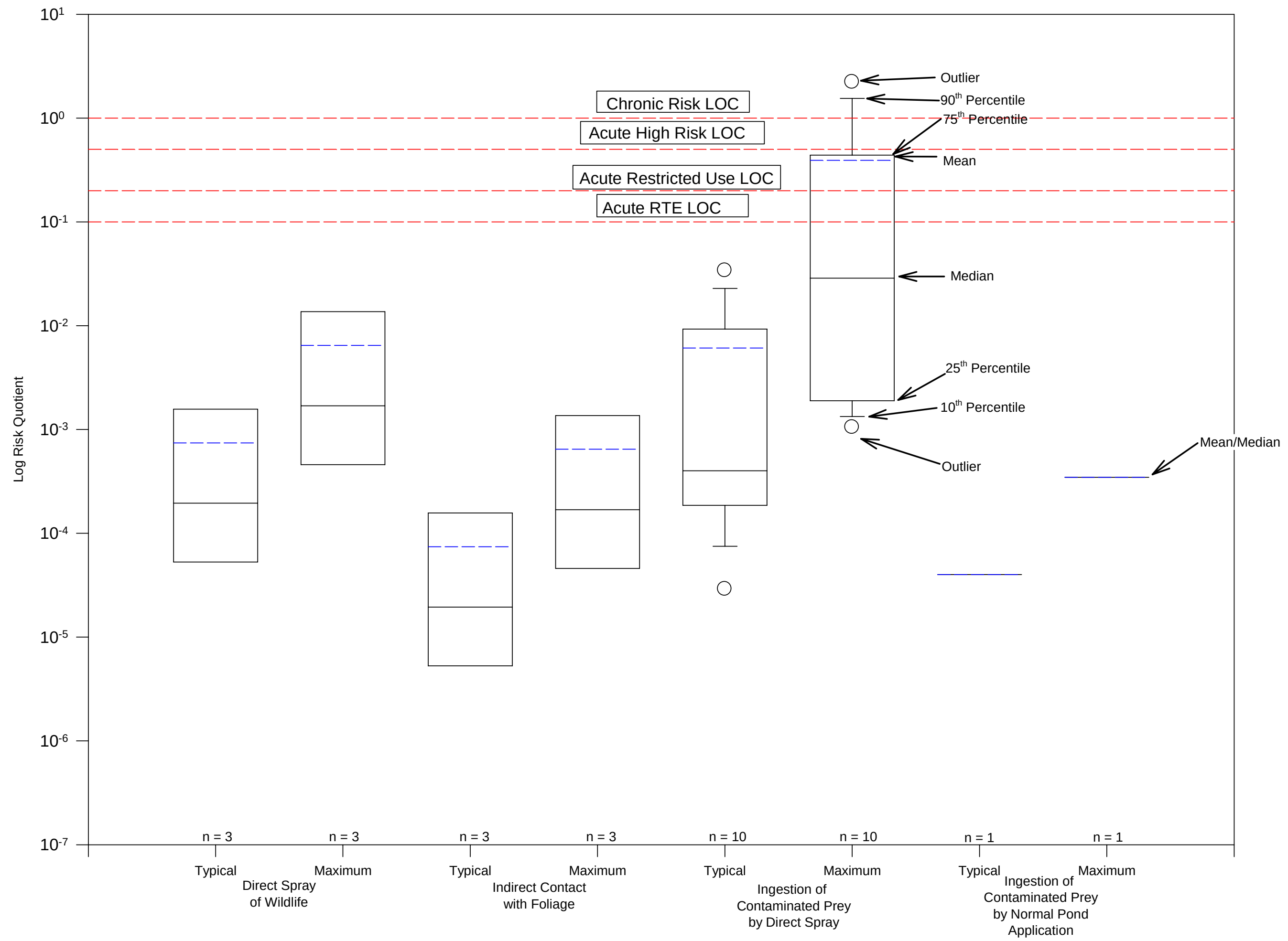


FIGURE 4-4. Accidental Direct Spray and Spills - Risk Quotients for Non-Target Aquatic Plants.

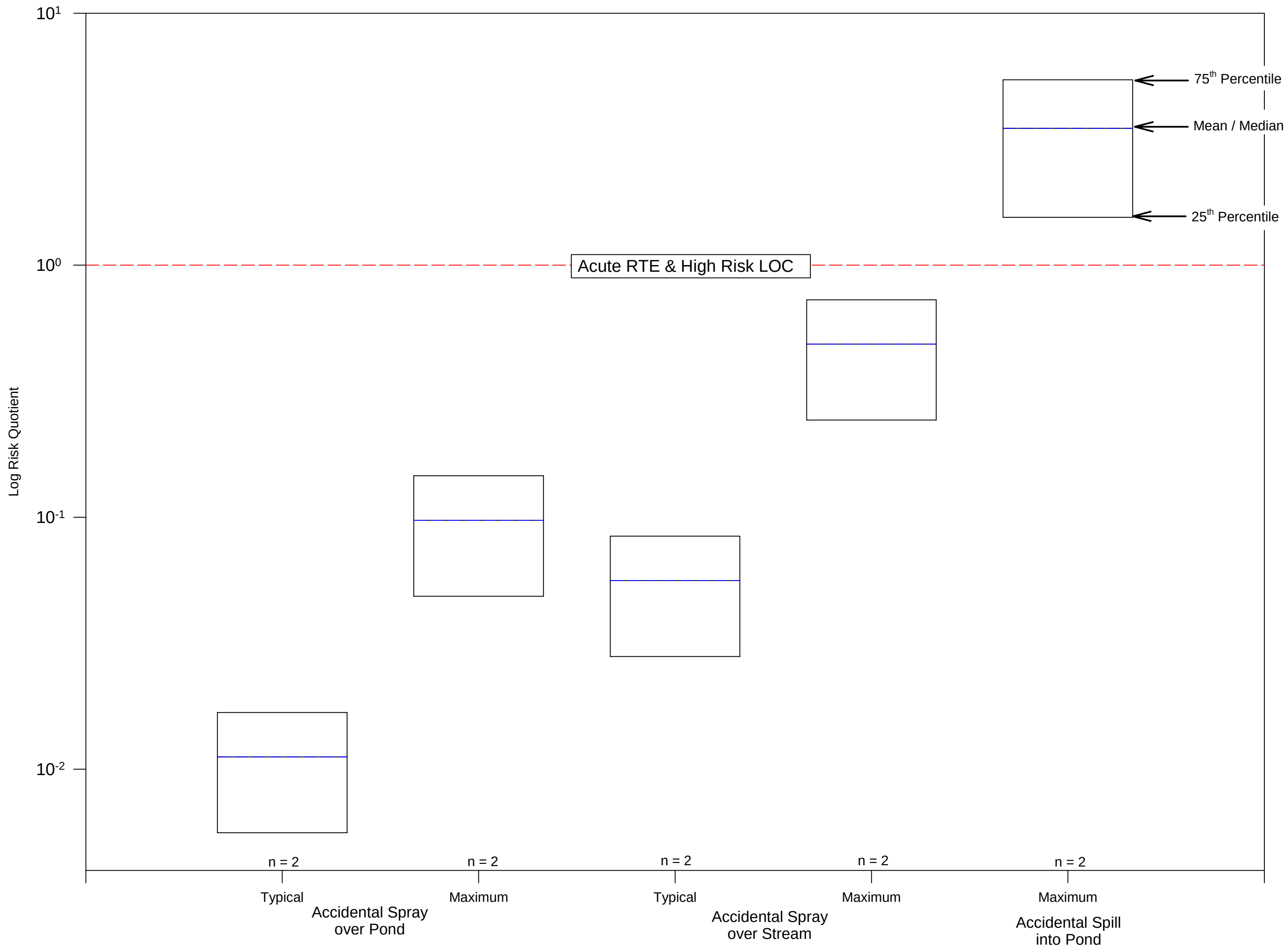


Figure 4-5. Accidental Direct Spray and Spills - Risk Quotients for Fish.

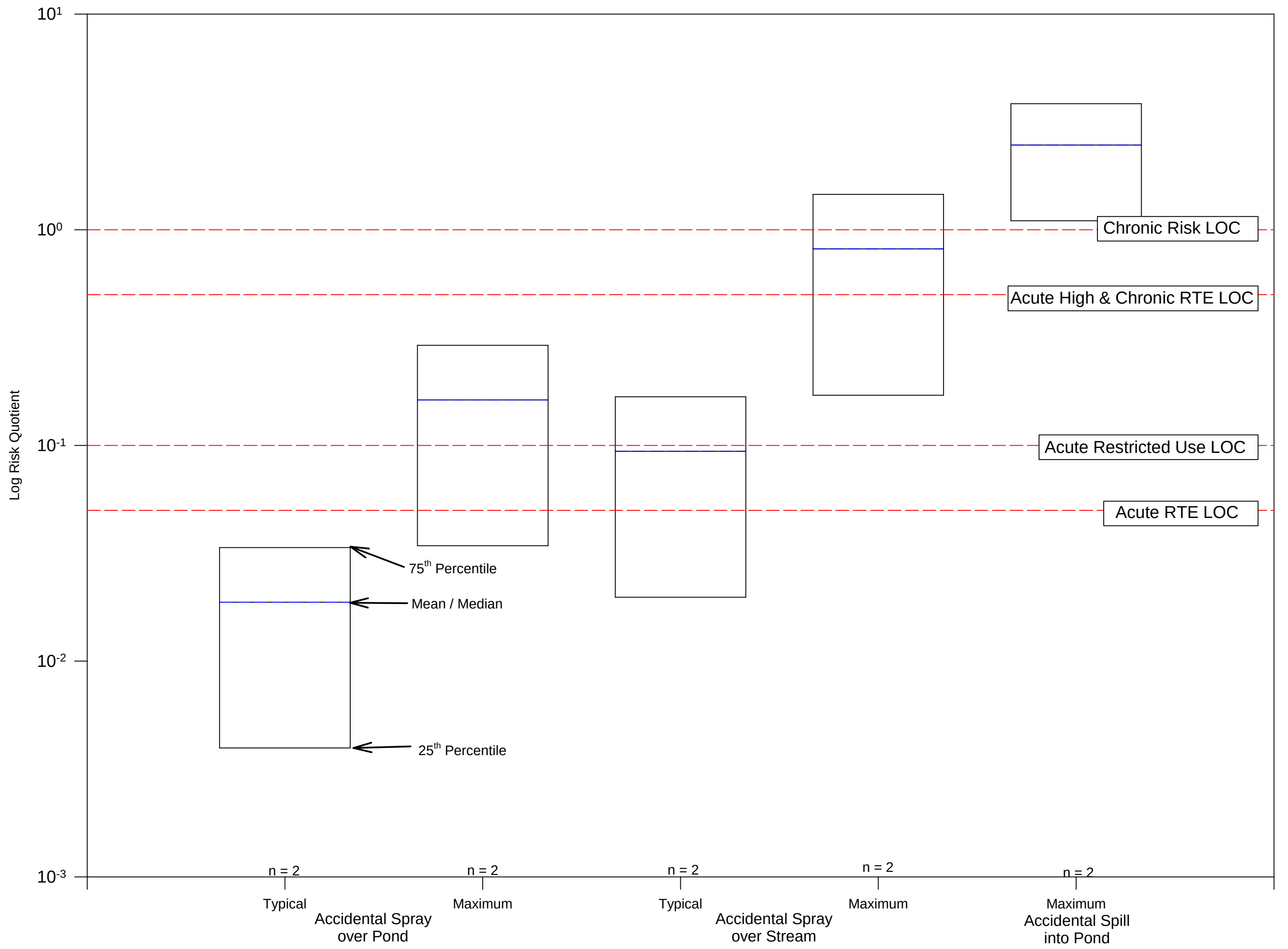
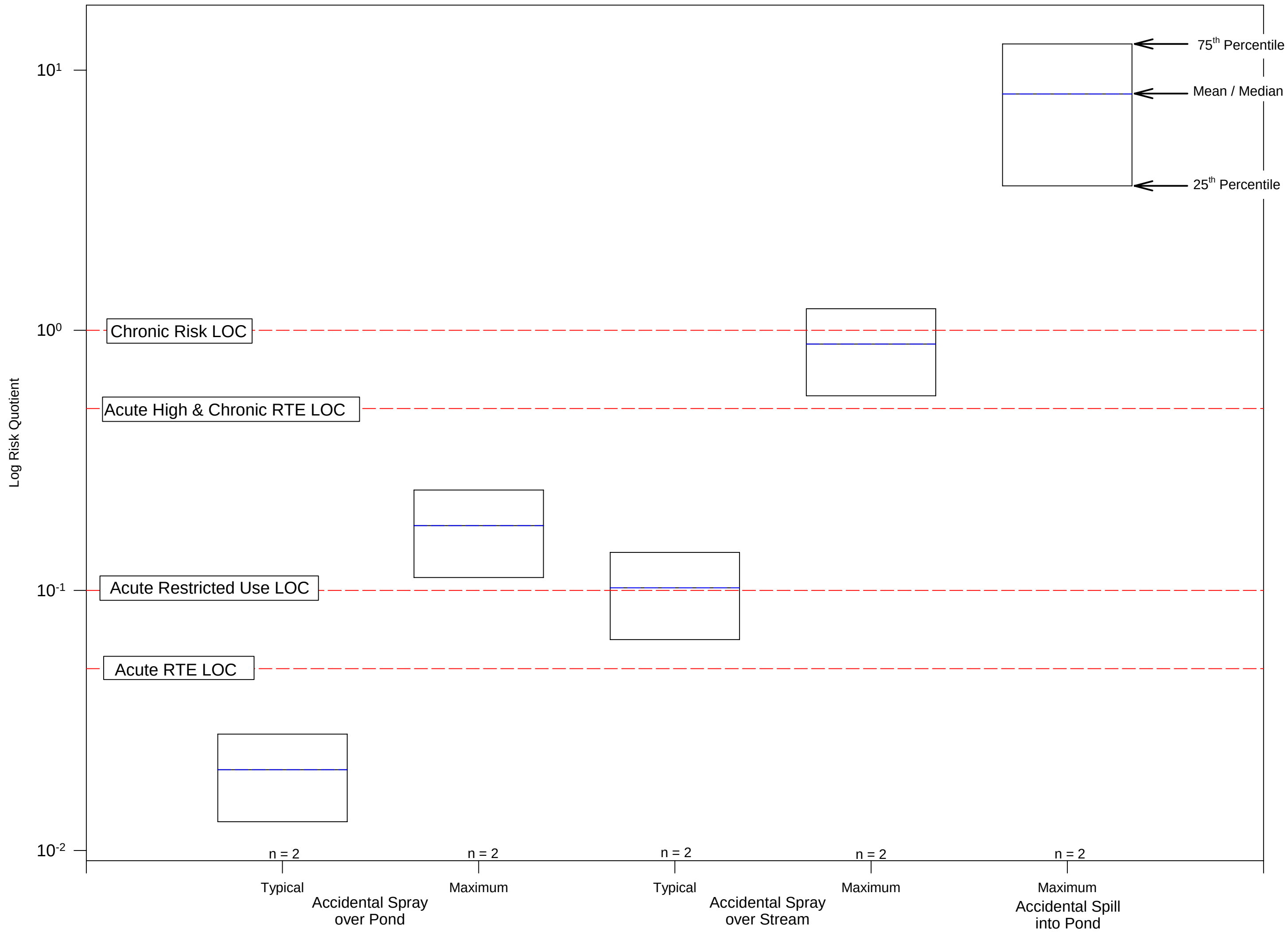


