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# Diquat Ecological Risk Assessment, Final Report

Bureau of Land Management

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# Bureau of Land Management

Reno, Nevada

Diquat
Ecological Risk Assessment

Final Report

**November 2005**

**Bureau of Land Management Contract No. NAD010156**
**ENSR Document Number 09090-020-650**

# Executive Summary

The United States Department of the Interior (USDI) Bureau of Land Management (BLM) 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 (US) 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 diquat, 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

Diquat is a nonselective contact herbicide for the control of broad-leaf aquatic weeds post-emergence. This herbicide is a cell membrane disrupter that is activated by exposure to sunlight to form oxygen compounds that damage cell membranes.

Diquat is used by the BLM for vegetation control in their Aquatics program. Application (to ponds but not streams) 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 and from all terrain vehicles or trucks equipped with spot or boom/broadcast sprayers. The BLM typically applies diquat at a rate of 1.0 pounds (lbs) a.i. per acre (a.i./ac). The maximum application rate is 4.0 lbs a.i./ac.

### Ecological Risk Assessment Guidelines

The main objectives of this ERA were to evaluate the potential ecological risks from diquat 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 diquat 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 Observed 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 diquat 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 US 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 diquat with 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, diquat has been associated with ten reported “ecological incidents” involving damage or mortality to non-target flora or fauna. It was listed as probable (seven incidents) or possible (three incidents) that registered use of diquat was responsible.

A review of the available ecotoxicological literature was conducted in order to evaluate the potential for diquat 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 diquat is moderately toxic to mammals, particularly via dermal exposure. Diquat is also moderately toxic to birds and honeybees (*Apis* spp.). In addition, adverse effects to non-target terrestrial plant species occurred with exposure to low concentrations of diquat (0.0047 lbs a.i./ac). Diquat has relatively low toxicity to fish and moderate toxicity to aquatic invertebrates. Diquat does not appear to appreciably bioconcentrate in fish tissue. Aquatic macrophytes were adversely affected by diquat concentrations as low as 0.00075 milligrams (mg) /liter (L) (typical herbicide application rates in this ERA resulted in pond concentrations of 0.11 mg a.i./L and stream concentrations of 0.56 mg a.i./L). There did not appear to be appreciable differences in

sensitivities among aquatic macrophytes, diatoms, and algae. No acute toxicity studies conducted on amphibian species were found in the literature reviewed.

# Ecological Risk Assessment Results

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

- Direct Spray – Risks were predicted for pollinating insects due to direct spray and indirect contact with contaminated foliage. Acute risks were predicted for 7 of 10 wildlife scenarios, and chronic risks were predicted for 6 of 10 wildlife scenarios (risk is somewhat lower at the typical application rate versus the maximum rate). Risk to terrestrial and aquatic non-target plants, fish, and aquatic invertebrates is likely when waterbodies are accidentally or intentionally (pond applications) sprayed. No risks were predicted for piscivorous wildlife.

- Off-Site Drift to Non-Target Terrestrial Plants – Risks to typical and RTE terrestrial plant species were predicted within 900 feet (ft) of all aerial applications at both the typical and maximum application rates. Risk quotients were also elevated for typical and RTE plant species within 100 ft of ground applications (low and high booms).

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

In addition, species that depend on non-target species for habitat, cover, and/or food may be indirectly impacted by possible reductions in terrestrial or aquatic vegetation or effects on terrestrial and aquatic wildlife, particularly in accidental direct spray and spill scenarios. For example, RTE salmonids may be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates) or a reduction in vegetative cover in the relatively unlikely case of an accidental spill to the stream.

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 diquat on BLM-managed lands. Although non-target terrestrial plants have the potential to be adversely affected by off-site drift of diquat, adherence to certain application guidelines (e.g., defined application rates, equipment, herbicide mixture, and downwind distance to potentially sensitive habitat) would minimize the potential effects on non-target plants and associated indirect effects on species that depend on those plants for food, habitat, and cover.

# Recommendations

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

- Select herbicide products carefully to minimize additional impacts from adjuvants. 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 and spill conditions to reduce the greatest potential impacts.

- Use the typical application rate, rather than the maximum application rate, to reduce risk to some wildlife receptors (e.g., large carnivore, small bird).

- Use ground application methods at the typical or maximum application rate with buffer zones of more than 100 ft to reduce potential risks to typical terrestrial plants from off-site drift.

- To reduce potential risks to RTE terrestrial plants from off-site drift during ground applications, use buffer zones of more than 100 ft at the typical rate, or buffer zones of at least 1,000 ft when using the maximum application rate (based on basic regression evaluation of RQs and distance).

- Buffer zones of greater than 900 ft may be necessary if aerial application is planned. Regression analysis indicates that a buffer zone of 1,200 ft should be protective of typical and RTE plant species.

- Avoid application to ponds where RTE species have been identified.

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 diquat 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 |
| CALPUFF | - | California Puff |
| CBI | - | Confidential Business Information |
| 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) |
| Ed. | - | Edition |
| 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 |
| 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 |
| MRID | - | Master Record Identification Number |
| MW | - | Molecular Weight |
| 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 Library |
| 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 the 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 United States Department of the Interior Bureau of Land Management (USDI BLM) is proposing a program to treat vegetation on up to six million acres of public lands annually in 17 western states in the continental US 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 is preparing a *Vegetation Treatments Programmatic EIS* to evaluate proposed vegetation treatment methods and alternatives on lands managed by the BLM in the western continental US and Alaska (USDI BLM 2005). 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 active ingredients 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 active ingredients (sulfometuron-methyl, imazapic, diflufenzopyr, dicamba, diquat, and fluridone) because the other active ingredients 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 active ingredients 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 diquat.

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 in three models utilized for this ERA (exposure point modeling using Groundwater Loading Effects of Agricultural Management Systems [GLEAMS], AgDRIFT®, and California Puff [CALPUFF]).

## 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 diquat, 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 diquat 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 diquat for the management of 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 proposed BLM use. Diquat application rates and methods discussed in this section are based on proposed BLM herbicide use and are in accordance with herbicide 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.

Diquat is a nonselective contact herbicide for the management of aquatic weed species. This herbicide is a cell membrane disrupter that is activated by exposure to sunlight to form oxygen compounds that damage cell membranes.

Diquat is proposed for use by the BLM in their Aquatics vegetation management program. The majority of application occurs in inland freshwater habitats; diquat is rarely used in marine or estuarine habitats. 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 and from all terrain vehicles or trucks equipped with spot or boom/broadcast sprayers. In addition, applications will be made using a boat with either a handgun, which will be used to make spot treatments, or a boom, which will be used to make broadcast applications on top of or under the water surface. The BLM typically applies diquat at a rate of 1.0 lbs a.i./ac. The maximum application rate is 4.0 lbs a.i./ac. Details regarding expected diquat usage by the 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 diquat as a potential source of the observed ecological damage.

The USEPA EIIS contained ten incident reports involving diquat. In seven incidents, mortality of fish and other fauna were attributed to diquat; three incidents involved plant damage. Two of the seven incidents had references to additional pesticide use (malathion and copper sulfate). The incident reports listed the likelihood that diquat actually caused the observed damaged as "probable" in seven incidents and "possible" in three incidents. A summary of these incidents is provided in Table 2-2 at the end of this section.

**TABLE 2-1**

**BLM Diquat 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 | 1.0 | 4.0 |
| | | Helicopter | Rotary | Yes | 1.0 | 4.0 |
| | Ground | Human | Backpack | Yes | 1.0 | 4.0 |
| | | | Horseback | Yes | 1.0 | 4.0 |
| | | ATV | Spot | Yes | 1.0 | 4.0 |
| | | | Boom/Broadcast | Yes | 1.0 | 4.0 |
| | | Truck | Spot | Yes | 1.0 | 4.0 |
| | | | Boom/Broadcast | Yes | 1.0 | 4.0 |

**TABLE 2-2**

**Diquat Incident Report Summary**

| Year | Application Area | Incident Type | Diquat Certainty | Other[1] | Dispersal | Organism | Distance[2] | Magnitude of Damage |
|---|---|---|---|---|---|---|---|---|
| 1982 | NA | Undetermined | Probable | Yes | Direct | Spot, Mullet, Trout, Drum, | Vicinity | Mortality |
| 1993 | Pond | Registered Use | Possible | No | Direct | Carp, Bream | Adjacent | Mortality - 15 carp, 10 bream |
| 1994 | Drain | Intentional Misuse | Probable | No | NA | Fish | Vicinity | Mortality - Unknown |
| 1994 | Municipal Operation | Undetermined | Possible | Yes | Direct | Frogs, Wildflowers | Vicinity | Mortality - Unknown |
| 1995 | Agricultural | Accident | Probable | No | Drift | Strawberry | Adjacent | Unknown |
| 1998 | Pond | Registered Use | Probable | No | Direct | Bass | 0 | Mortality |
| 1999 | Agricultural | Misuse | Probable | No | Drift | Alfalfa | NA | Plant damage |
| 2001 | Pond | Accident | Probable | No | Direct | Turtle, Alligator | 0 | Mortality - 400 turtles, 3 alligator |
| 2002 | Lake | Registered Use | Probable | Yes | Direct | Clams | 0 | Mortality - thousands |
| 2003 | NA | Undetermined | Possible | No | Drift | Spinach | NA | Plant damage - unknown |

[1] Other = other chemicals used in conjunction with diquat (yes/no).
[2] Distance = estimated distance from application area.
NA = information not available.

# 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 level of concern values selected for this risk assessment. Diquat'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 diquat 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 diquat and presents the TRVs selected for this risk assessment (Table 3-1). Appendix A presents a summary of the diquat 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., diquat); however, some data correspond to a specific product or applied mixture (e.g., Reward®) containing the a.i. under consideration, and potentially other ingredients (e.g., other active ingredients 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

Test materials for diquat dibromide (technical product) frequently contain 40% or less of diquat cation, and this is the highest purity produced (USEPA 1995a). Accordingly, most TRVs were based on products containing less than 100% a.i.. However, this toxicity data represents the potential impacts due to the types of formulations that BLM would likely use. End-use products (commercial herbicide products) contain only 2 percent more water than technical diquat. Consequently, the toxicity of diquat does not appreciably differ among various formulations (USEPA 1995a). Diquat is moderately toxic to mammals, particularly via dermal exposure. Diquat is also moderately toxic to birds and

honeybees. In addition, adverse effects to non-target terrestrial plant species occurred with exposure to low concentrations of diquat (0.0046 lbs a.i./ac – 0.5% of the typical application rate).

According to USEPA ecotoxicity classifications presented in registration materials[1], diquat has relatively low toxicity to fish and moderate toxicity to aquatic invertebrates. Diquat does not appear to appreciably bioconcentrate in fish tissue. No acute toxicity studies conducted on amphibian species were found in the literature reviewed. Aquatic macrophytes were adversely affected by diquat concentrations as low as 0.00075 mg/L. There did not appear to be appreciable differences in sensitivities among aquatic macrophytes, diatoms, and algae.

### 3.1.2 Toxicity to Terrestrial Organisms

Although diquat is an aquatic herbicide, it is important to discuss the herbicide's potential impacts on terrestrial organisms, as terrestrial exposure to diquat could occur as a result of off-site drift or an accidental application.

#### 3.1.2.1 Mammals

According to USEPA reregistration eligibility documents (USEPA 1995a), diquat is considered to be moderately toxic to mammals. Acute dermal exposure to concentrations of 262 mg a.i./kilogram (kg) BW caused 50 percent mortality in rabbits (*Leporidae* sp.; the $LD_{50}$) (USEPA 1995a, Master Record Identification Number [MRID] 001000614). In subchronic dermal exposures lasting 21 to 28 days, the LOAEL was 5 mg a.i./kg BW for rats (*Rattus norvegicus* spp.; USEPA 2002, MRID 40308101). Diquat also caused mortality when administered via oral dosing or the diet. In studies where rats were given a single oral gavage, the reported $LD_{50}$ value was 121 mg/kg BW using technical grade diquat (no % a.i. listed; Gaines and Linder 1986). In a 2-year feeding study, rats exhibited adverse effects when fed dietary concentrations of 75 parts per million (ppm; equivalent to 2.9 mg a.i./kg BW-day), while no adverse effects were observed in rats fed 15 ppm (equivalent to 0.58 mg a.i./kg BW-day) (USEPA 2002, MRID 00145855 and MRID 41085601). No adverse effects to reproduction were reported in a chronic feeding study where rodents were exposed to 16 ppm diquat (0.8 mg a.i./kg BW-day) (USEPA 2002, MRID 41531301).

*Based on these findings, the oral $LD_{50}$ (121 mg/kg BW) and chronic dietary NOAEL (0.8 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at 262 mg a.i./kg BW.*

Toxicity data for large mammals was limited to a single long-term dietary study. In this one-year feeding study, beagle dogs (*Canis familiaris*) exhibited adverse effects (changes in body and epididymides weight, cataracts in the eye) when fed dietary concentrations of 105 ppm (equivalent to 2.5 mg a.i./kg BW-day). No adverse effects occurred at 21 ppm (equivalent to 0.5 mg a.i./kg BW-day) (USEPA 2002, MRID 41730301).

*Since no large mammal $LD_{50}$s were identified in the available literature, the small mammal dermal $LD_{50}$ was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 0.5 mg a.i./kg BW-day.*

#### 3.1.2.2 Birds

Data from available literature indicate that diquat has moderate toxicity to birds. In acute oral exposures, 50 percent of mallards (*Anas platyrhynchos*) died when exposed to a single oral dose of 60.6 mg/kg BW of a 45% diquat product (USEPA 2003, MRID 00106559). Acute dietary tests reported 50 percent mortality in ring-necked pheasant (*Coturnix coturnix*) and Japanese quail (*Coturnix coturnix*) fed concentrations as low as 734 and 264 ppm (equivalent to 43 and 30 mg a.i./kg BW-day, respectively) (USEPA 1995a, MRID 00034769 and MRID 00116565). 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. This 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)

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

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 150 mg a.i./kg BW and 215 mg a.i./kg BW for the Japanese quail and pheasant, respectively.

