Bureau of Land Management

Reno, Nevada

Vegetation Treatments Programmatic EIS Ecological Risk Assessment Protocol

Final Report

October 2004

Bureau of Land Management Contract No. NAD010156
ENSR Document Number 09090-020-652

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LIST OF ACRONYMS, ABBREVIATIONS, AND SYMBOLS

AGDISP	-	Agricultural Dispersion Model
AE	-	Assessment Endpoint
a.i.	-	active ingredient
BA	-	Biological Assessment
BCF	-	Bioconcentration Factor
BLM	-	Bureau of Land Management
BW	-	Body Weight
CBI	-	Confidential Business Information
CF	-	Conversion Factor
cm	-	centimeter
cms	-	cubic meters per second
CREAMS	-	Chemical Runoff Erosion Assessment Management System
CTWA	-	Time-weighted Average Concentration
EC_{25}	-	25% Effect Concentration
EC_{50}	-	Median Effect Concentration
EEC	-	Estimated Exposure Concentration
EFED	-	Environmental Fate and Effects Division
EIS	-	Environmental Impact Statement
ERA	-	Ecological Risk Assessment
FCM	-	Food Chain Multiplier
g	-	gram
GLEAMS	-	Groundwater Loading Effects of Agricultural Management Systems
IRIS	-	Integrated Risk Information System
kg	-	kilogram
km	-	kilometers
K_{oc}	-	Octanol-water Partitioning Coefficient
L	-	Liter
lb	-	pound
LC_{50}	-	Median Lethal Concentration
LD_{50}	-	Median Lethal Dose
LOC	-	Levels of Concern
m	-	meter
mg/kg	-	milligrams per kilogram
mg/L	-	milligrams per liter
MW	-	Molecular Weight
NOAEC	-	No Observed Adverse Effect Concentration
NOAEL	-	No Observed Adverse Effect Level
OPP	-	Office of Pesticide Programs
RQ	-	Risk Quotient
RTE	-	Rare, Threatened, and Endangered
SDTF	-	Spray Drift Task Force
TL	-	Trophic Level
TRV	-	Toxicity Reference Value
USDA	-	U.S. Department of Agriculture
USEPA	-	U.S. Environmental Protection Agency
USLE	-	Universal Soil Loss Equation
µg	-	microgram
µm	-	micrometer

1.0 INTRODUCTION

To evaluate the effects of the proposed vegetation treatment methods and alternatives on public lands managed by the U.S. Department of Interior (USDI) Bureau of Land Management (BLM) in the western continental U.S., and Alaska, the BLM and its contractor, ENSR, has prepared a *Vegetation Treatment Using Herbicides on Bureau of Land Management Lands in 17 Western States Programmatic Environmental Impact Statement* (PEIS; USDI BLM 2005a). The PEIS serves to update the following four EISs developed by the BLM in the mid 1980s and early 1990s:

- Northwest Area Noxious Weed Control Program – 1986
- California Vegetation Management – 1988
- Vegetation Treatment of BLM Lands in Thirteen Western States – 1991
- Western Oregon Program Management of Competing Vegetation – 1992

As part of the PEIS, an ecological risk assessment (ERA) was conducted on several herbicide active ingredients (a.i.; hereafter simply referred to as herbicides) currently used, or proposed for use, by the BLM. The objective of the ERA was to evaluate potential risks to human health and the environment that may result from exposure to the herbicides both during and after treatment of public lands.

The 10 herbicides under consideration in the ERA include eight terrestrial herbicides: bromacil, chlorsulfuron, diflufenzopyr, diflufenzopyr + dicamba, diuron, imazapic, sulfometuron methyl, and tebuthiuron; and two aquatic herbicides: diquat and fluridone.

1.1 Purpose

This document provides the details of the ERA methodology used to evaluate risks associated with the use of herbicides for controlling invasive vegetation and to determine if these herbicides are safe for use by BLM. The protocol on which these methods are based was approved by the BLM in 2003.

In addition to the risk assessment methodology, this document provides an overview of the application of three software models used to predict the concentrations of herbicide in various environmental media (animal and plant tissues, soil, and water) due to spray drift, surface runoff, or herbicide transport on wind-blown dust. These programs are AgDRIFT®, GLEAMS (Groundwater Loading Effects of Agricultural Management Systems), and CALPUFF.

1.2 Document Organization

The remainder of this document is organized in the following manner:

- Section 2.0 presents a general overview of the ERA methodology.
- Section 3.0 presents a description of the exposure pathways and dose calculations.
- Section 4.0 presents an overview of the application of the three software models used to predict media concentrations.
- Section 5.0 presents references.