Figure 4-6. Accidental Direct Spray and Spills - Risk Quotients for Aquatic Invertebrates.



5.0 SENSITIVITY ANALYSIS

The sensitivity analysis was designed to determine which factors most greatly affect exposure concentrations. Changes in herbicide concentrations were modeled with respect to changes in pond and stream area and depth. The effects of off-site drift on terrestrial species were estimated using the AgDRIFT® model. A base case for the AgDRIFT® model was established, and from this base case various input factors were changed independently, thereby resulting in an estimate of the importance of that factor on exposure concentrations. Information regarding the AgDRIFT® model, its specific use and any inputs and assumptions made during the application of this model is provided in the Methods Document (ENSR 2004c).

5.1 Pond Volume and Stream Flow Sensitivity

The sensitivity analysis was designed to determine how pond and stream volumes affect exposure concentrations. A base case for each model was established. Input factors (e.g., area, depth) were changed independently, thereby resulting in an estimate of the importance of that factor on exposure concentrations. As described previously, surface runoff and wind erosion were not considered as transport mechanisms for the aquatic herbicides. The scenarios for the aquatic herbicides are relatively simplistic and essentially represent an instantaneous concentration in the waterbody due to direct applications. The predicted surface water concentrations are based on the application rate, and the surface area and depth of the waterbody. The surface water concentrations predicted in these scenarios are likely to be an overestimate since stream flow, degradation, and adsorption are not considered.

The base case for the pond consisted of a ¼ acre pond 1 meter deep. Table 5-1 presents the variations in the pond surface water concentrations as the area and depth of the pond are changed. This analysis indicates that changing the area of the pond does not alter the predicted surface water concentration because as more herbicide is sprayed over a larger area, there is a larger pond volume in which the herbicide is dissipated. However, changing the depth does have an impact on the pond concentration because the pond volume changes, but the amount of herbicide sprayed on the pond is unchanged. For example, an increase in the pond depth will decrease the associated herbicide concentration in the surface water.

The base case for the stream consisted of a stream 2 m wide and 0.2 m deep. The base case length was based on one side of a 100 acre square application area (636 m). Table 5-2 presents the variations in the stream surface water concentrations as the width, length, and depth of the impacted stream are changed. As observed in the pond sensitivity analysis, changes to stream area accomplished by varying the length or width do not result in changes to the surface water concentrations. Changes to the stream depth do result in associated changes to the stream concentrations. As the depth is increased, the stream concentration decreases and as the depth decreases, the stream concentration increases.

The results of this sensitivity analysis indicate that the size of the impacted water body does not have an effect on the surface water concentration (assuming that the entire waterbody is sprayed). However, depth has a dramatic impact on the associated surface water concentration (doubling the depth decreased the water concentration by ½). This indicates that shallow ponds and streams are more likely to be impacted by herbicide spray.

5.2 AgDRIFT® Sensitivity

Changes to individual input parameters of predictive models have the potential to substantially influence the results of an analysis such as that conducted in this ERA. This is particularly true for models such as AgDRIFT® which are intended to represent complex problems such as the prediction of off-target spray drift of herbicides. Predicted off-target spray drift and downwind deposition can be substantially altered by a number of variables intended to represent the herbicide application process including, but not limited to: 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. This section will present the changes that occur to the EEC with changes to important input parameters and assumptions used in the AgDRIFT® model. It is important to note that changes in the EEC directly affect the estimated RQ. Thus, this information is presented to help local land managers understand the factors that are likely to be related to higher potential ecological risk. Table 5-3 summarizes the relative change in exposure concentrations, and therefore ecological risk, based on specific model input parameters (e.g., mode of application, application rate).

Factors that are thought to have the greatest influence on downwind drift and deposition are: spray drop-size distribution, release height, and wind speed (Teske and Barry 1993; Teske et al. 1998; Teske and Thistle 1999, *as cited in SDTF 2002*). To better quantify the influence of these and other parameters, a sensitivity analysis was undertaken by the SDTF and documented in the AgDRIFT® user's manual. In this analysis AgDRIFT® Tier II model input parameters (model input parameters are discussed in Appendix B of the HHRA) were varied by 10% above and below the default assumptions (four different drop-size distributions were evaluated). The findings of this analysis indicate the following:

- The largest variation in predicted downwind drift and deposition patterns occurred as a result of changes in the shape and content of the spray drop size distribution.
- The next greatest change in predicted downwind drift and deposition patterns occurred as a result of changes in boom height (the release height of the spray mixture).
- Changes in spray boom length resulted in significant variations in drift and deposition within 200 ft downwind of the hypothetical application area.
- Changes in the assumed ambient temperature and relative humidity resulted in small variation in drift and deposition at distances > 200 ft downwind of the hypothetical application area.
- Varying the assumed number of application swaths (aircraft flight lines), application swath width, and wind speed resulted in little change in predicted downwind drift and deposition.
- Variation in nonvolatile fraction of the spray mixture showed no effect on downwind drift and deposition.

These results, except for the minor to negligible influence of varying wind speed and nonvolatile fraction, were consistent with previous observations. The 10% variation in wind speed and nonvolatile fraction was likely too small to produce substantial changes in downwind drift and deposition. It is expected that varying these by a larger percentage would eventually produce some effect. In addition, changes in wind speed resulted in changes in application swath width and swath offset, which masked the effect of wind speed alone on downwind drift and deposition.

Based on these findings, and historic field observations, the hierarchy of parameters that have the greatest influence on downwind drift and deposition patterns is as follows:

1. Spray drop size distribution
2. Application boom height
3. Wind speed
4. Spray boom length
5. Relative humidity
6. Ambient temperature
7. Nonvolatile fraction

An additional limitation of the AgDRIFT® user's manual sensitivity analysis is the focus on distances < 200 ft downwind of a hypothetical application area. From a land management perspective, distance downwind from the point of deposition may be considered to represent a hypothetical buffer zone between the application area and a potentially sensitive habitat. In this ERA, distances as great as 900 ft downwind of a hypothetical application were considered. In an effort to expand on the existing AgDRIFT® sensitivity analysis provided in the user's manual, the sensitivity of mode of application, application height or vegetation type, and application rate were evaluated. Results of this supplemental analysis are provided in Table 5-3.

The results of the expanded sensitivity analysis indicate that deposition and corresponding ecological risk drop off substantially between 25 and 900 ft downwind of hypothetical application area. Thus, from a land management perspective, the size of a hypothetical buffer zone (the downwind distance from a hypothetical application area to a potentially sensitive habitat) may be the single most controllable variable (other than the application rate, equipment and herbicide mixtures chosen) that has a substantial impact on ecological risk (Table 5-3).

The most conservative case at the typical application rate (using the smallest downwind distance measured in this ERA – 25 ft) was then evaluated using two different boom heights. Predicted concentrations were greater with high vs. low boom height (Table 5-3); ecological risk, therefore, increases with boom height. The effect of mode of application was evaluated using plane, helicopter and ground dispersal (using the typical application rate, smallest downwind distance, and non-forested cover or high boom height). Plane dispersal resulted in the highest predicted exposure concentrations, and therefore, represents the greatest risk. Ground applications resulted in the lowest predicted exposure concentrations. The effect of application rate (maximum vs. typical) was also tested, and as expected, predicted concentrations (and ecological risk) increase with increased application rates (Table 5-3). Concentrations were approximately four times greater using maximum application rates than using typical application rates.

TABLE 5-1
Relative Effects of Pond Variables on Herbicide Exposure Concentrations using Typical BLM Application Rate

Pond area (acres)	Pond depth (m)	Pond volume (L)	Mass sprayed on pond (mg)	Concentration in pond (mg/L)	Comments
0.25	1	1,011,714	17,010	0.02	Base case
100	1	404,685,642	6,803,886	0.02	Increased pond area; No change in concentration
1000	1	4,046,856,422	68,038,856	0.02	Increased pond area; No change in concentration
0.25	0.5	2,023,428,211	17,010	0.03	Decreased pond depth; Increased concentration
0.25	2	2,023,428	17,010	0.008	Increased pond depth; Decreased concentration
0.25	4	4,046,856	17,010	0.004	Increased pond depth; Decreased concentration

TABLE 5-2
Relative Effects of Stream Variables on Herbicide Exposure Concentrations using Typical BLM Application Rate

Stream width (m)	Stream depth (m)	Length of impacted stream (m) [1]	Stream volume (L)	Mass sprayed on stream (mg)	Concentration in stream (mg/L)	Comments
2	0.2	636	254,460	21,391	0.08	Base case
4	0.2	636	508,920	42,782	0.08	Increased stream width; No change in concentration
1	0.2	636	127,230	10,695	0.08	Decreased stream width; No change in concentration
2	0.4	636	508,920	21,391	0.04	Increased stream depth; Decreased concentration
2	0.1	636	127,230	21,391	0.17	Decreased stream depth; Increased concentration
2	0.2	201	80,468	6,764	0.08	Increased stream length; No change in concentration
2	0.2	2,012	804,672	67,644	0.08	Decreased stream length; No change in concentration

(1) – Length of impacted stream is based on size of application area. 10 acre application area = 201 meters impacted; 100 acre application area = 636 meters impacted; 1,000 acre application area = 2,012 meters impacted.