Long-term dietary exposure to diquat also elicited adverse effects in birds. Mallards exposed to diquat for an entire generation showed adverse effects when fed 25 ppm, equivalent to a dose of 2.5 mg a.i./kg BW-day (USEPA 2003, MRID 00114230). No adverse effects were noted at dietary concentrations of 5 ppm, equivalent to 0.6 mg a.i./kg BW-day. In bobwhite quail (*Colinus virginianus*), chronic dietary exposure failed to cause adverse effects at concentrations of 20 ppm, equivalent to a dose level of 12 mg a.i./kg BW-day (USEPA 2003, MRID 00119988).

*Based on these findings, the Japanese quail dietary $LD_{50}$ (150 mg a.i./kg BW) and the bobwhite quail chronic NOAEL (12 mg a.i./kg BW-day) were selected as the small bird dietary TRVs. The pheasant dietary $LD_{50}$ (215 mg a.i./kg BW) and the mallard dietary chronic NOAEL (0.6 mg a.i./kg BW-day) were selected as the large bird dietary TRVs.*

#### 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, technical diquat was directly applied to the bee's thorax and mortality was assessed during a 48-hr period. The USEPA reports a $LD_{50}$ value of 100 micrograms (μg)/bee using a 99.6% diquat product (USEPA 2003, MRID 00072012). A similar study conducted for 5 days reported an $LD_{50}$ of 47 μg/bee and a no effect level of 16 μg/bee using technical grade diquat (no % a.i. listed) (USEPA 2003, MRID 40208001).

*The honeybee dermal TRV was set at 47 μg/bee, the 5 day $LD_{50}$ value. Based on a honeybee weight of 0.093 g, this TRV was expressed as 505 mg/kg BW.*

#### 3.1.2.4 Terrestrial Plants

Toxicity tests were conducted on numerous, non-target plant species (plants tested were vegetable crop species and not western rangeland or forest species). Endpoints in the terrestrial plant toxicity tests were generally related to seed germination, seed emergence, and sub-lethal (i.e. growth) impacts observed during vegetative vigor assays. One study evaluated seed germination and emergence, and the remaining studies assessed vegetative vigor. Seed germination and emergence were evaluated by applying the herbicide to soil containing newly sown seed of 10 crop species. In this study, no effect to plants was observed at 7.49 lb a.i./ac (USEPA 1995a, MRID 40165101). However, a separate study indicated that vegetative vigor was a more sensitive endpoint. Vigor was adversely affected (25 percent of the plants affected; i.e., Effect Concentration [$EC_{25}$]) in plants exposed to concentrations as low as 0.0047 lb a.i./ac (USEPA 1995a, MRID 41883001).

*Two endpoints were used to evaluate terrestrial plant scenarios for aquatic herbicides. These included an $EC_{25}$ and a NOAEL. Since the lowest $EC_{25}$ identified in the database was lower than the lowest reported NOAEL, the terrestrial plant NOAEL was calculated by dividing the $EC_{25}$ value by an uncertainty factor of 3. 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). The resulting NOAEL TRV was 0.0016 lb a.i./ac derived from the $EC_{25}$ of 0.0047 lb a.i./ac.*

### 3.1.3 Toxicity to Aquatic Organisms

#### 3.1.3.1 Fish

The toxicity of diquat to freshwater fish was evaluated by testing both cold- and warmwater fish species. The acute toxic effects of diquat were evaluated for rainbow trout (*Oncorhynchus mykiss*), coho salmon (*Oncorhynchus kisutch*), and brown trout (*Salmo trutta*; coldwater fish species). These studies found 50% mortality occurred after 96 hours of exposure to concentrations (i.e., the $LC_{50}$) of 14.83 mg/L using a 19.8% diquat product (USEPA 2003, MRID 00138961). Acute toxicity tests were also conducted on 12 warmwater fish species. In these studies, the 96-hr $LC_{50}$ was found to be as low as 0.75 mg a.i./L (Paul et al. 1994).

In chronic studies, the LOAEL in coldwater fish was 0.5 mg a.i./L based on reduced swimming performance in rainbow trout (Dodson and Mayfield 1979). In this study, the NOAEL was less than 0.5 mg a.i./L, the lowest concentration tested. For warmwater fish, chronic studies found adverse effects in concentrations as low as 1.5 mg/L, while no effects were reported at 0.58 mg/L using a 41% diquat product (USEPA 2003, MRID 40380703). Results from coldwater and warmwater fish species suggest that diquat has relatively low toxicity to fish, but may be toxic to some species.

*The lower of the cold- and warmwater fish endpoints were selected as the TRVs for fish. Therefore the warmwater 96-hour $LC_{50}$ of 0.75 mg a.i./L was selected as the acute TRV. Since the NOAEL in the chronic study on rainbow trout was determined to be <0.5 mg a.i./L, the coldwater fish NOAEL was calculated by dividing this value by an uncertainty factor of 3. The resulting NOAEL TRV for coldwater fish species was 0.17 mg a.i./L . This was selected as the chronic fish TRV.*

Bioconcentration factors (BCFs) measured in fish tissue ranged from <0.6 to 1.4, indicating that the bioconcentration potential for diquat is low (HSDB 2003).

#### 3.1.3.2 Amphibians

Toxicity tests were conducted on several amphibian species, including frogs, toads, and newts. In a 16-day exposure, northern leopard frogs *(Rana pipiens)* were adversely affected by diquat concentrations as low as 5 mg/L, while no adverse effects were observed at 2 mg/L (Dial and Bauer-Dial 1987). No information was provided regarding the % a.i. in these tests.

*The NOAEL (2 mg/L) was selected as an amphibian chronic TRV.*

#### 3.1.3.3 Aquatic Invertebrates

Freshwater invertebrate toxicity tests are required for the USEPA pesticide registration process. Numerous acute and chronic toxicity tests were found in the literature. For acute studies, the statistical endpoint (Median Lethal Concentration [$LC_{50}$] or Median Effective Concentration [$EC_{50}$]) is the concentration that kills or immobilizes 50 percent of the test organisms. The lowest $EC_{50}$ or $LC_{50}$ reported from these studies was 0.14 mg/L of diquat tested using *Hyallela* (water scud) (USEPA 2003, MRID 00115862; no % a.i. listed). No observed adverse effect concentrations for aquatic invertebrates ranged from 0.044 to >2 mg/L (USEPA 1995a, MRID 00115862; Moss 1978).

*The $LC_{50}$ (0.14 mg/L) was selected as the invertebrate acute TRV and the NOAEL of 0.044 mg/L was selected as the chronic TRV.*

#### 3.1.3.4 Aquatic Plants

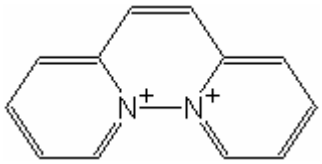
Standard toxicity tests were conducted on aquatic plants, including aquatic macrophytes, freshwater diatoms, and algae. In 14-day studies, 50 percent of the duckweed (*Lemna* sp.) plants were adversely affected by concentrations as low as 0.00075 mg/L of diquat (i.e., the $EC_{50}$) using a 35.3% diquat product (USEPA 2003, MRID 41883002).

Based on the available literature, there did not appear to be appreciable differences in sensitivities among aquatic macrophytes, diatoms, and algae.

*The $EC_{50}$ (0.00075 mg/L) was selected as the aquatic plant acute TRV. Since no NOAEL value in the reviewed literature was lower than the $EC_{50}$, the $EC_{50}$ was divided by an uncertainty factor of 3 to estimate a NOAEL TRV of 0.0003 mg./L.*

## 3.2 Herbicide Physical-Chemical Properties

The chemical formula for diquat is 6,7-dihydrodipyrido(1,2-a:2',1'-c)pyrazinedi-ium. The chemical structure is shown below:




**Diquat Chemical Structure**

The diquat molecule exists as a +2 cation when dissolved in water.

The physical-chemical properties and degradation rates critical to diquat'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 diquat 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.
- 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.
- 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.
- Hazardous Substances Data Bank (HSDB). 2003. 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. Springer-Verlag, New York.
- Howard, P (ed.). 1991. Handbook of Physical Properties of Organic Chemicals. CRC Lewis Publishers. Boca Raton. Florida.
- 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 obtained from the product label for the herbicide Reward® (Syngenta 2001) and from the EXTOXNET website (EXTOXNET 1996). Relevant papers from the scientific literature were obtained as part of the

literature review to define ecological toxicity endpoints. 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 diquat in aquatic systems. Values for foliar half-life and foliar washoff fraction were obtained from a database included in the GLEAMS computer model (U.S. 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

Diquat's terrestrial fate is dominated by sorption. When sorbed, diquat is immobile and protected from degradation (USEPA 1995a). 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. Diquat sorbs strongly to soils with reported $K_{oc}$ values ranging from 690 to 7,900,000. Based on this range of $K_{oc}$ values, diquat's mobility in soils will range from low to immobile (Table 3-2; Swann et al. 1983). Diquat exists in soil solution as a cation (Howard 1991). Thus, diquat's sorption mechanism is most accurately viewed as a cation exchange process (Howard 1991). For this reason, clays as well as organic matter can be important diquat sorbents in soil systems. When fixed in the interlayers of montmorrilonite clays, diquat sorption is irreversible (HSDB 2003). Diquat is also stable with respect to hydrolysis (USEPA 1995a). Based upon diquat's vapor pressure and its Henry's Law constant (the ratio of the chemical's distribution between the gas and liquid phases at equilibrium), volatilization from dry or wet soil surfaces should not be an important loss pathway (HSDB 2003). There is some evidence that the more loosely bound diquat fraction in soil may be subject to slow biodegradation (Howard 1991). Field half-lives for diquat ranging from 2.7 years to greater than 3 years have been reported (Table 3-2; HSDB 2003).

In aquatic systems, diquat sorbs to suspended solids, sediments, and also to aquatic vegetation (Simsiman and Chesters 1976; HSDB 2003). Loss of diquat from aquatic systems both through photolysis and biodegradation is possible, but only when the herbicide is not sorbed (HSDB 2003). When sorbed, it is protected from biodegradation and photolysis (Howard 1991). Based upon diquat's Henry's Law constant, volatilization from aquatic systems is not an important loss pathway (HSDB 2003). Although Petit et al. (1995) reports BCFs as high as 62, all other sources reviewed indicate that diquat has little tendency to bioaccumulate in aquatic species (Howard 1991; USEPA 1995a; Mackay et al. 1997; HSDB 2003). Diquat is removed quickly from the water column by sorption, with reported aquatic half-lives of 1-2 days (Table 3-2; HSDB 2003).

**TABLE 3-1**
**Selected Toxicity Reference Values for Diquat**

| Receptor | Selected TRV | Units | Duration | Endpoint | Species | Notes |
|---|---|---|---|---|---|---|
| **RECEPTORS INCLUDED IN FOOD WEB MODEL** | | | | | | |
| **Terrestrial Animals** | | | | | | |
| Honeybee | 47 | µg/bee | 5 d | $LD_{50}$ | | extrapolated from NOAEL; no % a.i. listed |
| Large Bird | 215 | mg a.i./kg bw | NR | $LD_{50}$ | ring neck pheasant | |
| Large Bird | 0.6 | mg a.i./kg bw-day | 1 generation | NOAEL | mallard | |
| Piscivorous Bird | 0.6 | mg a.i./kg bw-day | 1 generation | NOAEL | mallard | |
| Small Bird | 150 | mg a.i./kg bw | NR | $LD_{50}$ | Japanese quail | |
| Small Bird | > 12 | mg a.i./kg bw-day | NR | NOAEL | bobwhite quail | |
| Small Mammal | 0.8 | mg a.i./kg bw-day | 104 w | NOAEL | rat | |
| Small Mammal - dermal | 262 | mg a.i./kg bw | NR | $LD_{50}$ | rabbit | |
| Small Mammal - ingestion | 121 | mgkg bw | > 14 d | $LD_{50}$ | rat | water exposure; no diet available; no % a.i. listed |
| Large Mammal | 121 | mg/kg bw | > 14 d | $LD_{50}$ | rat | small mammal value; no % a.i. listed |
| Large Mammal | 0.5 | mg a.i./kg bw-day | 1 y | NOAEL | dog | |
| **Terrestrial Plants** | | | | | | |
| Typical Species-direct spray, drift | 0.0047 | lb a.i./ac | NR | $EC_{25}$ | cotton | vigor |
| RTE Species-direct spray, drift | 0.0016 | lb a.i./ac | NR | NOAEL | cotton | vigor; extrapolated from $EC_{25}$ |
| **Aquatic Species** | | | | | | |
| Aquatic Invertebrates | 0.14 | mg/L | 48 h | $EC_{50}$ | amphipod | no % a.i. listed |
| Fish | 0.75 | mg a.i./L | 96 h | $LC_{50}$ | walleye | |
| Aquatic Plants & Algae | 0.00075 | mg a.i./L | 14 d | $EC_{50}$ | giant duckweed | 35.3% a.i. product |
| Aquatic Invertebrates | 0.044 | mg a.i./L | life cycle | NOAEL | water flea | 41.4% a.i. product |
| Fish | 0.17 | mg a.i./L | NR | NOAEL | rainbow trout | extrapolated from LOAEL /swimming speed |
| Aquatic Plants & Algae | 0.0003 | mg a.i./L | 14 d | NOAEL | giant duckweed | extrapolated from $EC_{50}$ |

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

| Receptor | Selected TRV | Units | Duration | Endpoint | Species | Notes |
|---|---|---|---|---|---|---|
| **ADDITIONAL ENDPOINTS** | | | | | | |
| Amphibian | no data | | | | | |
| Amphibian | 2 | mg/L | 16 d | NOAEL | northern leopard frog | no % a.i. listed |
| Warmwater Fish | 0.75 | mg a.i./L | 96 h | $LC_{50}$ | walleye | |
| Warmwater Fish | 0.58 | mg a.i./L | 34 d | NOAEL | fathead minnow | 41% a.i. product |
| Coldwater Fish | 14.83 | mg/L | 96 h | $LC_{50}$ | rainbow trout | 19.8% a.i. product |
| Coldwater Fish | 0.17 | mg a.i./L | NR | NOAEL | rainbow trout | extrapolated from LOAEL /swimming speed |

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 or drift on typical species.