More extensive descriptions of the specific inputs and methodologies for the three models described in Section 4.0 of this document are provided in appendices to this protocol and in a supplemental report for the PEIS. This information may be found in the following locations:

- AgDRIFT® – Appendix A of this document
- GLEAMS – Appendix B of this document

- CALPUFF – *Vegetation Treatment Programmatic EIS Air Quality Impact Assessment Protocol* (ENSR 2004)

2.0 OVERVIEW OF ECOLOGICAL RISK ASSESSMENT

Ecological risk assessments were produced for 10 herbicides: bromacil, chlorsulfuron, dicamba, diflufenzopyr, diquat, diuron, fluridone, imazapic, sulfometuron methyl, and tebuthiuron (ENSR 2005a-j). The ERAs for each of the herbicides were produced as separate documents. While the risk assessments have been tailored to address the potential usage of each particular herbicide, they follow the same essential format and methodology, which is described below. Each ERA includes the following sections:

- Introduction – covers general concepts and document overview.
- BLM Herbicide Program Description – describes BLM-specific uses of the product, statistics of use to date, and incident reports compiled by the USEPA.
- Herbicide Toxicology, Environmental Fate, and Physical-Chemical Properties – discusses the review of toxicity literature and its results, environmental fate of the herbicide, and specific physical-chemical properties used in the ERA.
- Ecological Risk Assessment – evaluates potential risk to ecological receptors resulting from exposure to the herbicide in a number of different scenarios (discussed in more detail in this section).
- Sensitivity Analysis – discusses the sensitivity of predicted exposure concentrations to variation in environmental processes and the models used to represent them. This analysis is provided in order to verify that most predicted concentrations are overestimates, and to identify situations where the general assumptions of the models might be relaxed or should be made more stringent.
- Rare, Threatened, and Endangered (RTE) Species – discusses potential direct and indirect impacts to RTE species, including consideration of taxa for which ecotoxicological data are not available.
- Uncertainty in the Ecological Risk Assessment – describes data gaps, assumptions, and uncertainties of the risk assessment.
- Summary – summarizes the overall implications of the risk assessment.
- References – presents references considered in the document.

The following appendices are also included in each ERA:

- Summary of Available and Relevant Toxicity Data/Ecological Risk Assessment Literature Review
- Ecological Risk Assessment Worksheets
- Species Listed Under the Endangered Species Act for 17 BLM States
- Review of Confidential Business Information Memo
- Summary of Tank Mix Risk Quotients

Transport via surface runoff and wind-blown dust, and the resulting exposure concentrations, are not included in the ERAs for the aquatic herbicides (diquat and fluridone). Therefore, exposure scenarios and appendices relating to GLEAMS and CALPUFF are not included in the risk assessment documents for these herbicides.

The overall goal of the ERAs is to facilitate risk management decisions for the PEIS and support development of the Biological Assessment (BA) for the PEIS. An additional goal of this process is to provide risk managers with a tool that presents a range of generic risk estimates that vary as a function of site conditions. The tool to accomplish this primarily consists of the Excel spreadsheets (presented in the ERA Worksheets) that may be used to calculate exposure concentrations and evaluate potential risks provided in the risk assessment. For further site-specific evaluation of a particular herbicide, BLM land managers can modify specific variables in the worksheets.

The general approach and analytical methods for conducting the ERAs were based on U.S. Environmental Protection Agency's (USEPA) *Guidelines for Ecological Risk Assessment* (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 more detail in the following subsections.

2.1 Problem Formulation

The problem formulation provides a foundation for the entire ERA process by detailing the risk hypotheses to be evaluated in support of appropriate risk management decisions. The success of the problem formulation depends for the most part on the assessment endpoints (AE; expressions of the environmental value to be protected), development of appropriate conceptual models (graphical representations of exposure pathways), and statement of the analysis plan (specific risk assessment methods).

The problem formulation section presents information related to chemical characteristics, mode of toxic action, and herbicide usage presented in the BLM Herbicide Program Description and Herbicide Toxicology, Environmental Fate, and Physical-Chemical Properties sections of the ERA reports. This information is used to help identify management goals and AEs.

The primary management goal for the ERAs was to provide estimates of ecological risk for the herbicide active ingredients evaluated in the PEIS and BA. Estimates of ecological risk were necessary to help establish whether herbicide use may pose unacceptable risk to terrestrial wildlife, non-target plants, and aquatic organisms. More specifically, the management goals were to prevent/minimize direct (acute and chronic) and indirect effects of herbicides to these groups of organisms, including threatened and endangered species. Acute effects are those that have the potential to occur following a short-term exposure to a relatively high dose of a toxin, and are usually measured by mortality (death) of the organism. Chronic effects are those that have the potential to occur after long, often low-dose exposure, and may include reproductive, developmental, or cellular-level effects.