TABLE 5-3
Herbicide Exposure Concentrations Used During the Supplemental AgDRIFT® Sensitivity Analysis

Mode of Application	Application Height or Vegetation Type	Minimum Downwind Distance	Maximum Downwind Distance	Minimum Downwind Distance Concentration Pond (mg/L)	Maximum Downwind Distance Concentration Pond (mg/L)
Typical Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	2.94E-03	6.31E-04
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	2.50E-03	5.15E-04
Ground	Low Boom	25	900	2.79E-04	2.96E-05
	High Boom	25	900	4.49E-04	3.76E-05
Maximum Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	1.13E-02	2.80E-03
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	9.42E-03	2.01E-03
Ground	Low Boom	25	900	9.13E-04	9.67E-05
	High Boom	25	900	1.47E-03	1.23E-04

Effect of Downwind Distance

Mode of Application	Application Height or Vegetation Type	Minimum Downwind Distance	Maximum Downwind Distance	Concentration $_{900}$/ Concentration $_{25 \text{ or } 100}$	Relative Change in Concentration
Typical Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2146	-
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2060	-
Ground	Low Boom	25	900	0.1061	-
	High Boom	25	900	0.0837	-
Maximum Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2478	-
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2134	-
Ground	Low Boom	25	900	0.1059	-
	High Boom	25	900	0.0837	-

TABLE 5-3 (Cont.)
Herbicide Exposure Concentrations Used During the Supplemental AgDRIFT® Sensitivity Analysis

Effect of Application Vegetation Type or Boom Height

Mode of Application	Application Height or Vegetation Type	Vegetation Type or Boom Height[1]	Relative Change in Concentration
Typical Application Rate			
Plane	Forest/ Non-Forest	NA	NA
Helicopter	Forest/ Non-Forest	NA	NA
Ground	High/Low Boom	1.6093	+
Maximum Application Rate			
Plane	Forest/ Non-Forest	NA	NA
Helicopter	Forest/ Non-Forest	NA	NA
Ground	High/Low Boom	1.6101	+

Effect of Mode of Application

	Mode of Application[2]	Relative Difference
Typical Application Rate		
Plane vs. Helicopter	1.1760	+
Plane vs. Ground	6.5479	+
Helicopter vs. Ground	5.5679	+
Maximum Application Rate		
Plane vs. Helicopter	1.1996	+
Plane vs. Ground	7.6871	+
Helicopter vs. Ground	6.4082	+

Effect of Mode of Application Rate

	Application Rate[3]	Relative Difference
Maximum vs. Typical	3.2739	+

(1) using minimum buffer width concentrations.
(2) using minimum buffer width and non-forest or high boom concentrations.
(3) using ground dispersal, minimum buffer width, and high boom concentrations.
"+" = Increase in concentration = increase in RQ = increase in ecological risk.
"-" = Decrease in concentration = decrease in RQ = decrease in ecological risk.

6.0 RARE, THREATENED, AND ENDANGERED SPECIES

Rare, threatened, and endangered (RTE) species have the potential to be impacted by herbicides applied for vegetation control. RTE 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 our ability to evaluate site- and species-specific effects:

- 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, including this one, to be highly conservative. This includes assumptions such as 100% exposure to an herbicide by simulating scenarios where the organism lives year-round 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 fluridone screening level ERA incorporates additional conservatism in the assumptions used in the herbicide concentration models such as AgDRFIT® (Appendix A; ENSR 2004c). Even with highly conservative assumptions in the ERA, however, concern may still exist over the potential risk to specific RTE species.

To help address this potential concern, the following section will discuss the ERA assumptions as they relate to the protection of RTE species. The goals of this discussion are as follows:

- Present the methods the ERA employs to account for risks to RTE species and the reasons for their selection.
- Define the factors that might motivate a site- and/or species-specific evaluation[3] of potential herbicide impacts to RTE species and provide perspective useful for such an evaluation.
- Present information that is relevant to assessing the uncertainty in the conclusions reached by the ERA with respect to RTE species.

The following sections describe information used in the ERA to provide protection to RTE species, including mammals, birds, plants, reptiles, amphibians and fish (e.g., salmonids) potentially occurring on BLM-managed lands. It includes a discussion of the quantitative and qualitative factors used to provide additional protection to RTE species and a discussion of potential secondary effects of herbicide use on RTE species.

Section 6.1 provides a review of the selection of LOCs and TRVs with respect to providing additional protection to RTE species. Section 6.2 provides a discussion of species-specific traits and how they relate to the RTE protection strategy in this ERA. Section 6.2 also includes discussion of the selection of surrogate species (6.2.1), the RTE taxa of

[3] Such an evaluation might include site-specific estimation of exposure point concentrations using one or more models, more focused consideration of potential risk to individual RTE species; and/or more detailed assessment of indirect effects to RTE species, such as those resulting from impacts to habitat.

concern, and the surrogates used to represent them (6.2.2), and the biological factors that affect the exposure to and response of organisms to herbicides (6.2.3). This includes a discussion of how the ERA was defined to assure that consideration of these factors resulted in a conservative assessment. Mechanisms for extrapolating toxicity data from one taxon to another are briefly reviewed in Section 6.3. The potential for impacts, both direct and secondary, to salmonids is discussed in Section 6.4. Section 6.5 provides a summary of the section.

6.1 Use of LOCs and TRVs to Provide Protection

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 fluridone ERA by comparing calculated RQs to receptor-specific LOCs. As described in the methodology document for this ERA (ENSR 2004c), RQs are calculated as the potential dose or EEC divided by the TRV selected for that pathway. 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 4-1) were developed by the USEPA for the assessment of pesticides (LOC information obtained from Michael Davy, USEPA OPP on 13 June 2002). In essence, the LOCs act as uncertainty factors often applied to TRVs. For example, using an LOC of 0.1 provides the same result as dividing the TRV by 10. The LOC for avian and mammalian RTE species is 1.0 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 RTE species to account for the factors listed in the introduction to 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 (Section 3.1); 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 consistent with USEPA policy, no additional levels of protection were required for the LOC (all plant LOCs are 1).

6.2 Use of Species Traits to Provide Protection to RTE Species

Over 500 RTE species currently listed under the Federal Endangered Species Act (ESA) have the potential to occur in the 17 states covered under this Programmatic ERA. These species include 287 plants, 80 fish, 30 birds, 47 mammals, 15 reptiles, 13 amphibians, 34 insects, 10 arachnids (spiders), and 22 aquatic invertebrates (12 mollusks and 10 crustaceans).[4] Some marine mammals are included in the list of RTE species; but due to the limited possibility these species would be exposed to herbicides applied to BLM-managed lands, no surrogates specific to marine species are included in this ERA. However, the terrestrial mammalian surrogate species identified for use in the ERA include species that can be considered representative of these marine species as well. The complete list is presented in Appendix D.

Of the over 500 species potentially occurring in the 17 states, just over 300 species may occur on lands managed by the BLM. These species include 7 amphibians, 19 birds, 6 crustaceans, 65 fish, 30 mammals, 10 insects, 13 mollusks, 5 reptiles, and 151 plants. Protection of these species is an integral goal of the BLM, and they are the focus of the RTE evaluation for the ERA and EIS. These species are different from one another in regards to home range, foraging strategy, trophic level, metabolic rate, and other species-specific traits. Several methods were used in the ERA to take these differences into account during the quantification of potential risk. Despite this precaution, these traits are reviewed in order to provide a basis for potential site- and species-specific risk assessment. Review of these factors

[4] The number of RTE species may have changed slightly since the writing of this document.

provides a supplement to other sections of the ERA that discuss the uncertainty in the conclusions specific to RTE species.

6.2.1 Identification of Surrogate Species

Use of surrogate species in a screening ERA is necessary to address the broad range of species likely to be encountered on BLM-managed 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, are considered. As described in Section 3.0 of the Methods document (ENSR 2004c), surrogate species were selected to represent a broad range of taxa in several trophic guilds that could potentially be impacted by herbicides on BLM-managed lands. Generally, the surrogate species that were used in the ERA are species commonly used as representative species in ERA. Many of these species are common laboratory species, or are described in USEPA (1993a, b) Exposure Factors Handbook for Wildlife. Other species were included in the California Wildlife Biology, Exposure Factor, and Toxicity Database (CA OEHHA 2003),[5] or are those recommended by USEPA OPP for tests to support pesticide registration Surrogate species were used to derive TRVs, and in exposure scenarios that involve organism size, weight, or diet, surrogate species were exposed to the herbicide in the models to represent potential impact to other species that may be present on BLM lands.

Toxicity data from surrogate species were used in the development of TRVs because few, if any, data are available that demonstrate the toxicity of chemicals to RTE species. Most reliable toxicity tests are performed under controlled conditions in a laboratory, using standardized test species and protocols; RTE species are not used in laboratory toxicity testing. In addition, field-generated data, which are very limited in number but may include anecdotal information about RTE species, are not as reliable as laboratory data because uncontrolled factors may complicate the results of the tests (e.g., secondary stressors such as unmeasured toxicants, imperfect information on rate of exposure).

As described below, inter-species extrapolation of toxicity data often produces unknown bias in risk calculations. This ERA approached the evaluation of higher trophic level species by life history (e.g., large animals vs. small animals, herbivore vs. carnivores). Then surrogate species were used to evaluate all species of similar life history potentially found on BLM-managed lands, including RTE species. This 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.

The following subsections describe the selection of surrogate species used in two separate contexts in the ERA.

6.2.1.1 Species Selected in Development of TRVs

As presented in Appendix A of the ERA, limited numbers of species are used for toxicity testing of chemicals, including herbicides. Species are typically selected because they tolerate laboratory conditions well. The species used in laboratory tests have relatively well-known response thresholds to a variety of chemicals. Growth rates, ingestion rates, and other species-specific parameters are known; therefore, test duration and endpoints of concern (e.g., mortality, germination) have been established in protocols for many of these laboratory species. Data generated during a toxicity test, therefore, can be compared to data from other tests and relative species sensitivity can be compared. Of course, in the case of RTE species, it would be unacceptable to subject individuals to toxicity tests.

The TRVs used in the ERA were selected after reviewing available ecotoxicological literature for fluridone. Test quality was evaluated, and tests with multiple substances were not considered for the TRV. For most receptor groups, the lowest value available for an appropriate endpoint (e.g., mortality, germination) was selected as the TRV. Using

[5] On-line http://www.oehha.org/cal_ecotox/default.htm

the most sensitive species provides a conservative level of protection for all species. The surrogate species used in the fluridone TRVs are presented in Table 6-1.

6.2.1.2 Species Selected as Surrogates in the ERA

Plants, fish, insects, and other aquatic invertebrates were evaluated on a generic level. That is, the surrogate species evaluated to create the TRVs were selected to represent all potentially exposed species. For vertebrate terrestrial animals, in addition to these surrogate species, specific species were selected to represent the populations of similar species. The species used in the ERA are presented in Table 6-2.

The surrogate terrestrial vertebrate species selected for the ERA include species from several trophic levels that represent a variety of foraging strategies. Whenever possible, the species selected are found throughout the range of land included in the EIS; 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 1993 a, b; CA OEHHA 2003). Because species-specific data, including BW 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. As necessary, site-specific data can be used to estimate potential risk to species known to occur locally.

6.2.2 Surrogates Specific to Taxa of Concern

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) were represented by the surrogate species presented in Table 6-1. The evaluation of terrestrial vertebrates used surrogate species to develop TRVs and to estimate potential risk using simple food chain models. Tables 6-3 and 6-4 present the listed birds and mammals found on BLM-managed lands and their appropriate surrogate 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 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. For each trophic level of RTE reptile or adult amphibian, a comparable mammal or bird was selected to represent the potential risks. Table 6-5 presents the 7 listed reptiles found on BLM-managed lands and the surrogate species chosen to represent them in the ERA. Table 6-6 presents the listed amphibians found on BLM-managed lands and their surrogate species.

The sensitivity of reptiles and amphibians relative to other species is generally unknown. Some information about reptilian exposures to pesticides, including herbicides, is available. The following provides a brief summary of the data (*as cited in Sparling et al. 2000*), including data for pesticides not evaluated in this ERA:

- Mountain garter snakes (*Thamnophis elegans elegans*) were exposed to the herbicide thiobencarb in the field and in the laboratory. No effects were noted in the snakes fed contaminated prey or those caged and exposed directly to treated areas.

- No adverse effects to turtles were noted in a pond treated twice with the herbicide Kuron (2,4,5-T).