$EC_{05}$ or NOAEL - to address direct spray or drift 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 $EC_{50}$).

NOAEL - to address chronic exposure.

Value for fish is the lower of the warmwater and coldwater values.

Units represent those presented in the reviewed study

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

**TABLE 3-2**
**Physical-Chemical Properties of Diquat.**

| Parameter | Value |
|---|---|
| Herbicide family | quaternary ammonium herbicide (Compendium of Pesticide Common Names 2003). |
| Mode of action | Interferes with cell respiration (EXTOXNET 1996). |
| Chemical Abstract Service number | 85-00-7 (dibromide), 2764-72-9 (cation) (Tomlin 1994; Compendium of Pesticide Common Names 2003). |
| Office of Pesticide Programs chemical code | 032201 (USEPA 1995a). |
| Chemical name | 6,7-dihydrodipyrido(1,2-a:2,1-c)pyrazinedi-ium dibromide (dibromide salt) (USEPA 1995a). |
| Empirical formula | $C_{12}H_{12}Br_2N_2$ (dibromide salt), $C_{12}H_{12}N_2$ (cation) (USEPA 1995a). |
| Molecular weight (MW) | 344.1 (dibromide salt), **184.2** (cation) (Tomlin 1994). |
| Appearance, ambient | Pure diquat, odorless yellow solid (USEPA 1995a). |
| Acid / Base properties | Not available. |
| Vapor pressure (millimeters of mercury [mmHg] at 25ºC) | $< 4 \times 10^{-9}$ (USEPA 1995a); 0 (Hornsby et al. 1996; Mackay et al. 1997); $<9.7 \times 10^{-8}$ (Tomlin 1994; Montgomery 1997). |
| Water solubility (mg/L at 25ºC) | 700,000 (20ºC) (Howard 1991; USEPA 1995a; Montgomery 1997); 570 (Reinert 1989); 718,000 (20ºC, pH 7.2) (Hornsby et al. 1996; Mackay et al. 1997). |
| Log octanol-water partition coefficient (Log ($K_{OW}$)), unitless | 2.78 (Reinert 1989); **-4.6** (Tomlin 1994; Montgomery 1997); -3.05 (Howard 1991). |
| Henry's Law constant (atm-$m^3$/mole) | 0 (Mackay et al. 1997). |
| Soil / Organic matter sorption coefficients (Kd / $K_{oc}$) | 708 - 2863 (Kd) (Petit et al. 1995); 32,000 - 7,900,000 ($K_{oc}$) (Ritter et al. 2000); 690 ($K_{oc}$) (Reinert 1989); Freundlich Kd of 15 in sandy sediment, 36-42 in two sandy soils, and 1882-10740 in sandy loam, sandy clay loam, and loam soils (USEPA 1995a). |
| Bioconcentration factor (BCF) | <1 – 62 (Petit et al. 1995); **1.03X** for whole fish (bluegill sunfish) exposed to approximately 1030 ppb diquat ion for 14 days (lab study). Depuration from bluegill sunfish was rapid with 50% accumulated $^{14}C$ residues eliminated from fish after three days. In a field study, diquat did not significantly accumulate in tissues of tilapia and catfish from two Florida fish ponds treated with the equivalent of 0.36 ug diquat ion/ml for four monthly applications (USEPA 1995a). |
| Field or soil dissipation half-life | No degradation observed after three years in field studies in New York and Idaho (clay loam and loam soils respectively). No evidence of significant leaching during study. (USEPA 1995a); 1,000 days (Hornsby et al. 1996) |
| Aquatic dissipation half-life | A disappearance time of 30-33 hours was estimated for diquat added to three ponds in North Carolina at 1 ppm (Langeland and Warner 1986); Diquat applied to three test plots in eutrophic FL lake at 4.0 lb a.i. cation/acre with and without polymer. After 25-39 hours, at 61 m from edge of application area only 3 of 53 samples contained between 4 and 10% of application level. Drop in diquat concentration was greater than expected due to dilution (Langeland et al. 1994). Laboratory microcosms were used to study diquat fate in weed and sediment and sediment alone water systems. In systems with weeds, 32% was degraded to water soluble products and 19% sorbed to sediment after 22 days. In systems with only sediment and water, little degradation occurred during the 180 day experiment (Simsiman and Chesters 1976). Dissipation half-lives of 1-2 days were observed for diquat added to Florida pond water (USEPA 1995a). |
| Hydrolysis half-life | No hydrolysis measured after 30 days at pH 5 or 7; <10% hydrolysis observed after 30 days at pH 9 (Ritter et al. 2000); Stable to hydrolysis at pH 5, 7, 9 (USEPA 1995a). |

**TABLE 3-2 (Cont.)**
**Physical-Chemical Properties of Diquat.**

| Parameter | Value |
|---|---|
| Photodegradation half-life in water | 8 days (half-life, distilled water, 240-260 nm light); < 35 days for 4 µg/ml to degrade in distilled water exposed to sunlight (Mackay et al. 1997). Photodegrades in surface layer of water in 1-3 wks if not adsorbed to particles (Howard 1991). Photolysis: 48 hr to 11 days (Petit et al. 1995); 74 days at pH 7, 25ºC, sterile conditions (Tomlin 1994). Diquat can be considered to be photolytically stable in the environment. Calculated half-life of 74 days for diquat in water with Florida spring sunlight (USEPA 1995a). |
| Photodegradation half-life in soil | Radiolabeled diquat did not degrade on soils irradiated with a xenon arc lamp for 107.42 hours (equivalent to approximately 32 days of natural sunlight) (USEPA 1995a); 50-75% dry diquat decomposed in 48 - 96 hours in sunlight (Howard 1991). |
| Soil biodegradation half-life | Diquat at approximately 3 µg/g did not degrade in aerobic sandy loam incubated at 25ºC for 9 months (USEPA 1995a). In four fungi, tolerance to diquat was linked to their ability to degrade the herbicide. *Aspergillus niger* released radiolabeled carbon dioxide from $C^{14}$ diquat while *Penicillium frequentans*, *Mucor hiemalis*, and *Zygorhynchus heterogamous* did not. The later three fungi could not grow at low herbicide concentrations (Smith et al. 1976). |
| Residue Rate for grass (1) | **197 ppm** (maximum) and **36 ppm** (typical) per lb a.i./ac |
| Residue Rate for vegetation (2) | **296 ppm** (maximum) and **35 ppm** (typical) |
| Residue Rate for insects (3) | **350 ppm** (maximum) and **45 ppm** (typical) |
| Residue Rate for berries (4) | **40.7 ppm** (maximum) and **5.4 ppm** (typical) |
| Aquatic biodegradation half-life | 50 days (half-life, lake water), > 158 days and 2 days (half-lives in sediment and water, respectively, sediment-water microcosms) (Mackay et al. 1997); <15 to 32 days (Petit et al. 1995). Diquat did not degrade when incubated under anaerobic aquatic conditions for nine months at 25ºC; Diquat did not degrade under aerobic aquatic conditions for 31 days at 25ºC. (USEPA 1995a) |

Notes:
Values presented in bold were used in risk assessment calculations.
[1] Residue rates selected are the high and mean values for long grass (Fletcher et al. 1994).
[2] Residue rates selected are the high and mean values for leaves and leafy crops (Fletcher et al. 1994).
[3] Residue rates selected are the high and mean values for forage such as legumes (Fletcher et al. 1994).
[4] 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 diquat. The general approach and analytical methods for conducting the diquat 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 diquat 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 diquat 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 ERA 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, diquat is proposed for use by the BLM for vegetation control in their Aquatics program. The proposed BLM program involves the general use and application of herbicides on public lands. 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. Diquat 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 ac pond of 1 meter [m] depth, resulting in a volume of 1,011,715 liters [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 m per second (m/sec), 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 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; SDTF 2002).

### 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 diquat. The selection process is discussed in detail in Methods Document (ENSR 2004c), and the selected endpoints are presented below.

<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.

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).

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 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. With the exception of non-target plants, standard acute and chronic toxicity test endpoints were used for estimates of direct herbicide effects on RTE species. To add conservatism to the RTE assessment, levels of concern for RTE species were lower than for typical species. Lowest available germination NOAELs were used to evaluate non-target RTE plants. Impacts to RTE species are discussed in more detail in Section 6.0.

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 diquat 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 (Available at: http://www.epa.gov/oppfead1/endanger/effects.)

### 4.1.6 Development of the Conceptual Model

The diquat conceptual model (Figure 4-1) is presented as a series of working hypotheses about how diquat might pose hazards to the ecosystem and ecological receptors. The conceptual model indicates the possible exposure pathways for the herbicide, as well as the receptors 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 describes the source, fate, and distribution of the herbicide using standard models that predict concentrations in various environmental media (e.g., AgDRIFT®). All EECs predicted by the models 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., application by 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 diquat: 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 Body Weights (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]).

In addition, potential impacts to non-target terrestrial plants were considered by evaluating two plant receptors: the "typical" non-target species, and the RTE non-target species. Cotton was the surrogate species chosen to represent terrestrial plants (toxicity data are only available for vegetable crop species). Since diquat is an aquatic herbicide, it is not expected that terrestrial plants will be exposed to the herbicide. It is possible that the noncropland species present on BLM managed lands may be more tolerant of diquat than the surrogate species used in the ERA.

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 (*Sander vitreus*) were surrogates for fish, the water flea (*Daphnia magna*) and waterscud (*Hyalella* spp.) were surrogates for aquatic invertebrates, and non-target aquatic plants and algae were represented by giant duckweed.

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

#### 4.2.1.1 Direct Spray

Wildlife and non-target plant 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 food 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 food 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 Food Items Contaminated by Accidental Direct Spray
- Accidental Direct Spray Over Stream (diquat 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 diquat, ground applications were modeled using a low- or high-placed boom and aerial application was modeled from both a helicopter and a fixed-wing 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 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, and this allowed evaluation of impacts to off-site terrestrial plants.

#### 4.2.1.3 Accidental Spill to Pond

To represent worst-case potential impacts to ponds, 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 a 1/4 ac, 1 m 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 diquat.

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 Levels of Concern (LOC) 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 (e.g., rainfall, slope) and receptor assumptions (e.g., 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 and 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, spray drift, and wind erosion scenarios, the selected toxicity endpoints were an $EC_{25}$ for "typical" species and a NOAEL for RTE species. In runoff scenarios, high and low germination NOAELs were selected to evaluate exposure for typical and RTE species, respectively.

The evaluation of RTE terrestrial 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 4-5 and presented graphically in Figures 4-3 to 4-18. 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 4-18). 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 food items, accidental direct spray over stream). Table 4-2 presents the RQs for the above scenarios. Figures 4-3 to 4-7 present graphic representations of the range of RQs and associated LOCs.

### 4.3.1.1 Terrestrial Wildlife

Acute RQs for terrestrial wildlife (Figure 4-3) were above the most conservative LOC of 0.1 (acute endangered species) for several scenarios. Accidental direct spray of the pollinating insect resulted in elevated RQs at both the typical and maximum application rates. Risk was also predicted for the pollinating insect as a result of indirect contact with foliage accidentally sprayed at the maximum application rate. This is a highly conservative scenario assuming that the insect absorbs 100% of the herbicide after application with no degradation or limitations to uptake. This scenario may overestimate risk to the insect. No risks to the small mammal were predicted due to direct spray or indirect contact with foliage.

Acute exposure RQs were elevated above the associated LOC (0.1; acute risk to endangered species) for two scenarios using the typical application rate (large mammalian herbivore and small avian insectivore) and for five

scenarios at the maximum application rate (large and small mammalian herbivore, large avian herbivore, large mammalian carnivore, small avian insectivore).

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

This evaluation indicates that accidental direct spray impacts may pose a risk to insects, birds and wildlife, primarily when the maximum application rate is used. However, accidental spray of a terrestrial area during the application of an aquatic herbicide is an unlikely event.

#### 4.3.1.2 Non-target Plants – Terrestrial and Aquatic

As expected, because of the mode of action of herbicides, RQs for non-target terrestrial and aquatic plants impacted by direct spray were above the plant LOC of 1 for all modeled scenarios. RQs for direct spray of non-target terrestrial plants (Figure 4-3) ranged from 213 to 2500. RQs for non-target aquatic plants (Figure 4-4) impacted by normal application to the pond or accidental direct spray of the stream ranged from 149 to 7,472 (Table 4-2). Therefore, direct spray impacts pose a risk to plants in both aquatic and terrestrial environments. It may be noted that the aquatic scenarios are particularly conservative because they evaluate an instantaneous concentration and do not consider flow, adsorption to particles, or degradation that may occur over time within the pond or stream. In addition, direct spray of terrestrial plants is an unlikely scenario.