- Tortoises in Greece were exposed in the field to atrazine, paraquat, Kuron, and 2,4-D. No effects were noted on the tortoises exposed to atrazine or paraquat. In areas treated with Kuron and 2,4-D, no tortoises were noted following the treatment. The authors of the study concluded it was a combination of direct toxicity (tortoises were noted with swollen eyes and nasal discharge) and loss of habitat (much of the vegetation killed during the treatment had provided important ground cover for the tortoises).

- Reptilian LD_{50} values from six organochlorine pesticides were compared to avian LD_{50} values. Of the six pesticides, five lizard LD_{50}s were higher, indicating lower sensitivity. Overlapping data were available for turtle exposure to one organochlorine pesticide; the turtle was less sensitive than the birds or lizards.

- In general, reptiles were found to be less sensitive than birds to cholinesterase inhibitors.

Unfortunately, these observations do not provide any sort of rigorous review of dose and response. On the other hand, there is little evidence that reptiles are more sensitive to pesticides than other, more commonly tested organisms.

As with reptiles, some toxicity data are available describing the effects of herbicides on amphibians. The following provides a brief summary of the data (as cited in Sparling et al. 2000):

- Leopard frog *(Rana pipiens)* tadpoles exposed to up to 0.075 mg/L atrazine showed no adverse effects.

- In a field study, it was noted that frog eggs in a pond where atrazine was sprayed nearby suffered 100% mortality.

- Common frog (*Rana temporaria)* tadpoles showed behavioral and growth effects when exposed to 0.2 to 20 mg/L cyanatryn.

- Caged common frog and common toad (*Bufo bufo)* tadpoles showed no adverse effects when exposed to 1.0 mg/L diquat or 1.0 mg/L dichlobenil.

- All leopard frog eggs exposed to 2.0 to 10 mg/L diquat or 0.5 to 2.0 mg/L paraquat hatched normally, but showed adverse developmental effects. It was noted that commercial formulations of paraquat were more acutely toxic than technical grade paraquat. Tadpoles, however, showed significant mortality when fed paraquat-treated parrot feather watermilfoil (*Myriophyllum).*

- 4-chloro-2-methylphenoaxyacetic acid (MCPA) is relatively non-toxic to the African clawed frog (*Xenopus laevis)* with an LC_{50} of 3,602 mg/L and slight growth retardation at 2,000 mg/L.

- Approximately 86% of juvenile toads died when exposed to monosodium methanearsonate (ANSAR 259® HC) at 12.5% of the recommended application rate.

- Embryo hatch success, tadpole mortality, growth, paralysis, and avoidance behavior were studied in three species of ranid frogs (*Rana* sp.) exposed to hexazinone and triclopyr. No effects were noted in hexazinone exposure up to 100 mg/L. Two species showed 100% mortality at 2.4 mg/L triclopyr; no significant mortality was observed in the third species.

No conclusions can be drawn regarding the sensitivity of amphibians to exposure to fluridone relative to the surrogate species selected for the ERA. Amphibians are particularly vulnerable to changes in their environment (chemical and physical) because they have skin with high permeability, making them at risk to dermal contact, and have complex life cycles, making them vulnerable to developmental defects during the many stages of metamorphosis. Given the very low risks to animals in the modeled exposures, it is unlikely the concentrations of fluridone predicted to occur as a result of regular herbicide usage would cause adverse effects to amphibians. Nonetheless, it should be noted that amphibians can be sensitive to pesticides, and site- and species-specific risk assessment should be carefully considered in the event that amphibian RTE species are present near a site of application.

Although the uncertainties associated with the potential risk to RTE mammals, birds, reptiles, and amphibians are valid, the vertebrate RQs generated in the ERA for fluridone are generally very low (Section 4.3). None of the RQs exceed respective LOCs. Of the four general scenarios in which vertebrate receptors were evaluated, the highest RQ was 0.38 (chronic exposure of small mammalian herbivore ingesting prey contaminated by direct spray at maximum application rate). This RQ is lower than the chronic RTE LOC of 1. Most vertebrate RQs, including fish exposure to normal applications, were lower than respective LOCs by several orders of magnitude.

6.2.3 Biological Factors Affecting Impact from Herbicide Exposure

The potential for ecological receptors to be exposed to, and affected by, herbicide is dependent upon many factors. Many of these factors are independent of the biology or life history of the receptor (e.g., timing of herbicide use, distance to receptor). These factors were explored in the ERA by simulating scenarios that vary these factors (ENSR 2004c), and these scenarios are discussed in Section 5.0 of this document. However, there are differences in life history among and between receptors that also influence the potential for exposure. Therefore, individual species have a different potential for exposure as well as response. In order to provide perspective on the assumptions made here, as well as the potential need to evaluate alternatives, receptor traits that may influence species-specific exposure and response were examined. These traits are presented and discussed in Table 6-7.

In addition to providing a review of the approach used in the ERA, the factors listed in Table 6-7 can be evaluated in order to assess whether a site- and species-specific ERA should be considered to address potential risks to a given RTE. They also provide perspective on the uncertainty associated with applying the conclusions of the ERA to a broad range of RTE species.

6.3 Review of Extrapolation Methods Used to Calculate Potential Exposure and Risk

Ecological risk assessment relies on extrapolation of observations from one system (e.g., species and toxicity endpoint) to another (see Table 6-7). While every effort has been made to anticipate bias in these extrapolations and to use them to provide an overestimate of risk, it is worth evaluating alternative approaches.

Toxicity Extrapolations in Terrestrial Systems (Fairbrother and Kaputska 1996) is an opinion paper that describes the difficulties associated with trying to quantitatively evaluate a particular species when toxicity data for that species, and for the endpoint of concern, are not available. The authors provide an overview of uncertainty factors and methods of data extrapolation used in terrestrial organism TRV development, and suggest an alternative approach to establishing inter-species TRVs. The following subsections summarize their findings for relevant methods of extrapolation.

6.3.1 Uncertainty Factors

Uncertainty factors are used often in both human health and ERA. The uncertainty factor most commonly used in ERA is 10. This value has little empirical basis, but was developed and adopted by the risk assessment community because it seemed conservative and was "simple to use."[6] Six situations in which uncertainty factors may be applied in ecotoxicology were identified: (1) accounting for intraspecific heterogeneity, (2) supporting interspecific extrapolation, (3) converting acute to chronic endpoints and vice versa, (4) estimating LOAEL from NOAEL, (5) supplementing professional judgment, and (6) extrapolating laboratory data to field conditions. No extrapolation of toxicity data among Classes (i.e., between birds, mammals, and reptiles) was discussed. The methods to extrapolate available laboratory toxicity data to suit the requirements of the TRVs in this ERA are discussed in Section 3. For this reason, extrapolation used to develop TRVs is not discussed in this section.

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[6] Section 2, Fairbrother and Kaputska 1996. Page 7.

Empirical data for each of the situations discussed in the Fairbrother and Kaputska paper (as applicable) are presented in Tables 6-8 through 6-12. In each of these tables, Fairbrother and Kaputska (1996) have presented the percentage of the available data that is included within a stated factor. For example, 90% of the observed LD_{50} for bird species lie within a factor of ten (i.e., the highest LD_{50} within the central 90% of the population is 10-fold higher than the lowest value). This can be compared to the approach used in this ERA. For example, for aquatic invertebrates, a LOC was defined of 0.05. This is analogous to application of an uncertainty factor 20 to the relevant TRV. In this case, the selected TRV is not the highest or the mid-point of the available values but a value at the lower end of the available range. Thus, dividing the TRV by a factor of 20 is very likely to place it well below any observed TRV. With this perspective, the ranges (or uncertainty factors) provided by Fairbrother and Kaputska (1996) generally appear to support the approach used in the ERA (i.e., select low TRVs and consider comparison to an LOC < 1.0).

6.3.2 Allometric Scaling

Allometric scaling provides a formula based on BW that allows translation of doses from one animal species to another. In this ERA, allometric scaling was used to extrapolate the terrestrial vertebrate TRVs from the laboratory species to the surrogate species used to estimate potential risk. The Environmental Sciences Division of the Oak Ridge National Laboratory (ORNL) (Opresko et al. 1994 and Sample et al. 1996) has used allometric scaling for many years to establish benchmarks for vertebrate wildlife. The USEPA has also used allometric scaling in development of wildlife water quality criteria in the Great Lakes Water Quality Initiative (USEPA 1995) and in the development of ecological soil screening levels (USEPA 2000).

The theory behind allometric scaling is that metabolic rate is proportional to body size.[7] However, assumptions are made that toxicological processes are dependent on metabolic rate, and that toxins are equally bioavailable among species. Similar to other types of extrapolation, allometric scaling is sensitive to the species used in the toxicity test selected to develop the TRV. Given the limited amount of data, using the lowest value available for the most sensitive species is the best approach[4], although the potential remains for site-specific receptors to be more sensitive to the toxin. Further uncertainty is introduced to allometric scaling when the species-specific parameters (e.g., BW, ingestion rate) are selected. Interspecies variation of these parameters can be considerable, especially among geographic regions. Allometric scaling is not applicable between classes of organisms (i.e., bird to mammal). However, given these uncertainties, allometric scaling remains the most reliable easy-to-use means to establish TRVs for a variety terrestrial vertebrate species (Fairbrother and Kaputska 1996).

6.3.3 Recommendations

Fairbrother and Kaputska (1996) provided a critical evaluation of the existing, proposed, and potential means for intra-species toxicity value extrapolation. The paper they published describes the shortcomings of many methods of intra-specific extrapolation of toxicity data for terrestrial organisms. Using uncertainty factors or allometric scaling for extrapolation can often over- or underpredict the toxic effect to the receptor organism. Although using physiologically-based models may be a more scientifically correct way to predict toxicity, the logistics involved with applying them to an ERA on a large-scale make them impractical. In this ERA, extrapolation was performed using techniques most often employed by the scientific risk assessment community. These techniques included the use of uncertainty factors (i.e., potential use of LOC < 1.0) and allometric scaling.

6.4 Indirect Effects on Salmonids

In addition to the potential direct toxicity associated with herbicide exposure, organisms may be harmed from indirect effects, such as habitat degradation or loss of prey. Under Section 9 of the ESA of 1973, it is illegal to take an

[7] In the 1996 update to the ORNL terrestrial wildlife screening values document (Sample et al. 1996), studies by Mineau et al. (1996) using allometric scaling indicated that, for 37 pesticides studied, avian LD_{50}s varied from 1 to 1.55, with a mean of 1.148. The LD_{50} for birds is now recommended to be 1 across all species.

endangered species of fish or wildlife. "Take" is defined as "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (16 USC 1532(19)). The National Marine Fisheries Service (NMFS, NOAA 1999) published a final rule clarifying the definition of "harm" as it relates to take of endangered species in the ESA. NOAA Fisheries defines "harm" as any act that injures or kills fish and wildlife. Acts may include "significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding or sheltering." To comply with the ESA, potential secondary effects to salmonids were evaluated to ensure that use of fluridone on BLM-managed lands would not cause harm to these endangered fish.

Indirect effects can generally be categorized into effects caused by biological or physical disturbance. Biological disturbance includes impacts to the food chain; physical disturbance includes impacts to habitat.[8] (Freeman and Boutin 1994).NOAA Fisheries (2002) has internal draft guidance for their Section 7 pesticide evaluations. The internal draft guidance describes the steps that should be taken in an ERA to ensure salmonids are addressed appropriately. The following subsections describe how, consistent with internal draft guidance from NOAA Fisheries, the fluridone ERA dealt with the indirect effects assessment.

6.4.1 Biological Disturbance

Potential direct effects to salmonids were evaluated in the ERA. Sensitive endpoints were selected for the RTE species RQ calculations, and worst-case scenarios were assumed. No fluridone RQs for fish in normal (i.e., not accidental) scenarios exceeded the respective RTE LOC (Section 4.3). The maximum application rate RQs for fish exposed to a spill in a pond or in a stream from accidental spray slightly exceed their respective LOCs. Indirect effects caused by disturbance to the surrounding biological system were evaluated by looking at potential damage to the food chain.

The majority of the salmonid diet consists of aquatic invertebrates. Sustaining the aquatic invertebrate population is vital to minimizing biological damage to salmonids from herbicide use. Consistent with ERA guidance (USEPA 1997, 1998), protection of non-RTE species, such as the aquatic invertebrates serving as prey to salmonids, is at the population or community level, not the individual level. Sustainability of the numbers (population) or types (community) of aquatic invertebrates is the assessment endpoint. Therefore, unless acute risks are present, it is unlikely the herbicide will cause harm to the prey base of salmonids from direct damage to the aquatic invertebrates. As discussed in Section 4.3, with the exception of accidental spills or sprays, no aquatic invertebrate chronic scenario RQs exceeded respective LOCs. The aquatic invertebrate RQ from acute exposure to maximum application rate usage in a pond slightly exceeded the LOC. However, direct or indirect effects on streams, not ponds, are of primary concern to the protection of salmonids. Overall, the results of the ERA suggest that direct impacts to the forage of salmonids is unlikely.

As primary producers and the food base of aquatic invertebrates, disturbance to the aquatic vegetation may affect the aquatic invertebrate population, thereby affecting salmonids. With the exception of the accidental spill scenario, no risks to aquatic plants are estimated in the ERA. This suggests that the potential for impacts to aquatic vegetation and potential indirect effects on salmonids from the use of the herbicide are likely to be restricted to only a few extreme scenarios such as spills.

The actual food items of many aquatic invertebrates, however, are not leafy aquatic vegetation, but detritus or benthic algae. Should aquatic vegetation be affected by an accidental herbicide exposure, the detritus in the stream may increase. Disturbance of benthic algae communities as a result of herbicide application would cause an indirect effect (i.e., reduction in biomass at the base of the food chain) on all organisms living in the waterbody, including salmonids

[8] Physical damage to habitat may also be covered under an evaluation of critical habitat. Since all reaches of streams and rivers on BLM land may not be listed as critical habitat, a generalized approach to potential damage to any habitat was conducted. This should satisfy a general evaluation of critical habitats. Any potential for risk due to physical damage to habitat should be addressed specifically for areas deemed critical habitat.

(benthic algae are often the principal primary producers in streams). However, data for fluridone toxicity to benthic algae were not found.

Based on an evaluation of the RQs calculated for this ERA, it is unlikely RTE fish, including salmonids, would be at risk from the indirect effects this herbicide may have on the aquatic food chain. Exceptions to this include potential acute effects to aquatic life from accidental spills, an extreme and unlikely scenario considered in this ERA to add conservatism to the risk estimates. Appropriate and careful use of fluridone should preclude such an incident.

6.4.2 Physical Disturbance

The potential for indirect effects to salmonids due to physical disturbance is less easy to define that the potential for direct biological effects. Salmonids have distinct habitat requirements; any alteration to the coldwater streams in which they spawn and live until returning to the ocean as adults can be detrimental to the salmonid population. Out of the potential effects of herbicide application, it is likely the killing of instream and riparian vegetation would cause the most important physical disturbances. The potential adverse effects could include, but would not necessarily be limited to: loss of primary producers (Section 6.4.1); loss of overhead cover, which may serve as refuge from predators or shade to provide cooling to the waterbodies; and increased sedimentation due to loss of riparian vegetation.

Adverse effects caused by herbicides can be cumulative, both in terms of toxicity stress from break-down products and other chemical stressors that may be present, and in terms of the use of herbicides on lands already stressed at a larger scale. Cumulative watershed effects (CWEs) often arise in conjunction with other land use practices, such as prescribed burning.[9]. In forested areas, herbicides are generally used in areas that have been previously altered, such as cut or burned, during vegetative succession when invasive species may dominate. The de-vegetation of these previously stressed areas can delay the stabilization of the substrate, increasing the potential for erosion and resulting sedimentation in adjacent waterbodies.

No data to support the derivation of TRVs for terrestrial plants were found in the literature search. Therefore, the potential effects of fluridone accidental spray or drift onto terrestrial vegetation, including riparian cover in salmonid habitats, is not quantifiable. Having said this, land managers should consider the proximity of salmonid habitat to potential application areas. It may be productive to develop a more site- and/or species-specific ERA in order to ensure that the proposed herbicide application will not result in secondary impacts to salmonids especially associated with loss of riparian cover.

6.5 Conclusions

The fluridone ERA evaluated the potential risks to many species using many exposure scenarios. Some exposure scenarios are likely to occur, whereas others are unlikely to occur but were included to provide a level of conservatism to the ERA. Individual RTE species were not directly evaluated. Instead, surrogate species toxicity data were used to indirectly evaluate RTE species exposure. Higher trophic level receptors were also evaluated based on their life history strategies; RTE species were represented by one of several avian or mammalian species commonly used in ERA. To provide a layer of conservatism to the evaluation, lower LOCs and TRVs were used to assess the potential impacts to RTE species.

Uncertainty factors and allometric scaling were used to adjust the toxicity data on a species-specific basis when they were likely to improve applicability and/or conservatism. As discussed in Section 3.1, TRVs were developed using the best available data; uncertainty factors were applied to toxicity data consistent with recommendation of Chapman et al. (1998).

[9] The following website provides a more detailed discussion of CWEs http://www.humbolt1.com/~heyenga/Herb.Drft.8_12_99.html.

Potential secondary effects of fluridone use should be of primary concern for the protection of RTE species. Habitat disturbance and disruptions in the food chain are often the cause of population declines of species. For RTE species, habitat or food chain disruptions should be avoided to the extent practical. Some relationships among species are mutualistic, commensalistic, or otherwise symbiotic. For example, many species rely on a particular food source or habitat. Without that food or habitat species, the dependent species may be unduly stressed or extirpated. For RTE species, these obligatory habitats are often listed by USFWS as critical habitats. Critical habitats are afforded certain protection under the ESA. All listed critical habitat, as well as habitats that would likely support RTE species, should be avoided, as disturbance to the habitat may have an indirect adverse effect on RTE species.

Herbicides may reduce riparian zones or harm primary producers in the waterbodies. The results of the ERA indicate that non-target aquatic plants may be at risk from fluridone when accidents occur, such as spills. However, the effects of aquatic herbicides in water are expected to be relatively transient and stream flow is likely to reduce herbicide concentrations over time. Only very persistent pesticides would be expected to have effects beyond the year of their application. An OPP report on the impacts of a terrestrial herbicide on salmonids indicated that if a listed salmonid was not present during the year of application, there would likely be no concern (Turner 2003). Therefore, it is expected that potential adverse impacts to food and aquatic cover would not occur beyond the season of application.

Based on the results of the ERA, it is unlikely RTE species would be harmed by appropriate and responsible use of the herbicide fluridone on BLM-managed lands.

TABLE 6-1
Surrogate Species Used to Derive Fluridone TRVs

Species in Fluridone Laboratory/Toxicity Studies		Surrogate for
Honeybee	*Apis mellifera*	Pollinating insects
Mouse	*Cavia sp.*	Mammals
Rat	*Rattus norvegicus* spp.	Mammals
Dog	*Canis familiaris*	Mammals
Rabbit	*Leporidae sp*	Mammals
Mallard	*Anas platyrhynchos*	Birds
Bobwhite Quail	*Colinus virginianus*	Birds
Midge	*Chironomus tentans*	Aquatic invertebrates
Rainbow trout	*Oncorhynchus mykiss*	Fish/Salmonids
Fathead minnow	*Pimephales promelas*	Fish
American pondweed	*Potamogeton nodosus*	Non-target aquatic plants

TABLE 6-2
Surrogate Species Used in Quantitative ERA Evaluation

Species		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	*Odocolieus hemionus*	Mammalian herbivore/ gramivore	Ingestion
Bald eagle (northern)	*Haliaeetus leucocephalus alascanus*	Avian carnivore/ piscivore	Ingestion
Coyote	*Canis latrans*	Mammalian carnivore	Ingestion

TABLE 6-3
RTE Birds and Selected Surrogates

RTE Avian Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
Marbled murrelet	*Brachyramphus marmoratus marmoratus*	Piscivore	Bald eagle
Western snowy plover	*Charadrius alexandrinus nivosus*	Insectivore/ Piscivore	American robin
Piping plover	*Charadrius melodus*	Insectivore	American robin
Mountain plover	*Charadrius montanus*	Insectivore	American robin
Southwestern willow flycatcher	*Empidonax traillii extimus*	Insectivore	American robin
Northern aplomado falcon	*Falco femoralis septentrionalis*	Carnivore	Bald eagle Coyote
Cactus ferruginous pygmy-owl	*Glaucidium brasilianum cactorum*	Carnivore	Bald eagle Coyote
Whooping crane	*Grus Americana*	Piscivore	Bald eagle
California condor	*Gymnogyps californianus*	Carnivore	Bald eagle Coyote
Bald eagle	*Haliaeetus leucocephalus*	Piscivore	Bald eagle
Brown pelican	*Pelecanus occidentalis*	Piscivore	Bald eagle
Inyo California towhee	*Pipilo crissalis eremophilus*	Omnivore [Granivore/ Insectivore]	Canada goose American robin
Coastal California gnatcatcher	*Polioptila californica californica*	Insectivore	American robin
Stellar's eider	*Polysticta stelleri*	Piscivore	Bald eagle
Yuma clapper rail	*Rallus longirostris yumanensis*	Carnivore	Bald eagle Coyote
Spectacled eider	*Somateria fischeri*	Omnivore [Insectivore/ Herbivore]	American robin Canada goose
Least tern	*Sterna antillarum*	Piscivore	Bald eagle
Northern spotted owl	*Strix occidentalis caurina*	Carnivore	Bald eagle Coyote
Mexican spotted owl	*Strix occidentalis lucida*	Carnivore	Bald eagle Coyote
Least Bell's vireo	*Vireo bellii pusillus*	Insectivore	American robin

TABLE 6-4
RTE Mammals and Selected Surrogates

RTE Mammalian Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
Sonoran pronghorn	*Antilocapra americana sonoriensis*	Herbivore	Mule deer
Pygmy rabbit	*Brachylagus idahoensis*	Herbivore	Mule deer
Marbled murrelet	*Brachyramphus marmoratus marmoratus*	Piscivore	Bald eagle
Gray wolf	*Canis lupus*	Carnivore	Coyote
Utah prairie dog	*Cynomys parvidens*	Herbivore	Deer mouse
Morro Bay kangaroo rat	*Dipodomys heermanni morroensis*	Omnivore [Herbivore/ Insectivore]	Deer mouse American robin
Giant kangaroo rat	*Dipodomys ingens*	Granivore/ Herbivore	Deer mouse
Fresno kangaroo rat	*Dipodomys nitratoides exilis*	Granivore/ Herbivore	Deer mouse
Tipton kangaroo rat	*Dipodomys nitratoides nitratoides*	Granivore/ Herbivore	Deer mouse
Stephens' kangaroo rat	*Dipodomys stephensi (incl. D. cascus)*	Granivore	Deer mouse
Southern sea otter	*Enhydra lutris nereis*	Carnivore/ Piscivore	Coyote Bald eagle
Steller sea-lion	*Eumetopias jubatus*	Carnivore/ Piscivore	Coyote Bald eagle
Sinaloan jaguarundi	*Herpailurus (=Felis) yaguarundi tolteca*	Carnivore	Coyote
Ocelot	*Leopardus (=Felis) pardalis*	Carnivore	Coyote
Lesser long-nosed bat	*Leptonycteris curosoae yerbabuenae*	Frugivore/ Nectivore	Deer mouse
Mexican long-nosed bat	*Leptonycteris nivalis*	Herbivore	Deer mouse
Canada lynx	*Lynx canadensis*	Carnivore	Coyote
Amargosa vole	*Microtus californicus scirpensis*	Herbivore	Deer mouse
Hualapai Mexican vole	*Microtus mexicanus hualpaiensis*	Herbivore	Deer mouse
Black-footed ferret	*Mustela nigripes*	Carnivore	Coyote
Riparian (=San Joaquin Valley) woodrat	*Neotoma fuscipes riparia*	Herbivore	Deer mouse
Columbian white-tailed deer	*Odocolieus virginianus leucurus*	Herbivore	Mule deer
Bighorn sheep	*Ovis canadensis*	Herbivore	Mule deer
Bighorn sheep	*Ovis canadensis californiana*	Herbivore	Mule deer
Jaguar	*Panthera onca*	Carnivore	Coyote
Woodland caribou	*Rangifer tanandus caribou*	Herbivore	Mule deer
Northern Idaho ground squirrel	*Spermophilus brunneus brunneus*	Herbivore	Deer mouse
Grizzly bear	*Ursus arctos horribilis*	Omnivore [Herbivore/ Insectivore/ Piscivore]	American robin Mule deer Bald eagle
San Joaquin kit fox	*Vulpes macrotis mutica*	Carnivore	Coyote
Preble's meadow jumping mouse	*Zapus hudsonius preblei*	Omnivore [Herbivore/ Insectivore]	Deer mouse American robin

TABLE 6-5
RTE Reptiles and Selected Surrogates

RTE Reptilian Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
New Mexican ridge-nosed rattlesnake	*Crotalus willardi obscurus*	Carnivore/ Insectivore	Coyote/Bald eagle American robin
Blunt-nosed leopard lizard	*Gambelia silus*	Carnivore/ Insectivore	Coyote/Bald eagle American robin
Desert tortoise	*Gopherus agassizii*	Herbivore	Canada goose
Giant garter snake	*Thamnophis gigas*	Carnivore/ Insectivore/ Piscivore	Coyote American robin Bald eagle
Coachella Valley fringe-toed lizard	*Uma inornata*	Insectivore	American robin

Note: Five sea turtles are also listed species in the 17 states evaluated in this ERA. However, it is unlikely any exposure to herbicide would occur to marine species.