#### 4.3.1.3 Fish and Aquatic Invertebrates

Acute and chronic toxicity RQs for fish and aquatic invertebrates in the pond and stream (Figure 4-5 and 4-6) were above the most conservative associated LOCs (0.05 for acute endangered species; 0.5 for chronic risk endangered species).

These results indicate there is potential risk to aquatic species, especially endangered species, in a pond or stream sprayed with diquat. It may be noted that these spray scenarios are conservative because they are instantaneous concentrations and do not consider flow, adsorption to particles, or degradation that may occur over time. The herbicide concentration in the pond and stream are the instantaneous concentrations at the moment of the direct spray. The volume of the pond and the impacted segment of the stream were calculated and the mass of herbicide was calculated based on the surface area of the waterbody. There was no dilution due to degradation or stream flow. In addition, it is assumed that the pond and stream are adjacent to the herbicide application area.

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

### 4.3.2 Off-site Drift to 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 diquat 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 fixed-wing 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 area for aerial applications.

Table 4-3 presents the RQs for off-site drift to off-site soils and associated impacts to terrestrial plants. RQs for both typical and RTE terrestrial plant species (Figure 4-8) were elevated over the plant LOC of 1 for several scenarios. At

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

### 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 ac, 1 m 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 the helicopter, respectively, were mixed into the pond volume.

Risk quotients for the truck spill scenario (Table 4-2) were 19.1 for fish, 102 for aquatic invertebrates (Figure 4-6 and 4-7) and 19,129 for non-target aquatic plants (Figure 4-5). Risk quotients for the helicopter spill scenario were higher at 67 for fish, 359 for aquatic invertebrates, and 66,952 for non-target aquatic plants. Potential risks to fish, aquatic invertebrates, and non-target aquatic plants were indicated for the truck and helicopter spills mixed for the maximum application rate. However, these scenarios are highly conservative and represent unlikely and worst case conditions (limited waterbody volume, tank mixed for maximum application).

### 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 direct or indirect effects of diquat to salmonids and their habitat; therefore, only qualitative estimates of indirect effects are possible. These estimates were accomplished by evaluating predicted impacts to prey items and vegetative cover in the stream scenarios discussed above. Since terrestrial applications of aquatic herbicides are not planned, the primary scenario evaluated in this assessment was accidental direct spray over the stream (diquat is not generally intended 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 cold-water species identified during the literature search. Several laboratory studies with salmonids (rainbow trout, brown trout, coho salmon) were identified in the literature and considered in the selection of the fish TRVs (Appendix A). The chronic fish TRV was based on a reduction in swimming speed for the rainbow trout. The lowest acute toxicity value was for a warm-water fish species, the walleye, and resulted in an acute TRV of 0.75 mg a.i./L, based on a 96 hour $LC_{50}$. However, salmonid acute toxicity values ($LC_{50}$s) were much higher than the selected acute TRV, ranging from 19 mg a.i./L to >100 mg a.i./L. This suggests that acute, direct effects to salmonids may be overestimated in the risk assessment.

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 at both the typical and maximum application rates. 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 above the plant LOC at both the typical and maximum application rates, indicating the potential for a reduction in the aquatic plant community. However, this is a conservative scenario in which it is assumed that a stream is accidentally directly sprayed by a herbicide used for pond applications. This is unlikely to occur as a result of BLM practices and represents a worst-case scenario. In addition, no reduction in herbicide concentration is calculated due to stream flow in this scenario. Nonetheless, there would be the potential for indirect impacts to salmonids due to a reduction in available cover if the stream is accidentally sprayed.

Although not specifically evaluated in the stream scenarios of the ERA, terrestrial plants were evaluated for their potential to provide overhanging cover for salmonids. A reduction in riparian cover has the potential to indirectly impact salmonids within the stream. RQs for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community and potential indirect impacts to salmonids. However, as discussed above, this event is unlikely to occur as a result of BLM practices and represents a worst-case scenario.

#### 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) and a reduction in vegetative cover. However, this evaluation is based on worst-case accidental exposure scenarios, which are not likely to occur as a result of BLM practices. Decreasing the application rate and avoiding application on non-target areas would reduce the likelihood of these impacts.

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/Semi-Aquatic 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 $_{BW}$ for acute scenarios and mg $_{prey}$/kg $_{BW}$/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 | 2.48E-02 | 9.93E-02 |
| Pollinating insect - 100% absorption | **3.14E-01** | **1.26E+00** |
| Small mammal - 1st order dermal adsorption | 1.19E-04 | 4.76E-04 |
| **Indirect Contact With Foliage After Direct Spray** | | |
| Small mammal - 100% absorption | 2.48E-03 | 9.93E-03 |
| Pollinating insect - 100% absorption | 3.14E-02 | **1.26E-01** |
| Small mammal - 1st order dermal adsorption | 1.19E-05 | 4.76E-05 |
| **Ingestion of Food Items Contaminated by Direct Spray** | | |
| Small mammalian herbivore - acute exposure | 1.60E-02 | **4.82E-01** |
| Small mammalian herbivore - chronic exposure | **1.01E+00** | **3.05E+01** |
| Large mammalian herbivore - acute exposure | **1.03E-01** | **2.25E+00** |
| Large mammalian herbivore - chronic exposure | **4.20E+00** | **9.19E+01** |
| Small avian insectivore - acute exposure | **1.36E-01** | **4.23E+00** |
| Small avian insectivore - chronic exposure | 7.15E-01 | **2.23E+01** |
| Large avian herbivore - acute exposure | 3.99E-02 | **1.35E+00** |
| Large avian herbivore - chronic exposure | **6.02E+00** | **2.04E+02** |
| Large mammalian carnivore - acute exposure | 6.66E-02 | **2.67E-01** |
| Large mammalian carnivore - chronic exposure | 3.13E-02 | 1.25E-01 |

| Semi-Aquatic Wildlife | Typical Application Rate | Maximum Application Rate |
|---|---|---|
| **Ingestion of Prey Items Contaminated by Normal Application to Pond** | | |
| Avian piscivore – chronic exposure | 1.52E-02 | 6.08E-02 |

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

| Terrestrial Plants | Typical Species | | RTE Species | |
|---|---|---|---|---|
| | Typical Application Rate | Maximum Application Rate | Typical Application Rate | Maximum Application Rate |
| **Direct Spray of Terrestrial Plants** | | | | |
| Accidental direct spray | **2.13E+02** | **8.51E+02** | **6.25E+02** | **2.50E+03** |

| 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 | **1.49E-01** | **5.98E-01** | **8.01E-01** | **3.20E+00** | **1.49E+02** | **5.98E+02** |
| Chronic | **6.59E-01** | **2.64E+00** | **2.55E+01** | **1.02E+02** | **3.74E+02** | **1.49E+03** |
| **Direct Spray Over Stream – Accidental Spray** | | | | | | |
| Acute | **7.47E-01** | **2.99E+00** | **4.00E+00** | **1.60E+01** | **7.47E+02** | **2.99E+03** |
| Chronic | **3.30E+00** | **1.32E+01** | **1.27E+02** | **5.09E+02** | **1.87E+03** | **7.47E+03** |
| **Accidental spill** | | | | | | |
| Truck spill into pond | -- | **1.91E+01** | -- | **1.02E+02** | -- | **1.91E+04** |
| Helicopter spill into pond | -- | **6.70E+01** | -- | **3.59E+02** | -- | **6.70E+04** |

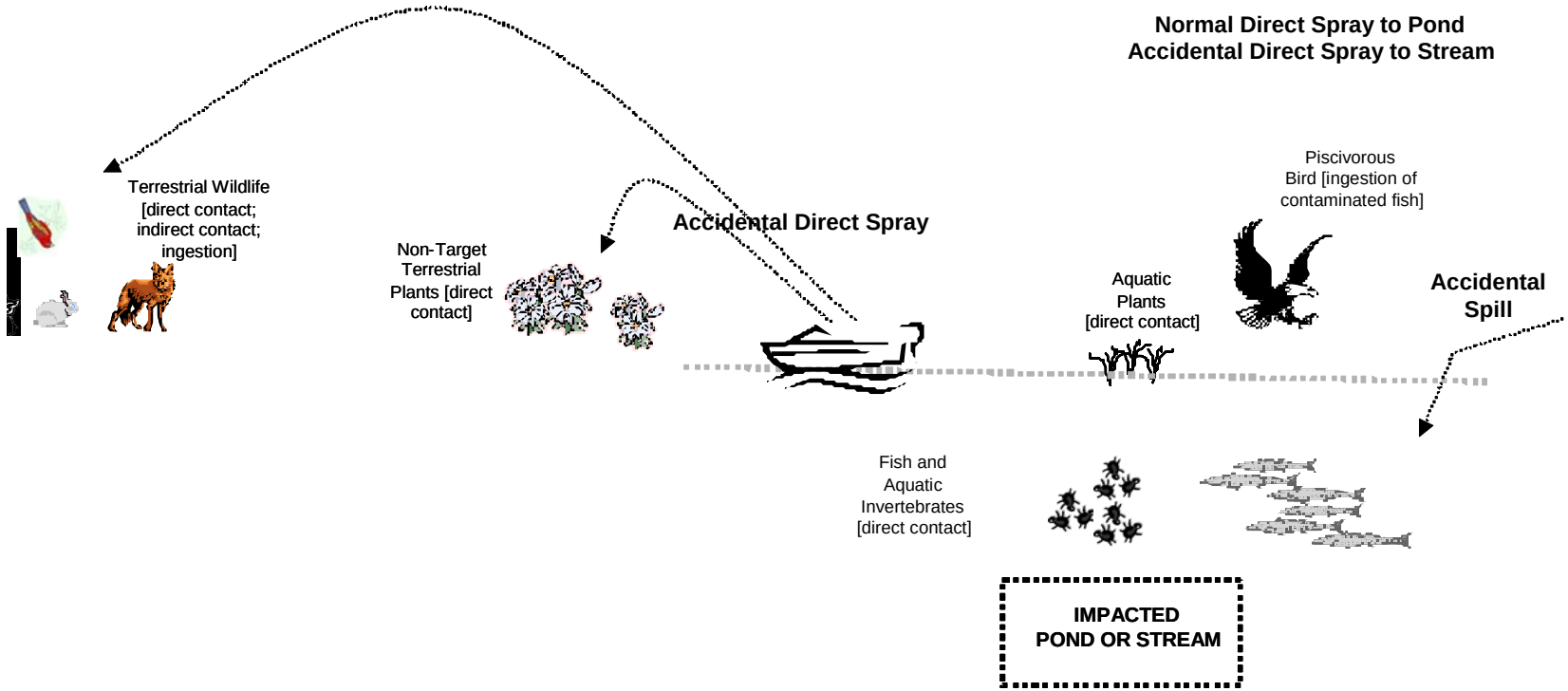
Shading and boldface indicates terrestrial animal acute RQs greater than 0.1 (LOC for acute risk to endangered species - most conservative).
Shading and boldface indicates terrestrial animal chronic RQs greater than 1 (LOC for chronic risk).
Shading and boldface indicates plant RQs greater than 1 (LOC 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**
**Risk Quotients for Off-Site Drift Scenarios**

| Potential Risk to Terrestrial Plants | | | | | | |
|---|---|---|---|---|---|---|
| | | | Typical Species | | Rare, Threatened, and Endangered Species | |
| **Mode of Application** | **Application Height or Type** | **Distance From Receptor (ft)** | **Typical Application Rate** | **Maximum Application Rate** | **Typical Application Rate** | **Maximum Application Rate** |
| **Spray Drift to Off-Site Soil** | | | | | | |
| Plane | Non-Forested | 100 | **2.09E+01** | **9.68E+01** | **6.13E+01** | **2.84E+02** |
| Plane | Non-Forested | 300 | **9.49E+00** | **4.37E+01** | **2.79E+01** | **1.28E+02** |
| Plane | Non-Forested | 900 | **3.80E+00** | **1.14E+01** | **1.12E+01** | **3.35E+01** |
| Helicopter | Non-Forested | 100 | **1.90E+01** | **8.35E+01** | **5.58E+01** | **2.45E+02** |
| Helicopter | Non-Forested | 300 | **7.59E+00** | **3.23E+01** | **2.23E+01** | **9.48E+01** |
| Helicopter | Non-Forested | 900 | **3.80E+00** | **5.69E+00** | **1.12E+01** | **1.67E+01** |
| Ground | Low Boom | 25 | **1.90E+00** | **1.14E+01** | **5.58E+00** | **3.35E+01** |
| Ground | Low Boom | 100 | 9.43E-01 | **3.80E+00** | **2.77E+00** | **1.12E+01** |
| Ground | Low Boom | 900 | 1.45E-01 | 5.81E-01 | 4.26E-01 | **1.71E+00** |
| Ground | High Boom | 25 | **3.80E+00** | **1.71E+01** | **1.12E+01** | **5.02E+01** |
| Ground | High Boom | 100 | **1.49E+00** | **5.69E+00** | **4.37E+00** | **1.67E+01** |
| Ground | High Boom | 900 | 1.86E-01 | 7.42E-01 | 5.45E-01 | **2.18E+00** |

Shading and boldface indicates plant RQs greater than 1 (LOC for all plant risks).
RTE – Rare, Threatened, and Endangered