TABLE 6-6
RTE Amphibians and Selected Surrogates

RTE Amphibious Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
California tiger salamander	*Ambystoma californiense*	Invertivore[1] Vermivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
Sonoran tiger salamander	*Ambystoma tigrinum stebbinsi*	Invertivore, Insectivore[1] Carnivore, Ranivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
Desert slender salamander	*Batrachoseps aridus*	Invertivore	American robin[4,5]
Wyoming toad	*Bufo baxteri*	Insectivore	Bluegill sunfish/Rainbow trout[3] American robin[4]
Arroyo toad (=Arroyo southwestern toad)	*Bufo californicus*	Herbivore[1] Invertivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
California red-legged frog	*Rana aurora draytonii*	Herbivore[1] Invertivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
Chiricahua leopard frog	*Rana chiricahuensis*	Herbivore[1] Invertivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]

(1) Diet of juvenile (larval) stage.
(2) Diet of adult stage.
(3) Surrogate for juvenile stage.
(4) Surrogate for adult stage.
(5) *Bratrachoseps aridus* is a lungless salamander that has no aquatic larval stage, and is terrestrial as an adult.

TABLE 6-7

Species and Organism Traits That May Influence Herbicide Exposure and Response

Characteristic	Mode of Influence	ERA Solution
Body size	Larger organisms have more surface area potentially exposed during a direct spray exposure scenario. However, larger organisms have a smaller surface area to volume ratio, leading to a lower per body weight dose of herbicide per application event.	To evaluate potential impacts from direct spray, small organisms were selected (i.e., honeybee and deer mouse).
Habitat preference	Not all of BLM lands are subject to nuisance vegetation control.	It was assumed that all organisms evaluated in the ERA were present in habitats subject to herbicide treatment.
Duration of potential exposure /home range	Some species are migratory or present during only a fraction of year and larger species have home ranges that likely extend beyond application areas, thereby reducing exposure duration..	It was assumed that all organisms evaluated in the ERA were present within the zone of exposure full-time (i.e., home range = application area).
Trophic level	Many chemical concentrations increase in higher trophic levels.	Although the herbicides evaluated in the ERA have very low potential to bioaccumulate, BCFs were selected to estimate uptake to trophic level 3 fish (prey item for the piscivores), and several trophic levels (primary producers through top-level carnivore) were included in the ERA.
Food preference	Certain types of food or prey may be more likely to attract and retain herbicide.	It was assumed that all types of food were susceptible to high deposition and retention of herbicide.
Food ingestion rate	On a mass ingested per body weight basis, organisms with higher food ingestion rates (e.g., mammals versus reptiles) are more likely to ingest large quantities of food (therefore, herbicide).	Surrogate species were selected that consume large quantities of food, relative to body size. When ranges of ingestion rates were provided in the literature, the upper end of the values was selected for use in the ERA.
Foraging strategy	The way an organism finds and eats food can influence its potential exposure to herbicide. Organisms that consume insects or plants that are underground are less likely to be exposed via ingestion than those that consume exposed food items, such as grasses and fruits.	It was assumed all food items evaluated in the ERA were fully exposed to herbicide during spray or runoff events.
Metabolic and excretion rate	While organisms with high metabolic rates may ingest more food, they may also have the ability to excrete herbicides quickly, lowering the potential for chronic impact.	It was assumed that no herbicide was excreted readily by any organism in the ERA.
Rate of dermal uptake	Different organisms will assimilate herbicides across their skins at different rates. For example, thick scales and shells of reptiles and the fur of mammals are likely to present a barrier to uptake relative to bare skin.	It was assumed that uptake across the skin was unimpeded by scales, shells, fur, or feathers.
Sensitivity to herbicide	Species respond to chemicals differently; some species may be more sensitive to certain chemicals.	The literature was searched and the lowest values from appropriate toxicity studies were selected as TRVs. Choosing the sensitive species as surrogates for the TRV development provides protection to more species.
Mode of toxicity	Response sites to chemical exposure may not be the same among all species. For instance, the presence of aryl hydrocarbon (Ah) receptors in an organism increase its susceptibility to compounds that bind to proteins or other cellular receptors. However, not all species, even within a given taxonomic group (e.g., mammals) have Ah receptors.	Mode of toxicity was not specifically addressed in the ERA. Rather, by selecting the lowest TRV, it was assumed that all species evaluated in the ERA were also sensitive to the mode of toxicity.

TABLE 6-8
Summary of Findings: Interspecific Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within a Factor of:								
	2	4	10	15	20	50	100	250	300
Bird LD_{50}	--	--	90%	--	--	--	99%	100%	--
Mammal LD_{50}	--	58%	--	--	90%	--	96%	--	--
Bird and Mammal Chronic	--	--	--	--	--	94%	--	--	--
Plants	93%[(a)] 80%[(b)]	--	--	80%[(c)]	--	--	--	--	80%[(d)]

(a) Intra-genus extrapolation.
(b) Intra-family extrapolation.
(c) Intra-order extrapolation.
(d) Intra-class extrapolation.

TABLE 6-9
Summary of Findings: Intraspecific Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within Factor of 10	Citation from Fairbrother and Kaputska 1996
490 probit log-dose slopes	92%	Dourson and Starta 1983 *as cited in Abt Assoc., Inc. 1995*
Bird LC_{50}:LC_1	95%	Hill et al. 1975
Bobwhite quail LC_{50}:LC_1	71.5%	Shirazi et al. 1994

TABLE 6-10
Summary of Findings: Acute-to-Chronic Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within Factor of 10	Citation from Fairbrother and Kaputska 1996
Bird and mammal dietary toxicity NOAELs (n=174)	90%	Abt Assoc., Inc. 1995

TABLE 6-11
Summary of Findings: LOAEL-to-NOAEL Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within Factor of:		Citation from Fairbrother and Kaputska 1996
	6	10	
Bird and mammal LOAELs and NOAELs	80%	97%	Abt Assoc., Inc. 1995

TABLE 6-12
Summary of Findings: Laboratory to Field Extrapolations

Type of Data	Response	Citation from Fairbrother and Kaputska 1996
Plant EC_{50} Values	3 of 20 EC_{50} lab study values were 2-fold higher than field data. 3 of 20 EC_{50} values from field data were 2-fold higher than lab study data	Fletcher et al. 1990
Bobwhite quail	Shown to be more sensitive to cholinesterase-inhibitors when cold-stressed (i.e., more sensitive in the field).	Maguire and Williams 1987
Gray-tailed vole and deer mouse	Laboratory data over-predicted risk	Edge et al. 1995

7.0 UNCERTAINTY IN THE ECOLOGICAL RISK ASSESSMENT

Every time an assumption is made, some level of uncertainty is introduced into the risk assessment. A thorough description of uncertainties is a key component that serves to identify possible weaknesses in the ERA analysis, and to elucidate what impact such weaknesses might have on the final risk conclusions. This uncertainty analysis lists the uncertainties, with a discussion of what bias—if any—the uncertainty may introduce into the risk conclusions. This "bias" is represented in qualitative terms that best describe whether the uncertainty might 1) underestimate risk, 2) overestimate risk, or 3) be neutral with regard to the risk estimates, or whether it cannot be determined without additional study.

Uncertainties in the ERA process are summarized in Table 7-1. Several of the uncertainties warrant further evaluation and are discussed below. In general, the assumptions made in this risk assessment have been designed to yield a conservative evaluation of the potential risks to the environment from herbicide application.

7.1 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 often 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 actually overestimate risk relative to field studies (Fairbrother and Kapustka 1996).

Only one fluridone incident report was available from the USEPAs Environmental Fate and Effects Division (EFED). Incident reports can be used to validate both exposure models and hazards to ecological receptors. This report, described in Section 2.3, listed direct contact with fluridone as the "probable" cause of tomato plant damage. No terrestrial plant toxicity data was identified in the TRV derivation process, and impacts to terrestrial plants were not assessed in the risk assessment. This incident report suggests that impacts to non-target terrestrial plants may be of concern in accidental direct spray scenario. However, the use and severity of the impact were undetermined so it is impossible to correlate the concentrations predicted by the accidental spay scenario with the incident report.

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, yet they can be maintained under laboratory conditions. However, the selected toxicity value for a receptor was based on a thorough review of the available data by qualified toxicologists and the selection of the most appropriate sensitive surrogate species. The surrogate species used in the registration testing are not an exact match to the wildlife receptors included in the ERA. For example, the only avian data available is 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 or species from different taxonomic groups may be more or less sensitive to the herbicide than those species tested in the laboratory.

In general, the most sensitive available endpoint for the appropriate surrogate test species was used to derive TRVs. This is a conservative approach since there may be a wide range of data and effects for different species. This selection criterion for the TRVs has the potential to overestimate risk within the ERA. In some cases (i.e., coldwater fish), chronic data was unavailable and chronic TRVs were derived from acute toxicity data, adding an additional level of conservatism.

There is also some uncertainty in the conversion of food concentration-based toxicity values (mg herbicide per kg food) to dose-based values (mg herbicide per kg BW) for birds and mammals. Converting the concentration-based endpoint to a dose-based endpoint is dependent upon certain assumptions, specifically the test animal ingestion rate and test animal BW. Default ingestion rates for different test species were used in the conversions unless test-specific values were measured and given. The ingestion rate was assumed to be constant throughout a test. However, it is possible that a test chemical may positively or negatively affect ingestion, thus resulting in an over- or underestimation of total dose.

For the purposes of pesticide registration, tests are conducted according to specific test protocols. For example, in the case of an avian oral LD_{50} study, test guidance follows the harmonized Office of Pollution Prevention and Toxic Substances (OPPTS) protocol 850.2100, Avian Acute Oral Toxicity Test or its Toxic Substances Control Act (TSCA) or FIFRA predecessor (e.g, 40 CFR 797.2175 and OPP 71-1). In this test the bird is given a single dose, by gavage, of the chemical and the test subject is observed for a minimum of 14 days. The LD_{50} derived from this test is the true dose (mg herbicide per kg BW). However, dietary studies were selected preferentially for this ERA and historical dietary studies followed 40 CFR 797.2050, OPP 71-2, or OECD 205, the procedures for which are harmonized in OPPTS 850.2200, Avian Dietary Toxicity Test. In this test, the test organism is presented with the dosed food for 5 days, with 3 days of additional observations after the chemical-laden food is removed. The endpoint for this assay is reported as an LC_{50} representing mg herbicide per kg food. For this ERA, the concentration-based value was converted to a dose-based value following the methodology presented in the Methods Document (ENSR 2004c)[10]. Then the dose-based value was multiplied by the number of days of exposure (generally 5) to result in an LD_{50} value representing the full herbicide exposure over the course of the test.

For fluridone, no toxicity data was identified for terrestrial plant species. This is a type of testing generally required for pesticide registrations, but no information was identified in the FOIA review or other sources. This results in a data gap, and therefore no quantitative evaluation of potential risks to non-target terrestrial plants was possible in the risk assessment. As discussed above, one ecological incident was reported, which associated impacts to tomato plants with fluridone. In addition the manufacturer's user's guide for the Sonar aquatic herbicide (Eli Lilly and Company 2003), indicated that some upland terrestrial species (i.e., grasses, sedges) are considered to be "sensitive" or "intermediate" in their tolerance to the herbicide, while shoreline plants, (i.e., willow, cypress), were considered "tolerant." The Sonar labels (SePRO 2002a,b,c; SePRO 2003) warn against using treated water for irrigation purposes for seven to thirty days after treatment. Even at the low fluridone concentrations used to treat milfoil, some terrestrial plants may be sensitive to fluridone if they are watered with treated lake water. The incident report, the user's guide, and the herbicide labels indicate that fluridone may cause negative impacts to terrestrial plants (e.g., tomatoes, grasses, sedges), but that shoreline plants are more tolerant. It is these more tolerant shoreline plants that are more likely to come in contact with fluridone during normal pond applications.

As indicated in Section 3.1, the toxicity data within the ERAs are presented in the units used in the reviewed studies. Attempts were not made to adjust toxicity data to the % a.i. since it was not consistently provided in all reviewed materials. In most cases the toxicity data applies to the a.i. itself; however, some data corresponds to a specific product containing the a.i. under consideration, and potentially other ingredients (e.g., other a.i. or inert ingredients). The assumption has been made that the toxicity observed in the tests is due to the a.i. under consideration. However, it is possible that the additional ingredients in the different formulations also had an effect. The OPP's Ecotoxicity Database (a source of data for the ERAs) does not adjust the toxicity data to the % a.i. and presents the data directly from the registration study in order to capture the potential effect caused by various inerts, additives, or other a.i. in the tested product. In many cases the tested material represents the highest purity produced and higher exposure to the a.i. would not be likely.

For fluridone, the percent a.i., listed in Appendix A when available from the reviewed study, ranged from 0.48% to 99%. The lowest % a.i. used in the actual TRV derivation was 33.3% in the study used to derive the acute TRV for the honeybee. Adjusting the TRV to 100% of the a.i. (by multiplying the TRV by the % a.i. in the study) would lower

[10] Dose-based endpoint $_{(mg/kg\ BW/day)}$ = [Concentration-based endpoint $_{(mg/kg\ food)}$ x Food Ingestion Rate $_{(kg\ food/day)}$]/BW $_{(kg)}$

the bee TRV from 1,088 ug/bee to 362 ug/bee. Although this would increase the associated RQs, it would not result in any additional LOC exceedances. The remaining TRVs are based on studies with at least 95% a.i., so the RQ changes would be minimal. Several of the fish studies included in Appendix A were conducted with products containing 41 to 48% fluridone. However, to reduce the uncertainties in whether the toxicity in these studies was due to fluridone or to other components, the values selected to derive the fish TRVs were based on studies containing 89 to 99% fluridone. Selection of alternative studies and adjustment to reflect the % a.i. could result in a lower TRV[11], but there would be a level of uncertainty in this TRV due to the potential toxicity of the other components in the product.

7.2 Potential Indirect Effects on Salmonids

No actual field studies or ecological incident reports related to the effects of fluridone on salmonids were identified during the ERA. Therefore, any discussion of direct or indirect impacts to salmonids was limited to qualitative estimates of potential impacts to salmonid populations and communities. The acute fish TRV used in the risk assessment was based on laboratory studies conducted with a salmonid, the rainbow trout, reducing the uncertainties in this evaluation.

A discussion of the potential indirect impacts to salmonids is presented in Section 4.3.6, and Section 6.6 provides a discussion of RTE salmonid species. These evaluations indicated that, in the conservative accidental exposure scenarios evaluated, salmonids may be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates), but not a reduction in aquatic vegetative cover.

It is anticipated that these qualitative evaluations over-estimate the potential risk to salmonids due to the conservative selection of TRVs for salmonid prey and vegetative cover, 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).

7.3 Ecological Risks of 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 inert ingredients (inerts), adjuvants, surfactants, and degradates. Other herbicides may also factor into the risk estimates, as many herbicides can be tank mixed to expand the level of control and to accomplish multiple identified tasks. However, using currently available models (e.g., AgDRIFT®), it is only practical to calculate deterministic risk calculations (i.e., exposure modeling, effects assessment, and RQ calculations) 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 potential effects for risks from inert ingredients, adjuvants, and tank mixtures.

7.3.1 Degradates

The potential toxicity of degradates, also called herbicide transformation products (TPs), 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 containing fluridone. Degradates may be more or less mobile and

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[11] Selection of the channel catfish study conducted using 41% fluridone and adjustment of that 96 hour LC_{50} (13.2 mg/L) to reflect the % active ingredient would result in a warm water fish acute TRV of 5.4 mg/L. This value is lower than the selected value of 8.2 mg/L conducted with a product containing 98 to 99% fluridone.

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 TPs makes prediction of potential TP impacts challenging. For example, a less toxic, but more mobile bioaccumulative, or persistent TP may have the potential to have a greater adverse impact on the environment resulting from residual concentrations in the environment. A recent study indicated that 70% of TPs had either similar or reduced toxicity to fish, daphnids, and algae than the parent pesticide. However, 4.2% of the TPs 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 evaluation of impacts to terrestrial species was conducted in this study. The lack of data on the toxicity of degradates of fluridone represents a source of uncertainty in the risk assessment.

7.3.2 Inerts

Pesticide products contain both active and inert ingredients. The terms “active ingredient” and “inert ingredient” have been defined by Federal law—the 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.

In September 1997, the USEPA issued Pesticide Regulation Notice 97-6, which encouraged manufacturers, formulators, producers, and registrants of pesticide products to voluntarily substitute the term “other ingredients” as a heading for the inert ingredients in the ingredient statement. The USEPA made this change after learning the results of a consumer survey on the use of household pesticides. Many consumers are misled by the term “inert ingredient,” believing it to mean “harmless.” Since 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. Whether referred to as “inerts” or “other ingredients,” these components within an herbicide have the potential to be toxic.

BLM scientists received clearance from the USEPA to review CBI on inert compounds in the following herbicides under consideration in ERAs: bromacil, chlorsulfuron, diflufenzopyr, Overdrive® (a mix of dicamba and diflufenzopyr), diquat, diuron, fluridone, imazapic, sulfometuron-methyl, and tebuthiuron. The information received listed the inert ingredients, their chemical abstract number, supplier, USEPA registration number, percentage of the formulation and purpose in the formulation. This information is confidential, and is therefore not disclosed in this document. However, a review of available data for the herbicides is included in Appendix D.

The USEPA has a listing of regulated inert ingredients at http://www.epa.gov/opprd001/inerts/index.html. This listing categorizes inert ingredients into four lists. The listing of categories and the number of inert ingredients found among the ingredients listed for the herbicides are shown below:

- 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. Over 50.

Nine inerts were not found on EPA’s lists.

Toxicity information was also searched in the following sources:

- TOMES (a proprietary toxicological database including EPA's IRIS, the Hazardous Substance Data Bank, the Registry of Toxic Effects of Chemical Substances [RTECS]).
- EPA'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 (MSDS) 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 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 MSDSs or published data.

As a tool to evaluate List 3 and unlisted inerts in the ERA, the exposure concentration of the inert compound was calculated and compared to toxicity information. As described in more detail in Appendix D, toxicity information from the above sources was used in addition to the work of Muller (1980), Lewis (1991), Dorn et al. (1997), and Wong et al. (1997) concerning aquatic toxicity of surfactants. These sources generally suggested that acute toxicity to aquatic life for surfactants and anti-foam agents ranged from 1 to 10 mg/L, and that chronic toxicity ranged as low as 0.1 mg/L.

Appendix D presents the following general observation for fluridone: low application rates for fluridone resulted in low exposure concentrations of inerts of much < 1 mg/L in all modeled cases. This indicates that inerts associated with the application of fluridone are not predicted to occur at levels that would cause acute toxicity to aquatic life. However, given the lack of specific inert toxicity data, it is not possible to state that the inerts in fluridone will not result in adverse ecological impacts. 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 ingredients.

7.3.3 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, the composition of such mixtures is highly site-specific, and thus nearly impossible to address at the level of the programmatic EIS.

Herbicide label information indicates whether a particular herbicide can be tank mixed with other pesticides. Adjuvants, such as surfactants, crop oil concentrates, fertilizers, etc., may also be added to the spray mixture to improve the herbicide efficacy. 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.

7.3.3.1 Adjuvants

Adjuvants generally function to enhance or prolong the activity of an a.i. For terrestrial herbicides, adjuvants aid in the absorption of the a.i. into plant tissue. Adjuvant is a broad term and 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 and the USEPA does not register or approve the labeling of spray adjuvants. Individual herbicide labels identify which types of adjuvants are approved for use with a particular herbicide.

In reviewing the labels of the a.i. fluridone, it is noted that there is not discussion regarding the addition of an adjuvant, indicating that the herbicide does not need to have an adjuvant added to the spray mixture in order to manage the vegetation. If an adjuvant is considered in the future, it is recommended that a compound with low toxicity and low required volumes be selected to reduce the potential for the adjuvant to influence the toxicity of the herbicide.

7.3.3.2 Tank Mixtures

In reviewing various labels of the different formulations of fluridone, the tank mixing of other aquatic herbicides is presented as an option, but the specific a.i. are not identified. However, it is not generally within BLM practice to tank mix fluridone with any other products. Therefore, additional modeling of tank mixes was not performed for fluridone.

In general it may be noted that 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 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 already have the potential for risk from an individual herbicide (e.g., runoff to ponds in sandy watersheds). Use of a tank mix under these conditions is likely to increase the level of uncertainty in the potential unintended risk to the environment.

7.4 Uncertainty Associated with Herbicide Exposure Concentration Models

The ERA relies on different models to predict the off-site impacts of herbicide use. These models have been developed and applied in order to develop a conservative estimate of herbicide loss from the application area to the off-site locations.

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 biggest numeric impact on model outputs. This has important implications not only for the uncertainty analysis itself, but also for the ability to apply risk calculations to different site characteristics from a risk management point of view.

7.4.1 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 ERA 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 BLM 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 including, but not limited to: 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 lack of absolute knowledge regarding all of the scenarios likely to be encountered in the BLM land management program, these assumptions were developed to be conservative and likely result in overestimation of actual off-site spray drift and environmental impacts.

7.5 Summary of Potential Sources of Uncertainty

The analysis presented in this section has identified several potential sources of uncertainty that may introduce bias into the risk conclusions. This bias has the potential to 1) underestimate risk, 2) overestimate risk, or 3) be neutral with regard to the risk estimates, or be undetermined without additional study. In general, few of the sources of uncertainty in this ERA are likely to underestimate risk to ecological receptors. Risk is more likely to be overestimated or the impacts of the uncertainty may be neutral or impossible to predict.

The following bullets summarize the potential impacts on the risk predictions based on the analysis presented above:

- Toxicity Data Availability – Although the species for which toxicity data are available may not necessarily be the most sensitive species to a particular herbicide, the TRV selection methodology has focused on identifying conservative toxicity values that are likely to be protective of most species; the use of various LOCs contributes an additional layer of protection for species that may be more sensitive than the tested species (i.e., RTE species).

- Potential Indirect Effects on Salmonids – Only a qualitative evaluation of indirect risk to salmonids was possible since no relevant studies or incident reports were identified; it is likely that this qualitative evaluation overestimates the potential risk to salmonids due to the numerous conservative assumptions related to TRVs and exposure scenarios, and the application of additional LOCs (with uncertainty/safety factors applied) to assess risk to RTE species.

- Ecological Risks of Degradates, Inerts, Adjuvants, and Tank Mixtures – Only limited information is available regarding the toxicological effects of degradates, inerts, adjuvants, and tank mixtures; in general, it is unlikely that highly toxic degradates or inerts are present in approved herbicides. Also, selection of tank mixes and adjuvants is under the control of BLM land managers and to reduce uncertainties and potential risks products should be thoroughly reviewed and mixtures with the least potential for negative effects should be selected.

- Uncertainty Associated with Herbicide Exposure Concentration Models – Environmental characteristics (e.g., soil type, annual precipitation) will impact the three models used to predict the off-site impacts of herbicide use (i.e., AgDRIFT, GLEAMS, CALPUFF); in general, the assumptions used in the models were developed to be conservative and likely result in overestimation of actual off-site environmental impacts.

- General ERA Uncertainties – The general methodology used to conduct the ERA is more likely to overestimate risk than to underestimate risk due to the use of conservative assumptions (i.e., entire home range and diet is assumed to be impacted, aquatic waterbodies are relatively small, herbicide degradation over time is not applied in most scenarios).