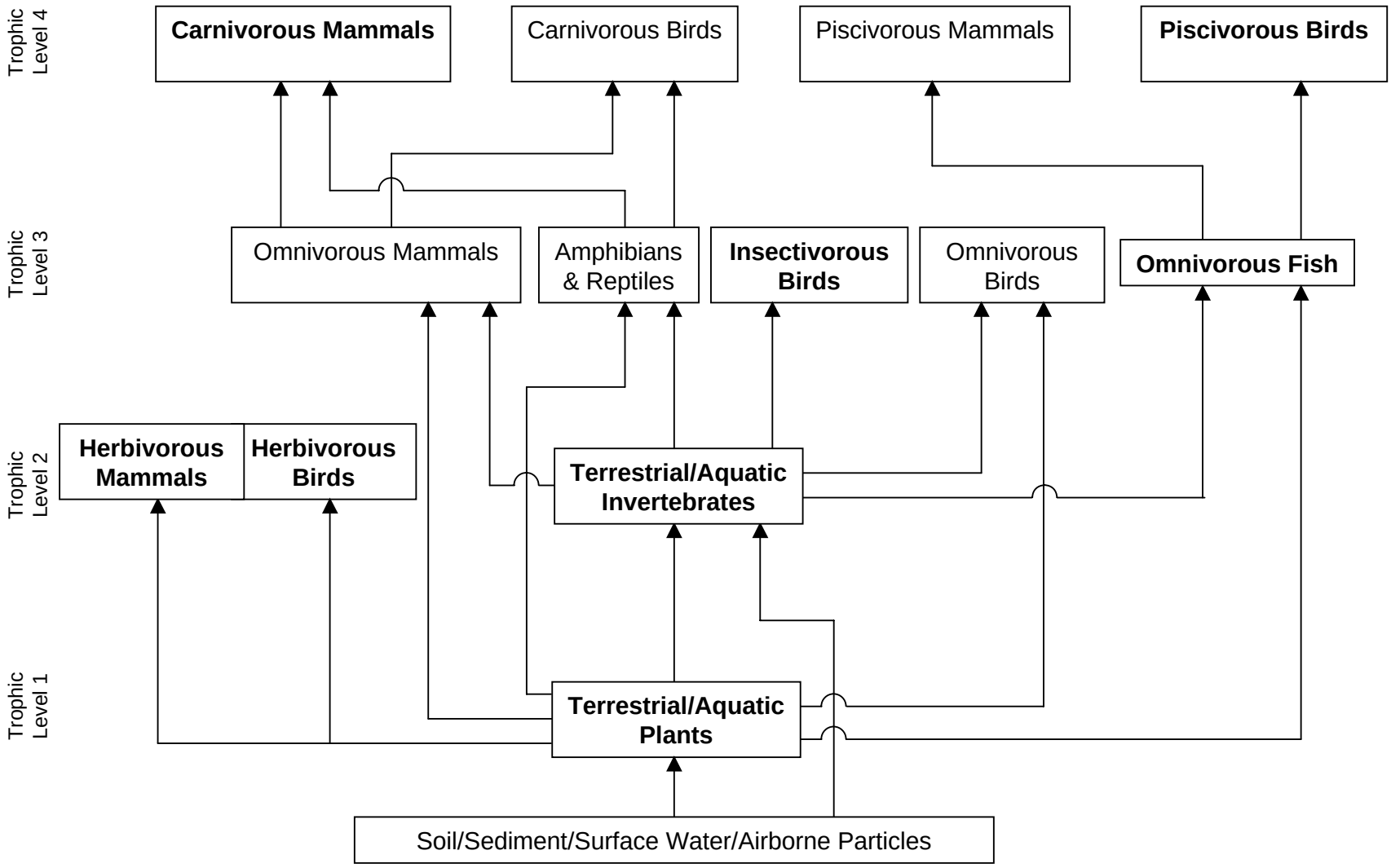
**FIGURE 4-1. Conceptual Model for Aquatic Herbicides.**




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

See Figure 4-2 for simplified food web & evaluated receptors.

**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.**

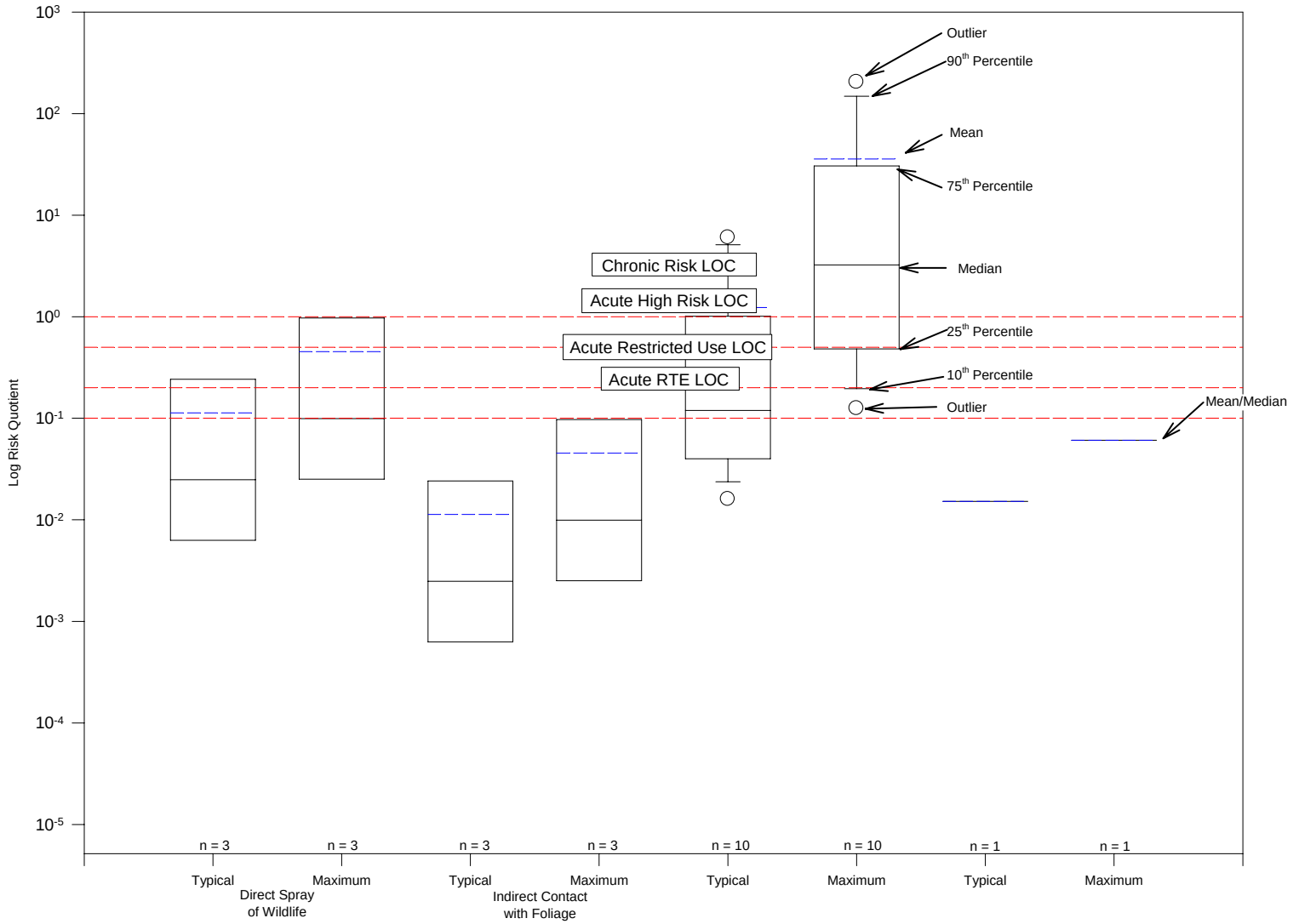


**FIGURE 4-4. Direct Spray - Risk Quotients for Non-Target Terrestrial Plants.**

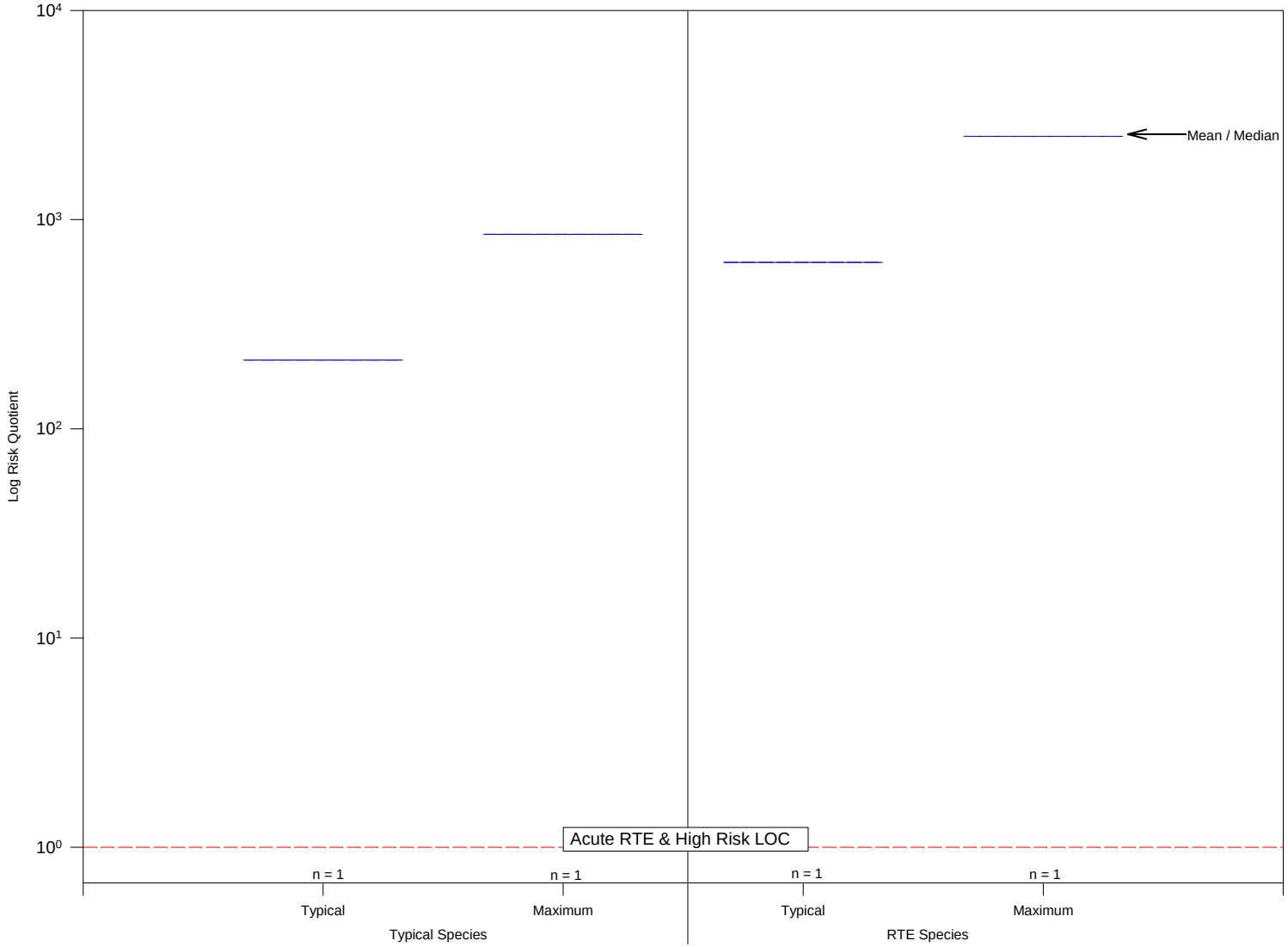


**FIGURE 4-5. Accidental Direct Spray and Spills - Risk Quotients for Non-Target Aquatic Plants.**

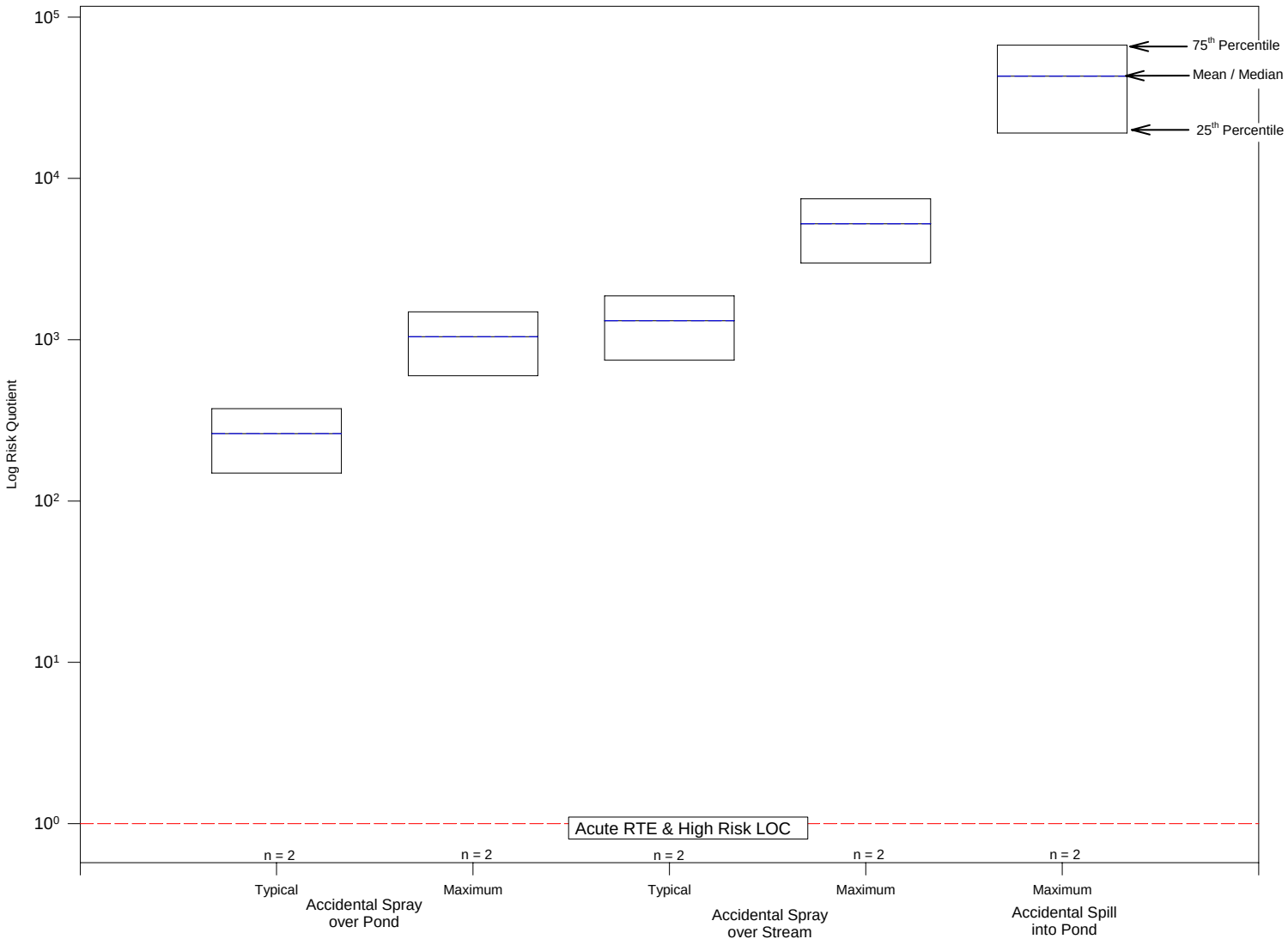


**FIGURE 4-6. Accidental Direct Spray and Spills - Risk Quotients for Fish.**

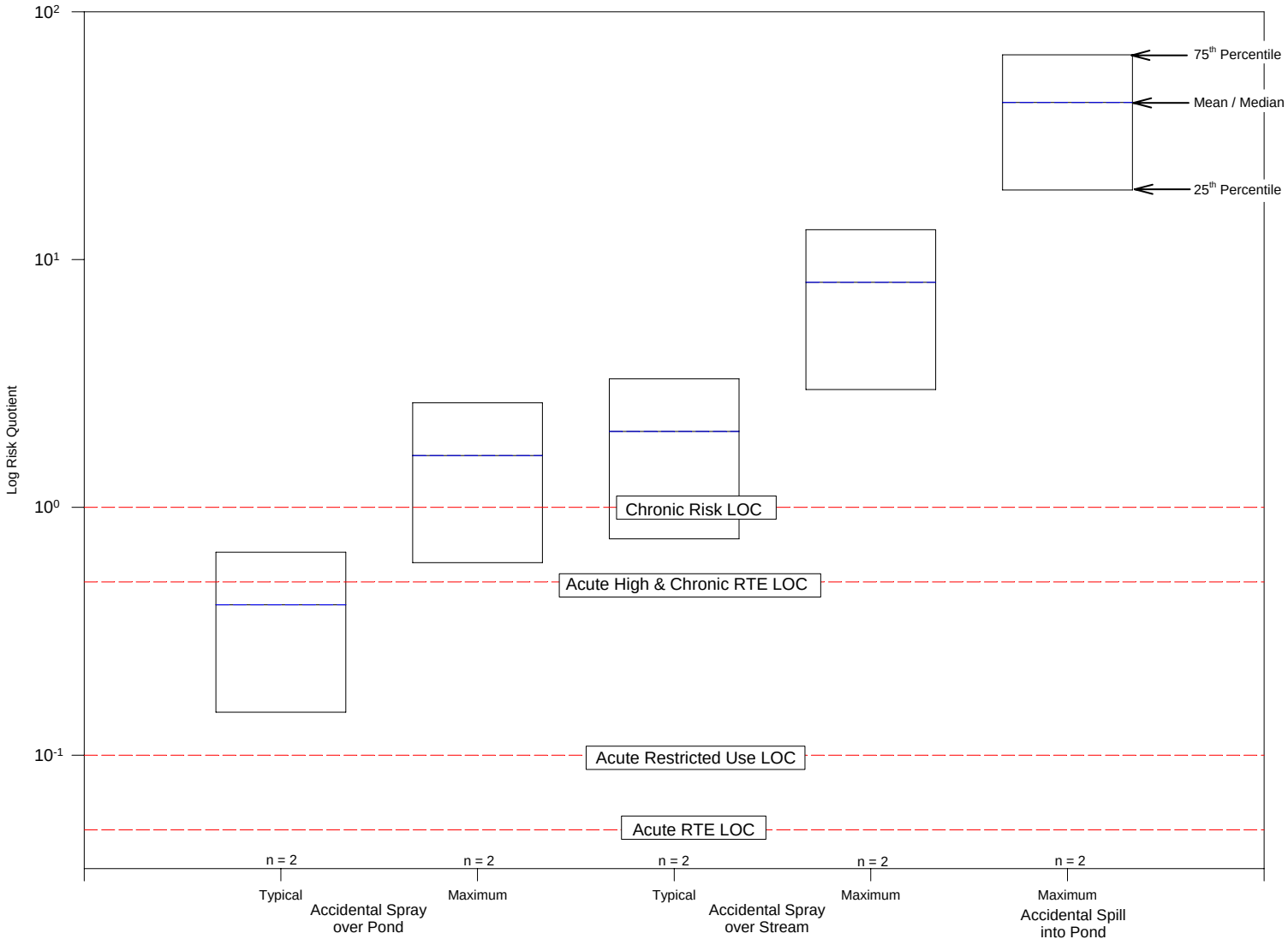


**FIGURE 4-7. Accidental Direct Spray and Spills - Risk Quotients for Aquatic Invertebrates.**

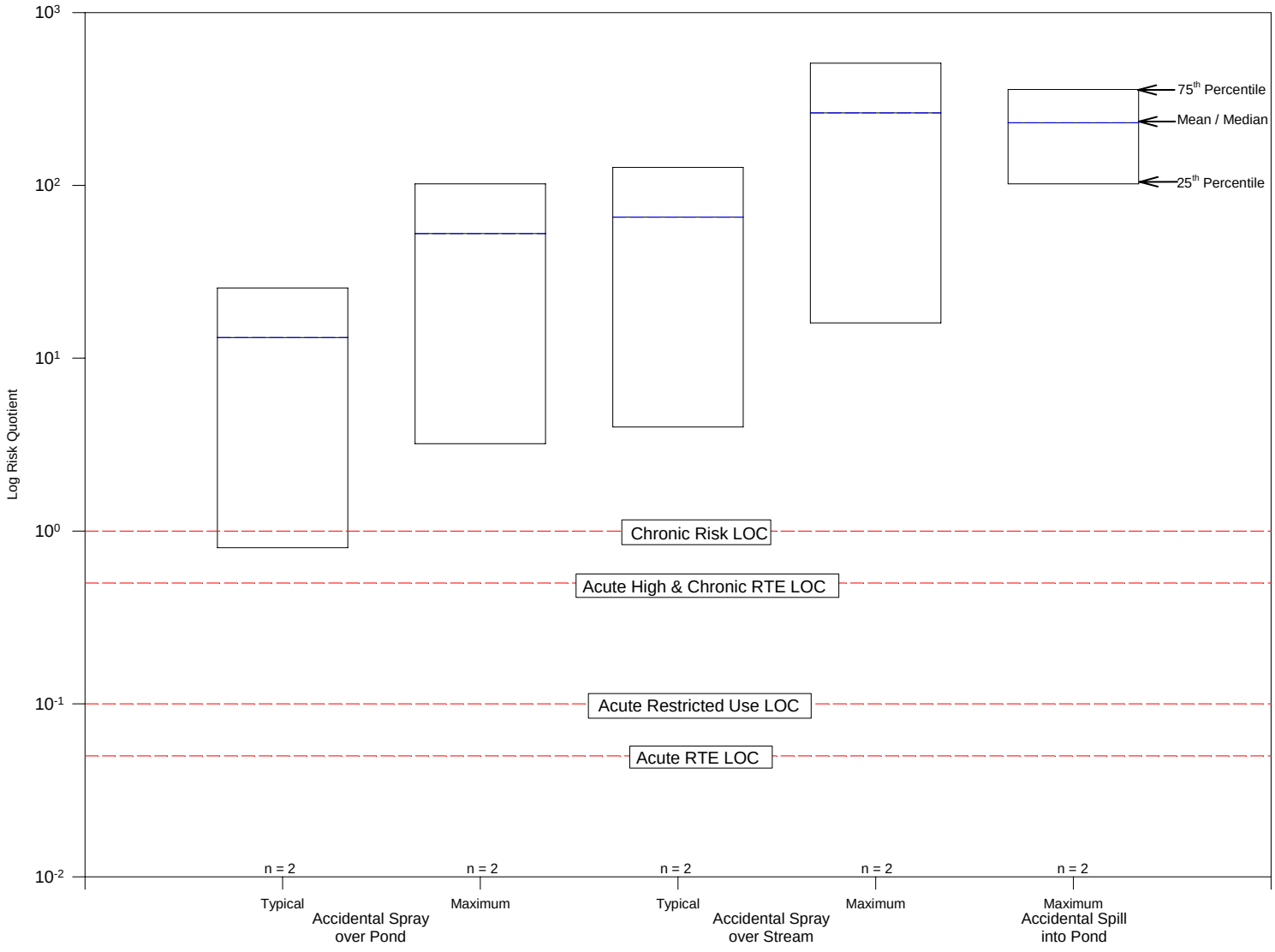
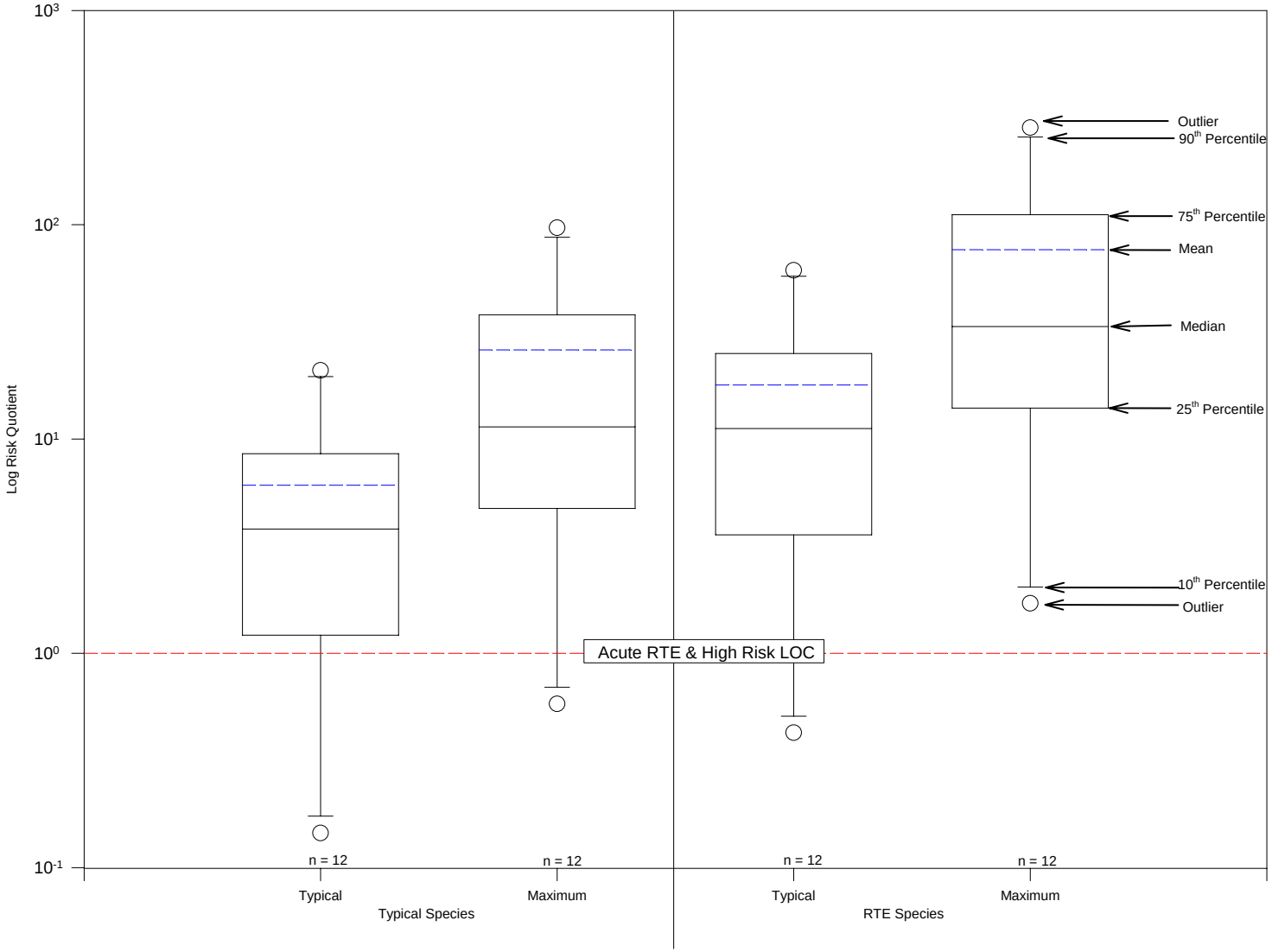


**FIGURE 4-8. Off-Site Drift - Risk Quotients for Non-Target Terrestrial Plants.**



# 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 ac pond 1 m 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 ac 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 greater than 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 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 less than 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. In general, the evaluation presented in Table 5-3 indicates that there is a decrease in herbicide migration and associated ecological risk, with increased downward distance (i.e., buffer zone) and an increase in herbicide migration with increasing application height and rate.