TABLE 7-1
Potential Sources of Uncertainty in the ERA Process

Potential Source of Uncertainty	Direction of Effect	Justification
Physical-chemical properties of the active ingredient	Unknown	Available sources were reviewed for a variety of parameters. However, not all sources presented the same value for a parameter (e.g., water solubility) and some values were estimated.
Food chain assumed to represent those found on BLM lands	Unknown	BLM lands cover a wide variety of habitat types. A number of different exposure pathways have been included, but additional pathways may occur within management areas.
Receptors included in food chain model assumed to represent those found on BLM lands	Unknown	BLM lands cover a wide variety of habitat types. A number of different receptors have been included, but alternative receptors may occur within management areas.
Food chain model exposure parameter assumptions	Unknown	Some exposure parameters (e.g., body weight, food ingestion rates) were obtained from the literature and some were estimated. Efforts were made to select exposure parameters representative of a variety of species or feeding guilds.
Assumption that receptor species will spend 100% of time in impacted aquatic or terrestrial area (home range = application area)	Overestimate	These model exposure assumptions do not take into consideration the ecology of the wildlife receptor species. Organisms will spend varying amounts of time in different habitats, thus affecting their overall exposures. Species are not restricted to one location within the application area, may migrate freely off-site, may undergo seasonal migrations (as appropriate), and are likely to respond to habitat quality in determining foraging, resting, nesting, and nursery activities. A likely overly conservative assumption has been made that wildlife species obtain all their prey items from the application area.
Waterbody characteristics	Overestimate	The pond and stream were designed with conservative assumptions resulting in relatively small volumes. Larger waterbodies are likely to exist within application areas.
Extrapolation from test species to representative wildlife species	Unknown	Species differ with respect to absorption, metabolism, distribution, and excretion of chemicals. The magnitude and direction of the difference may vary with species. It should be noted, though, that in most cases, laboratory studies actually overestimate risk relative to field studies (Fairbrother and Kapustka 1996).
Consumption of contaminated prey	Unknown	Toxicity to prey receptors may result in sickness or mortality. Fewer prey items would be available for predators. Predators may stop foraging in areas with reduced prey populations, discriminate against, or conversely, select contaminated prey.
No evaluation of inhalation exposure pathways	Underestimate	The inhalation exposure pathways are generally considered insignificant due to the low concentration of contaminants under natural atmospheric conditions. However, under certain conditions, these exposure pathways may occur.
Assumption of 100% drift for chronic ingestion scenarios	Overestimate	It is unlikely that 100% of the application rate would be deposited on a plant or animal used as prey by another receptor. As indicated with the AgDRIFT® model (used to evaluate other herbicides in the EIS), off-site drift is only a fraction of the applied amount.
Ecological exposure concentration	Overestimate	It is unlikely any receptor would be exposed continuously to the full predicted EEC.
Oversimplification of dietary composition in food web models	Unknown	Assumptions were made that contaminated prey (e.g., vegetation, fish) were the primary prey items for wildlife. In reality, other prey items are likely consumed by these organisms.

TABLE 7-1 (Cont.)
Potential Sources of Uncertainty in the ERA Process

Potential Source of Uncertainty	Direction of Effect	Justification
Degradation or adsorption of herbicide	Overestimate	Risk estimates for direct spray and off-site drift scenarios generally do not consider degradation or adsorption. Concentrations will tend to decrease over time from degradation. Organic carbon in water or soil/sediment may bind to herbicide and reduce bioavailability.
Bioavailability of herbicides	Overestimate	Most risk estimates assume a high degree of bioavailability. Environmental factors (e.g., binding to organic carbon, weathering) may reduce bioavailablity.
Limited evaluation of dermal exposure pathways	Unknown	The dermal exposure pathway is generally considered insignificant due to natural barriers found in fur and feathers of most ecological receptors. However, under certain conditions (e.g., for amphibians), these exposure pathways may occur.
Amount of receptor's body exposed	Unknown	More or less than ½ of the honeybee or small mammal may be affected in the accidental direct spray scenarios.
Lack of toxicity information for amphibian and reptile species	Unknown	Information is not available on the toxicity of herbicides to reptiles and amphibians resulting from dietary or direct contact exposures.
Lack of toxicity information for RTE species	Unknown	Information is not available on the toxicity of herbicides to RTE species resulting from dietary or direct contact exposures. Uncertainty factors have been applied to attempt to assess risk to RTE receptors. See Section 7.2 for additional discussion of salmonids.
Safety factors applied to TRVs	Overestimate	Assumptions regarding the use of 3-fold uncertainty factors are based on precedent, rather than scientific data.
Use of lowest toxicity data to derive TRVs	Overestimate	The lowest data point observed in the laboratory may not be representative of the actual toxicity that might occur in the environment. Using the lowest reported chronic toxicity data point as a benchmark concentration is a very conservative approach, especially when there is a wide range in reported toxicity values for the relevant species. See Section 7.1 for additional discussion.
Use of NOAELs	Overestimate	Use of NOAELs may over-estimate effects since this measurement endpoint does not reflect any observed impacts. LOAELs may be orders of magnitudes above observed literature-based NOAELs, yet NOAELs were generally selected for use in the ERA.
Use of chronic exposures to estimate effects of herbicides on receptors	Overestimate	Chronic toxicity screening values assume that ecological receptors experience continuous, chronic exposure. Exposure in the environment is unlikely to be continuous for many species that may be transitory and move in and out of areas of maximum herbicide concentration.
Use of measures of effect	Overestimate	Although an attempt was made to have measures of effect reflect assessment endpoints, limited available ecotoxicological literature resulted in the selection of certain measures of effect that may overestimate assessment endpoints.
Lack of toxicity information for mammals or birds	Unknown	TRVs for certain receptors were based on a limited number of studies conducted primarily for pesticide registration. Additional studies may indicate higher or lower toxicity values. See Section 7.1 for additional discussion.
Lack of seed germination toxicity information	Unknown	TRVs were based on a limited number of studies conducted primarily for pesticide registration. A wide range of germination data was not always available. Emergence or other endpoints were also used and may be more or less sensitive to the herbicide.

TABLE 7-1 (Cont.)
Potential Sources of Uncertainty in the ERA Process

Potential Source of Uncertainty	Direction of Effect	Justification
Species used for testing in the laboratory assumed to be equally sensitive to herbicide as those found within application areas.	Unknown	Laboratory toxicity tests are normally conducted with species that are highly sensitive to contaminants in the media of exposure. Guidance manuals from regulatory agencies contain lists of the organisms that they consider to be sensitive enough to be protective of naturally occurring organisms. However, reaction of all species to herbicides is not known, and species found within application areas may be more or less sensitive than those used in the laboratory toxicity testing. See Section 7.1 for additional discussion.
Risk evaluated for individual receptors only	Overestimate	Effects on individual organisms may occur with little population or community level effects. However, as the number of affected individuals increases, the likelihood of population-level effects increases.
Lack of predictive capability	Unknown	The RQ approach provides a conservative estimate of risk based on a "snapshot" of conditions; this approach has no predictive capability.
Unidentified stressors	Unknown	It is possible that physical stressors other than those measured may affect ecological communities.
Effect of decreased prey item populations on predatory receptors	Unknown	Adverse population effects to prey items may reduce the foraging population for predatory receptors, but may not necessarily adversely impact the population of predatory species.
Multiple conservative assumptions	Overestimate	Cumulative impact of multiple conservative assumptions predicts high risk to ecological receptors.
Impact of the other ingredients (e.g., inerts, adjuvants) in the application of the herbicide	Unknown	Only the active ingredient has been investigated in the ERA. Inerts, adjuvants, and tank mixtures may increase or decrease the impacts of the active ingredient. These uncertainties are discussed further in Section 7.3.

8.0 SUMMARY

Based on the ERA conducted for fluridone, there is the potential for risk to selected ecological receptors from exposure to herbicides under specific conditions on BLM-managed lands. Table 8-1 summarizes the relative magnitude of risk predicted for ecological receptors for each route of exposure. This was accomplished by comparing the RQs against the most conservative LOC, and ranking the results for each receptor-exposure route combination from 'no potential' to 'high potential' for risk. As expected, accidental exposure scenarios (i.e., direct spray and accidental spills) may result in risk for non-target species (i.e., fish, aquatic invertebrates).

The following bullets summarize the risk assessment findings for fluridone under these conditions:

- Direct Spray – No acute risks were predicted for terrestrial wildlife (i.e., insects, birds, or mammals). Chronic risk was only predicted for one receptor scenario, the small mammalian herbivore at the maximum application rate. All other terrestrial animal exposure scenarios had RQs below the associated LOC. Risks to terrestrial plants could not be evaluated as a result of a lack of toxicity information; however, one ecological incident report suggests the potential for risk to terrestrial plants. No risks to non-target aquatic plants are predicted when waterbodies are accidentally (streams) or intentionally (ponds) sprayed, but risks to fish or aquatic invertebrates may occur when waterbodies are accidentally or intentionally sprayed.

- Off-Site Drift to Non-Target Terrestrial Plants – Risks to terrestrial plants could not be evaluated because of a lack of toxicity information; however, product literature and one ecological incident report suggest the potential for risk.

- Accidental Spill to Pond – Risk to fish, aquatic invertebrates, and non-target aquatic plants may occur when herbicides are spilled directly into the pond.

Based on the results of the ERA, it is unlikely that RTE species would be harmed by appropriate use of the herbicide fluridone on BLM-managed lands.

8.1 Recommendations

The following recommendations are designed to reduce potential unintended impacts to the environment from the application of fluridone:

- Select adjuvants carefully (none are currently ingredients in fluridone-containing Sonar products) since these have the potential to increase the level of toxicity above that predicted for the a.i. alone. This is especially important for application scenarios that already predict potential risk from the a.i. itself.

- Review, understand, and conform to "Environmental Hazards" section on 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 the environment.

- Avoid accidental direct spray on the stream to reduce the most significant potential impacts.

- Use the typical application rate in the pond, rather than the maximum application rate, to reduce risk to fish and aquatic invertebrates.

- Because the effects of normal herbicide application on terrestrial plants are uncertain, limit fluridone use in areas where RTE plants are near application areas. Avoid accidental direct spray and off-site drift to terrestrial plants to reduce potential impacts observed in a previous ecological incident report (Section 2.3).

- Observe buffer areas of at least 100 ft from terrestrial habitats for plane and helicopter application of fluridone if potential impacts to terrestrial RTE species are of concern.

- Limit fluridone application in wind, and monitor effects on adjacent terrestrial vegetation.

The results from this ERA assist the evaluation of proposed alternatives in the EIS and contribute to the development of a BA, specifically addressing the potential impacts to proposed and listed RTE species on western BLM treatment lands. Furthermore, this ERA will inform BLM field offices on the proper application of fluridone to ensure that impacts to plants and animals and their habitat are minimized to the extent practical.

TABLE 8-1
Typical Risk Levels Resulting from Fluridone Application

Exposure Category	Direct Spray/Spill		Off-Site Drift		Surface Runoff		Wind Erosion	
Receptor Group	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate
Terrestrial Animals	0 [16: 16]	0 [15: 16]	NE	NE	NA	NA	NA	NA
Terrestrial Plants (Typical Species)	NE	NE	NE	NE	NA	NA	NA	NA
Terrestrial Plants (RTE Species)	NE	NE	NE	NE	NA	NA	NA	NA
Fish In The Pond	0 [2: 2]	M [2: 4]	NA	NA	NA	NA	NA	NA
Fish In The Stream	0 [2: 2]	L [2: 2]	NA	NA	NA	NA	NA	NA
Aquatic Invertebrates In The Pond	0 [2: 2]	H [1: 4]	NA	NA	NA	NA	NA	NA
Aquatic Invertebrates In The Stream	L [1: 2]	M [1: 2]	NA	NA	NA	NA	NA	NA
Aquatic Plants In The Pond	0 [2: 2]	L [2: 4]	NA	NA	NA	NA	NA	NA
Aquatic Plants In The Stream	0 [2: 2]	0 [2: 2]	NA	NA	NA	NA	NA	NA
Piscivorous Bird	0 [1:1]	0 [1:1]	NA	NA	NA	NA	NA	NA

Risk Levels:
0 = No Potential for Risk (majority of RQs < most conservative LOC).
L = Low Potential for Risk (majority of RQs 1-10 times the most conservative LOC).
M = Moderate Potential for Risk (majority of RQs 10-100 times the most conservative LOC).
H = High Potential for Risk (majority of RQs >100 times the most conservative LOC).
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, 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 to determine the specific scenarios that result in the displayed level of risk for a given receptor group.
Number in brackets represents Number of RQs in the Indicated Risk Level: Number of Scenarios Evaluated.
NA = Not applicable. No RQs calculated for this scenario.
In cases of a tie, the more conservative (higher) risk level was selected.

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