**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 | 113,398 | 0.11 | Base case |
| 100 | 1 | 404,685,642 | 45,359,237 | 0.11 | Increased pond area; No change in concentration |
| 1,000 | 1 | 4,046,856,422 | 453,592,370 | 0.11 | Increased pond area; No change in concentration |
| 0.25 | 0.5 | 2,023,428,211 | 453,592,370 | 0.22 | Decreased pond depth; Increased concentration |
| 0.25 | 2 | 2,023,428 | 113,398 | 0.056 | Increased pond depth; decreased concentration |
| 0.25 | 4 | 4,046,856 | 113,398 | 0.028 | 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 | 142,606 | 0.56 | Base case |
| 4 | 0.2 | 636 | 508,920 | 285,212 | 0.56 | Increased stream width; No change in concentration |
| 1 | 0.2 | 636 | 127,230 | 71,303 | 0.56 | Decreased stream width; No change in concentration |
| 2 | 0.4 | 636 | 508,920 | 142,606 | 0.28 | Increased stream depth; Decreased concentration |
| 2 | 0.1 | 636 | 127,230 | 142,606 | 1.12 | Decreased stream depth; Increased concentration |
| 2 | 0.2 | 201 | 80,468 | 45,096 | 0.56 | Increased stream length; No change in concentration |
| 2 | 0.2 | 2,012 | 804,672 | 450,959 | 0.56 | Decreased stream length; No change in concentration |

(1) – Length of impacted stream is based on size of application area. 10 ac application area = 201 m impacted; 100 ac application area = 636 m impacted; 1,000 ac application area = 2,012 m 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 | 8.08E-03 | 1.96E-03 |
| Helicopter | Forest | 100 | 900 | NA | NA |
| | Non-Forest | 100 | 900 | 6.82E-03 | 1.58E-03 |
| Ground | Low Boom | 25 | 900 | 6.82E-04 | 7.22E-05 |
| | High Boom | 25 | 900 | 1.09E-03 | 9.16E-05 |
| **Maximum Application Rate** | | | | | |
| Plane | Forest | 100 | 900 | NA | NA |
| | Non-Forest | 100 | 900 | 3.89E-02 | 5.85E-03 |
| Helicopter | Forest | 100 | 900 | NA | NA |
| | Non-Forest | 100 | 900 | 3.23E-02 | 2.55E-03 |
| Ground | Low Boom | 25 | 900 | 2.73E-03 | 2.89E-04 |
| | High Boom | 25 | 900 | 4.38E-03 | 3.66E-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 | 0.2426 | - |
| Helicopter | Forest | 100 | 900 | NA | NA |
| | Non-Forest | 100 | 900 | 0.2317 | - |
| Ground | Low Boom | 25 | 900 | 0.1059 | - |
| | High Boom | 25 | 900 | 0.0840 | - |
| **Maximum Application Rate** | | | | | |
| Plane | Forest | 100 | 900 | NA | NA |
| | Non-Forest | 100 | 900 | 0.1504 | - |
| Helicopter | Forest | 100 | 900 | NA | NA |
| | Non-Forest | 100 | 900 | 0.0789 | - |
| Ground | Low Boom | 25 | 900 | 0.1059 | - |
| | High Boom | 25 | 900 | 0.0836 | - |

**TABLE 5-3 (Cont.)**
**Herbicide Exposure Concentrations used during the Supplemental AgDRIFT® Sensitivity Analysis**

**Effect of Application Height (Vegetation Type or Boom Height)**

| Mode of Application | Application Height or Vegetation Type | Concentration Ratio[1] | Relative Change in Concentration |
|---|---|---|---|
| **Typical Application Rate** | | | |
| Plane | Forest/ Non-Forest | NA | NA |
| Helicopter | Forest/ Non-Forest | NA | NA |
| Ground | High/Low Boom | 1.5982 | + |
| **Maximum Application Rate** | | | |
| Plane | Forest/ Non-Forest | NA | NA |
| Helicopter | Forest/ Non-Forest | NA | NA |
| Ground | High/Low Boom | 1.6044 | + |

**Effect of Mode of Application**

| | Concentration Ratio[2] | Relative Difference |
|---|---|---|
| **Typical Application Rate** | | |
| Plane vs. Helicopter | 1.1848 | + |
| Plane vs. Ground | 7.4128 | + |
| Helicopter vs. Ground | 6.2569 | + |
| **Maximum Application Rate** | | |
| Plane vs. Helicopter | 1.2043 | + |
| Plane vs. Ground | 8.8813 | + |
| Helicopter vs. Ground | 7.3744 | + |

**Effect of Application Rate**

| | Application Rate[3] | Relative Difference |
|---|---|---|
| Maximum vs. Typical | 4.0183 | + |

[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 diquat screening level ERA incorporates additional conservatism in the assumptions used in the herbicide concentration models such as AgDRIFT® (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 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 diquat 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 0.1 for acute and chronic exposures. For RTE fish and aquatic invertebrates, acute and chronic LOCs were 0.05 and 0.5, respectively. Therefore, up to a 20-fold uncertainty factor has been included in the TRVs for animal species. As noted below, such uncertainty factors provide a greater level of protection to the RTE 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 (i.e., 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 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 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

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

reviewed in order to provide a basis for potential site- and species-specific risk assessment. Review of these factors 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 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 food 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 diquat. 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 the

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

most sensitive species provides a conservative level of protection for all species. The surrogate species used in the diquat 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 1993a, 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 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 diquat 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. However, there is no evidence that the organisms evaluated in the ERA fail to provide reasonable protection as surrogates. Given the very low risks to animals in the modeled exposures, it is unlikely the concentrations of diquat predicted to occur as a result of regular herbicide usage would cause adverse effects to amphibians. Nonetheless, it should be noted that certain 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.

With the exception of the piscivorous birds, ingestion pathway RQs exceeded LOCs for all terrestrial vertebrate surrogate species evaluated in the ERA. The elevated RQs, however, are based on a direct spray of diquat onto the food source of these species. This scenario was included in the ERA to provide a worst-case scenario for terrestrial wildlife exposure, and many uncertainties, as described in Section 7.0, may result in overestimation of the calculated RQs. 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 diquat do exceed LOCs. Careful use of diquat in aquatic systems should minimize terrestrial wildlife exposure and avert potential risks to terrestrial vertebrate RTE wildlife species.

### 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/or 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

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

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.

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; 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 1995b) 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.

[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.

# 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 Endangered Species Act (ESA) of 1973, it is illegal to take an 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 NOAA Fisheries (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 diquat 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 diquat 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. 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 and other fish. 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 and fish 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 and fish 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 and fish. As discussed in Section 4.3, aquatic invertebrate and fish acute and chronic scenario RQs exceeded respective LOCs suggesting potential direct impacts to the forage of salmonids. The only stream evaluation conducted for this risk assessment was an accidental spray scenario and may overestimate risk to aquatic stream receptors. Nonetheless, 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.

As primary producers and the food base of aquatic invertebrates, disturbance to aquatic vegetation may affect the aquatic invertebrate population, thereby affecting salmonids. As diquat is an aquatic herbicide, high RQs for aquatic plants by normal application and other scenarios (i.e., accidental spill or spray) are anticipated. The greatest potential for risk to aquatic vegetation would occur under an accidental spill of diquat into a pond. RQs for aquatic vegetation exceeded LOCs by four orders of magnitude under this spill scenario. RQs in the direct spray to a pond scenario exceeded LOCs by two orders of magnitude, and RQs for accidental spray of a stream exceeded LOCs by three orders of magnitude. Therefore, the use of diquat in streams (the primary salmonid habitat) would be likely to impact aquatic

[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.

vegetation, and potentially indirectly impact salmonids. However, diquat is not used directly to control aquatic vegetation in streams (it is primarily used in ponds); therefore, responsible use of diquat should preclude risks to salmonids.

Although not specifically evaluated in the stream scenarios of the ERA, terrestrial plants were evaluated for their potential to provide overhanging cover for salmonids. A reduction in the riparian cover has the potential to indirectly impact salmonids within the stream. RQs for terrestrial plants were elevated above the LOC for accidental direct spray scenarios at both the typical and maximum application rates, indicating the potential for a reduction in this plant community. The potential for adverse effects on terrestrial plants from spray drift of diquat was also evaluated. In the scenarios using aerial application, non-target terrestrial plant RQs exceeded LOCs for all scenarios. In scenarios where diquat is applied from the ground, there is relatively little risk to typical terrestrial species as close as 100-ft from the source at the typical application rate. It is therefore important to monitor carefully the use of diquat near stream riparian zones. Preferential use of ground applications (instead of aerial applications) and typical application rates would decrease the off-site drift risk to terrestrial plants and the indirect risk to salmonids from loss of vegetative cover.

### 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 4.6.1); 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 herbicide on lands already stressed on 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.

Based on the results of the ERA, there is potential for risk to non-target terrestrial and aquatic receptors (animal and plant) in extreme circumstances, such as incidents of spills or accidental direct spray (Sections 4.3.1 and 4.3.5). Therefore, while it is unlikely that responsible use of diquat by BLM land managers will indirectly affect RTE salmonids through the killing of in-stream or riparian vegetation, using diquat frequently or in large amounts (particularly if applied aerially) may increase risk to RTE species (salmonids) as a result of impacts to riparian vegetation. It may be productive to develop a more site- and/or species-specific ERA in order to assure that the proposed herbicide application will not result in secondary impacts to salmonids associated with loss of riparian cover.

## 6.5 Conclusions

The diquat 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, toxicity data for surrogate species were used

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

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 ERAs. 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).

Potential primary and secondary effects of diquat use should be of concern for the protection of RTE species. Habitat disturbance and disruptions in the food chain are often the cause of declines of populations and 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.

Aquatic herbicides may harm primary producers in the waterbodies or reduce riparian zones. The results of the ERA indicate that non-target terrestrial and aquatic plants are at risk from diquat, especially when accidents occur, such as spills or accidental spraying. Accidents also pose some risk to fish (including RTE salmonids) and aquatic invertebrates, both directly and indirectly via impacts to aquatic plants.

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 be occur beyond the season of application.

Based on the results of the ERA, it is important that diquat be used in the most appropriate and responsible manner to prevent harm to RTE species on BLM-managed lands. Certain application guidelines and restrictions (e.g., application rate, buffer distance, avoidance of designated critical habitat) for appropriate and responsible use of the herbicide diquat on BLM-managed lands would reduce risk to some non-target species (see Section 8).

**TABLE 6-1**
**Surrogate Species Used to Derive Diquat TRVs**

| Species in Diquat Laboratory/Toxicity Studies | | Surrogate for |
|---|---|---|
| Honeybee | *Apis mellifera* | Pollinating insects |
| Rat | *Rattus norvegicus* spp. | Mammals |
| Dog | *Canis familiaris* | Mammals |
| Ring-necked pheasant | *Phasianus colchicus* | Birds |
| Japanese Quail | *Coturnix coturnix* | Birds |
| Bobwhite Quail | *Colinus virginianus* | Birds |
| Mallard | *Anas platyrhynchos* | Birds |
| Rabbit | *Leporidae sp.* | Mammals |
| Cotton | *Gossypium* spp. | Non-target terrestrial plants |
| Amphipod | *Hyalella azteca* | Aquatic invertebrates |
| Walleye | *Stizostedion vitreum* | Fish |
| Daphnid | *Daphnia magna* | Aquatic invertebrates |
| Giant Duckweed | *Lemna sp.* | Non-target aquatic plants |
| Rainbow trout | *Oncorhynchus mykiss* | Fish/Salmonids |

**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**
**Rare, Threatened, and Endangered Birds and Selected Surrogates**

| RTE Avian Species Potentially Occurring on BLM-managed 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<br>Coyote |
| Cactus ferruginous pygmy-owl | *Glaucidium brasilianum cactorum* | Carnivore | Bald eagle<br>Coyote |
| Whooping crane | *Grus Americana* | Piscivore | Bald eagle |
| California condor | *Gymnogyps californianus* | Carnivore | Bald eagle<br>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<br>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<br>Coyote |
| Spectacled eider | *Somateria fischeri* | Omnivore [Insectivore/herbivore] | American robin<br>Canada goose |
| Least tern | *Sterna antillarum* | Piscivore | Bald eagle |
| Northern spotted owl | *Strix occidentalis caurina* | Carnivore | Bald eagle<br>Coyote |
| Mexican spotted owl | *Strix occidentalis lucida* | Carnivore | Bald eagle<br>Coyote |
| Least Bell's vireo | *Vireo bellii pusillus* | Insectivore | American robin |

**TABLE 6-4**
**Rare, Threatened, and Endangered Mammals and Selected Surrogates**

| RTE Mammalian Species Potentially Occurring on BLM-managed 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<br>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<br>Bald eagle |
| Steller sea-lion | *Eumetopias jubatus* | Carnivore/piscivore | Coyote<br>Bald eagle |
| Sinaloan jaguarondi | *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 Joiquin 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<br>Mule deer<br>Bald eagle |
| San Joaquin kit fox | *Vulpes macrotis mutica* | Carnivore | Coyote |
| Preble's meadow jumping mouse | *Zapus hudsonius preblei* | Omnivore [herbivore/ insectivore] | Deer mouse<br>American robin |

Note: Four whales and one seal 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 6-5**
**Rare, Threatened, and Endangered Reptiles and Selected Surrogates**

| RTE Reptilian Species Potentially Occurring on BLM-managed lands | | RTE Trophic Guild | Surrogates |
|---|---|---|---|
| New Mexican ridge-nosed rattlesnake | *Crotalus willardi obscurus* | Carnivore/insectivore | Coyote<br>Bald eagle<br>American robin |
| Blunt-nosed leopard lizard | *Gambelia silus* | Carnivore/insectivore | Coyote<br>Bald eagle<br>American robin |
| Desert tortoise | *Gopherus agassizii* | Herbivore | Canada goose |
| Giant garter snake | *Thamnophis gigas* | Carnivore/insectivore/piscivore | Coyote<br>American robin<br>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**
**Rare, Threatened, and Endangered Amphibians and Selected Surrogates**

| RTE Amphibious Species Potentially Occurring on BLM-managed lands | | RTE Trophic Guild | Surrogates |
|---|---|---|---|
| California tiger salamander | *Ambystoma californiense* | Invertivore[1] | Bluegill sunfish<br>Rainbow trout[3] |
| | | Vermivore[2] | American robin[4] |
| Sonoran tiger salamander | *Ambystoma tigrinum stebbinsi* | Invertivore/insectivore[1] | Bluegill sunfish<br>Rainbow trout[3] |
| | | Carnivore/ranivore[2] | American robin[4] |
| Desert slender salamander | *Batrachoseps aridus* | Invertivore | American robin[4,5] |
| Wyoming toad | *Bufo baxteri* | Insectivore | Bluegill sunfish<br>Rainbow trout[3]<br>American robin[4] |
| Arroyo toad (=Arroyo southwestern toad) | *Bufo californicus* | Herbivore[1] | Bluegill sunfish<br>Rainbow trout[3] |
| | | Invertivore[2] | American robin[4] |
| California red-legged frog | *Rana aurora draytonii* | Herbivore[1] | Bluegill sunfish<br>Rainbow trout[3] |
| | | Invertivore[2] | American robin[4] |
| Chiricahua leopard frog | *Rana chiricahuensis* | Herbivore[1] | Bluegill sunfish<br>Rainbow trout[3] |
| | | Invertivore[2] | 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 BW 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-managed 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. |
| 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 BW 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 prey 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 increases 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 TRVs, 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]<br>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. | Fletcher et al. 1990 |
| | 3 of 20 $EC_{50}$ values from field data were 2-fold higher than lab study data. | |
| 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 overpredicted 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).

Ten diquat incident reports were available from the USEPA's Environmental Fate and Effects Division (EFED) (Table 2-2). Incident reports can be used to validate exposure models and hazards to ecological receptors. These reports, described in Section 2.3, indicated that in seven incidents, mortality of fish and other fauna were attributed to diquat; three incidents referenced plant damage. Two of the seven incidents had references to additional pesticide use (malathion and copper sulfate). The incident reports listed the likelihood that diquat caused the observed damaged as "probable" in seven incidents and "possible" in three incidents. These results support the risk assessment predictions of risk to fish, aquatic invertebrates and aquatic plants due to direct spray of the stream and pond. However, since the incident reports provide limited information, and diquat was mixed with other products, it is impossible to correlate the impacts predicted in the ERA with the incident reports.

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. As discussed previously, diquat is an aquatic herbicide, and it is not expected that terrestrial plants will be exposed to this herbicide. It is possible that the non-cropland species present on BLM managed lands may be more tolerant of diquat than the surrogate species used in the ERA.

In general, the most sensitive available endpoint for the appropriate surrogate test species was used to derive TRVs. This approach is conservative since there may be a wide range of data and effects for different species. As described

in Section 4.3.3.1, although salmonids are often considered to be highly sensitive receptors, the acute fish TRV is actually based on a warmwater species. The selected $LC_{50}$ for the walleye, 0.17 mg a.i./L, is significantly lower than the lowest salmonid $LC_{50}$, 19 mg a.i./L. This selection criterion for the TRVs has the potential to overestimate risk within the ERA.

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.

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 active ingredients 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 active ingredients 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 diquat, the percent active ingredients, listed in Appendix A when available from the reviewed study, ranged from 0.02% to 99.6%, with most formulations in the 25% to 45% range. As indicated in Section 3.1, test materials for diquat dibromide frequently contain 40% or less of diquat cation, and this is the highest purity produced (USEPA 1995a). These toxicity data likely represent the potential impacts due to the types of formulations that BLM would likely use. However, the concentrations of diquat presented in the spreadsheets represent the a.i. itself, not the formulation. Therefore, TRVs based on a product with relatively low % diquat could underestimate potential risks. A review of the data identified three TRVs that were based on data presented on a product, not a.i., basis. These TRVs were used to evaluate acute impacts to aquatic invertebrates and acute and chronic impacts to aquatic plants. However, impacts are already predicted for both aquatic invertebrates and aquatic plants and algae, so lowering the TRVs (based on 41.4% and 35.3% a.i., respectively) would not change the overall conclusions of the risk assessment significantly. The remaining TRVs are based on studies presenting data on an a.i. basis.

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

## 7.2 Potential Indirect Effects on Salmonids

No actual field studies or ecological incident reports on the effects of diquat 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 on salmonid populations and communities. The chronic 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. The acute fish TRV is based on toxicity values lower than those observed for salmonids, indicating that acute risks to salmonids may be overestimated.

A discussion of the potential indirect impacts to salmonids is presented in Section 4.3.4, and Section 6.4 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) or a reduction in vegetative cover. It is anticipated that these qualitative evaluations overestimate the potential risk to salmonids due to the conservative selection of TRVs for salmonid prey and terrestrial vegetation, 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, adjuvants, and degradates. Other herbicides may also factor into the risk estimates, as many herbicides are 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 compare deterministic risk calculations (i.e., exposure modeling, effects assessment, and RQ calculations) for a single a.i..

In addition, information on inerts, adjuvants, surfactants, 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 due to degradates, 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 diquat. Degradates may be more or less mobile and more or less toxic in the environment than their source herbicides (Battaglin et al. 2003). Differences in environmental behavior (e.g., mobility) and toxicity between parent herbicides and 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 diquat represents a source of uncertainty in the risk assessment.

### 7.3.2 Inerts

Pesticide products contain both active and inert ingredients. The terms “a.i.” 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 mislead 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 Integrated Risk Information System [IRIS], the HSDB, 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 diquat: low application rates for diquat resulted in low exposure concentrations of inerts of much less than 1 mg/L in all modeled cases. This indicates that inerts associated with the application of diquat 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 diquat 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 or not a particular herbicide can be tank mixed with other pesticides. Adjuvants (e.g., surfactants, crop oil concentrates, fertilizers) may also be added to the spray mixture to improve the herbicide efficacy. 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 the particular herbicide.

In reviewing the labels for Reward® (Syngenta 2001), the following adjuvants (only non-ionic surfactants) were identified on the labels:

- Spot spray – 1 to 2 quarts Reward® plus the labeled rate of a 75% or greater nonionic surfactant per 100 gal of water, or 0.75 oz Reward® plus the labeled rate of a 75% or greater nonionic surfactant per 1 gal of water.

- Broadcast Spray – 1 to 2 pints Reward® in a minimum of 15 gal of water per ac. Add labeled rate of a 75% or greater nonionic surfactant per 100 gal of water.

- Broadcast Spray for Ornamental Seed Crops– 1.5 to 2 pints Reward® plus the labeled rate of a 75% or greater nonionic surfactant per ac in sufficient water (minimum of 5 gal by air or 15 gal by ground).

In general, adjuvants compose a relatively small portion of the volume of herbicide applied. However, it is recommended that an adjuvant with low toxic potential be selected. Potential toxicity of any material should be considered prior to its use as an adjuvant.

As described in Section 7.3.2, sources (Muller 1980; Lewis 1991; Dorn et al. 1997; Wong et al. 1997) 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. Since no application rates were provided for the nonionic surfactants, the maximum application rate for diquat was used (4.0 mg a.i./L). With a 75% nonionic surfactant, this predicted a maximum concentration of 3.0 mg/L, within the range indicated for acute toxicity to aquatic life. This concentration is likely an overestimate of the concentration of any inert that would be present because the Reward® label did not provide application rate information for adjuvants. This calculation was performed assuming that the adjuvant application rate equals the diquat application rate and the adjuvant was present at 75% of the diquat concentration.

This evaluation indicates that, for herbicides with high application rates, adjuvants have the potential to cause acute and potentially chronic risk to aquatic species. However, more specific modeling and toxicity data would be necessary to define the level of uncertainty. Selection of adjuvants is under the control of the BLM land managers and it is recommended that land managers follow all label instructions and abide by any warnings. Selection of adjuvants with limited toxicity and low volumes is recommended to reduce the potential for the adjuvant to increase the toxicity of the herbicide.

#### 7.3.3.2 Tank Mixtures

According to the labels, diquat, as the a.i. in Reward®, may be mixed with pre-emergent herbicides (specific herbicides are not indicated). However, it is not generally within BLM practice to tank mix diquat with any other products. Therefore, additional modeling of tank mixes was not performed for diquat.

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 when using 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., wind speed, nozzle type) will impact the models used to predict the off-site drift impacts of herbicide use (i.e., AgDRIFT®); 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., BW, 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 terrestrial or aquatic 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 food 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 food | Unknown | Toxicity to food receptors may result in sickness or mortality. Fewer food items would be available for predators. Predators may stop foraging in areas with reduced prey populations, discriminate against, or conversely, select contaminated food items. |
| 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 food by another receptor. As indicated with the AgDRIFT® model, off-site drift is only a fraction of the applied amount. |

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

| Potential Source of Uncertainty | Direction of Effect | Justification |
|---|---|---|
| Ecological exposure concentration | Overestimate | It is unlikely any receptor would be exposed continuously to the full predicted EEC. |
| Over-simplification of dietary composition in the food web models | Unknown | Assumptions were made that contaminated food items (i.e., vegetation, fish) were the primary food items for wildlife. In reality, other food items are likely consumed by these organisms. |
| 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 reptile and amphibian species 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 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 overestimate 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. |

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

| Potential Source of Uncertainty | Direction of Effect | Justification |
|---|---|---|
| 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. |
| 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 diquat, there is the potential for risk to 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 due to the mode of action of herbicides, the highest risk is predicted for non-target terrestrial and aquatic plant species, generally under accidental exposure scenarios (i.e., direct spray and accidental spills). Minimal risk was predicted for terrestrial animals. Risk to fish and aquatic invertebrates was predicted in the direct spray and spill scenarios.

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

- Direct Spray – Risks were predicted for pollinating insects due to direct spray and indirect contact with contaminated foliage. Acute risks were predicted for 5 of 10 wildlife scenarios, and chronic risks were predicted for 9 of 10 wildlife scenarios (risk is somewhat lower at the typical application rate versus the maximum rate). Risk to terrestrial and aquatic non-target plants, fish, and aquatic invertebrates is likely when waterbodies are accidentally or intentionally (pond applications) sprayed. No risks were predicted for piscivorous wildlife.

- Off-Site Drift to Non-Target Terrestrial Plants – Risks to typical and RTE terrestrial plant species were predicted within 900 ft of all aerial applications at both the typical and maximum application rates. RQs were also elevated for typical and RTE plant species within 100 ft of ground applications (low and high booms).

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

In addition, species that depend on non-target species for habitat, cover, and/or food may be indirectly impacted by a possible reduction in terrestrial or aquatic vegetation, particularly in accidental direct spray and spill scenarios. For example, RTE salmonids may be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates) or a reduction in vegetative cover in the relatively unlikely case of an accidental spill to the stream.

Based on the results of the ERA, it is unlikely that RTE species would be harmed by appropriate use of the herbicide diquat on BLM-managed lands. Although non-target terrestrial plants have the potential to be adversely affected by off-site drift of diquat, adherence to certain application guidelines (e.g., defined application rates, equipment, herbicide mixture, and downwind distance to potentially sensitive habitat) would minimize the potential effects on non-target plants and associated indirect effects on species that depend on those plants for food, habitat, and cover.

## 8.1 Recommendations

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

- Select herbicide products carefully to minimize additional impacts from adjuvants. 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 and spill conditions to reduce the greatest potential impacts.

- Use the typical application rate, rather than the maximum application rate, to reduce risk to some wildlife receptors (e.g., large carnivore, small bird).

- Use ground application methods at the typical or maximum application rate with buffer zones of more than 100 ft (no risk predicted at 900 ft) to reduce potential risks to typical terrestrial plants from off-site drift.

- To reduce potential risks to RTE terrestrial plants from off-site drift during ground applications, use buffer zones of more than 100 ft at the typical rate, or buffer zones of at least 1,000 ft when using the maximum application rate (based on simple regression evaluation of RQs and distance).

- Buffer zones of greater than 900 ft may be necessary if aerial application is planned. Regression analysis indicates that a buffer zone of 1,200 ft should be protective of typical and RTE plant species.

- Avoid application to ponds where RTE species have been identified.

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 diquat 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 Diquat 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 [10: 16] | M [7: 16] | NA | NA | NA | NA | NA | NA |
| **Terrestrial Plants (Typical Species)** | H [1: 1] | H [1: 1] | L [7: 12] | M [7: 12] | NA | NA | NA | NA |
| **Terrestrial Plants (RTE Species)** | H [1: 1] | H [1: 1] | M [7: 12] | M [7: 12] | NA | NA | NA | NA |
| **Fish In The Pond** | L [2: 2] | H [2: 4] | NA | NA | NA | NA | NA | NA |
| **Fish In The Stream** | M [1: 2] | M [2: 2] | NA | NA | NA | NA | NA | NA |
| **Aquatic Invertebrates In The Pond** | M [2: 2] | H [3: 4] | NA | NA | NA | NA | NA | NA |
| **Aquatic Invertebrates In The Stream** | H [1: 2] | H [2: 2] | NA | NA | NA | NA | NA | NA |
| **Aquatic Plants In The Pond** | H [2: 2] | H [4: 4] | NA | NA | NA | NA | NA | NA |
| **Aquatic Plants In The Stream** | H [2: 2] | H [2: 2] | NA | NA | NA | NA | NA | NA |
| **Piscivorous Bird** | NA | NA | 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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