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

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

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

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

- **Assessment Endpoint 2:** Acute mortality to fish, aquatic invertebrates, and aquatic plants. **Measures of Effect** include median lethal effect concentrations (e.g., LC_{50}) from acute toxicity tests with these organisms or suitable surrogates (e.g., data from other coldwater fish were used to represent threatened and endangered salmonids).

- **Assessment Endpoint 3:** Adverse direct effects on growth, reproduction, or other ecologically important sublethal processes. **Measures of Effect** include standard chronic toxicity test endpoints such as the No Observed Adverse Effect Level ([NOAEL] the concentration tested at which no adverse effects to test organisms were noted[1]) for both terrestrial and aquatic organisms. Depending on data available for a given herbicide, chronic endpoints reflect either individual-impacts[2] (e.g., survival, individual growth, or sub-lethal responses such as physiological impairment or behavior), or population-level impacts[3] (e.g., reproduction; [Barnthouse 1993]). For salmonids, careful attention was paid to smoltification (i.e., development of tolerance to seawater and other changes of parr [freshwater stage salmonids] to adulthood), thermoregulation (i.e., ability to maintain body temperature), migratory behavior, and other important life processes, if such data were available. With the exception of non-target terrestrial plants, standard acute and chronic toxicity test endpoints were used for estimates of direct herbicide effects on rare, threatened, and endangered (RTE) species. To add conservatism to the RTE assessment, levels of concern (LOCs) for RTE species were lower (more sensitive) than for typical species. For terrestrial plant species, the LOC was set at 1 for all scenarios and different toxicity values were used to provide extra protection for RTE plant species. In the direct spray, spray drift, and wind erosion scenarios, the selected toxicity endpoints were a 25% Effect Concentration (EC_{25}) for "typical" species and a NOAEL for RTE species for any toxicity endpoint (i.e., vigor, emergence, germination). In runoff scenarios, high and low germination NOAELs were selected to evaluate exposure for typical and RTE species, respectively. When germination was unavailable, seed emergence data was acceptable to evaluate surface runoff exposure.

- **Assessment Endpoint 4:** Adverse indirect effects on the survival, growth, or reproduction of salmonids. **Measures of Effect** for this AE depended on the availability of appropriate scientific data. Unless literature studies were found that explicitly evaluated the indirect effects of the target herbicides to salmonids and their habitat, estimates of indirect effects were qualitative. Such qualitative estimates of indirect effects include general evaluations of the potential risks to food (typically represented by acute and/or chronic toxicity to aquatic invertebrates) and cover (typically represented by potential for destruction of riparian vegetation), as

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[1] NOAEC (no observed adverse effect concentration) is sometimes used. The terms NOAEL and NOAEC are essentially interchangeable.

[2] Individual impacts are those that adversely affect a single individual member of the species. RTE species are evaluated using individual effects as the measure of effect because each individual is important to the maintenance of the species.

[3] Population impacts adversely affect the maintenance of a population—group of the same species occupying the same time and place.

well as habitat modification from increased sedimentation and temperature. The USEPA Office of Pesticide Programs (OPP) is currently applying approaches similar to these qualitative evaluations for RTE species effects determinations and consultations.[4]

Problem formulation also included a graphical conceptual model indicating the possible exposure pathways for each herbicide, and thus which types of surrogate species (i.e., receptors) were evaluated for each herbicide. The conceptual models for terrestrial and aquatic herbicide active ingredients are presented in Figures 2-1 and 2-2, respectively. An overall analysis plan was also presented that consists of a general statement of the kinds of data and decision-making methods that are proposed for use in the ERA. Figure 2-3 presents the trophic levels (TLs) evaluated for terrestrial and aquatic herbicide active ingredients.

2.2 Analysis

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 exposure point concentrations in various environmental media (e.g., GLEAMS). The ecological effects characterization consists, for the most part, of compiling exposure-response relationships from all available toxicity studies for each herbicide.

2.2.1 Exposure Characterization

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

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

- Direct spray of the receptor or waterbody
- Indirect contact with dislodgeable foliar residue
- Ingestion of contaminated food items
- Off-site drift of spray to terrestrial areas and waterbodies
- Surface runoff from the application area to off-site soils or waterbodies
- Wind erosion resulting in deposition of contaminated dust
- Accidental spills to waterbodies

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

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

Additional details regarding particular receptors (e.g., receptor size, diet, consideration of threatened and endangered species), application rates (i.e., typical and maximum application rates and accidental spills), duration of herbicide exposure (i.e., one time event or longer-term exposure) and toxicity endpoints (i.e., acute, chronic) are discussed below. Further information can be found in the individual risk assessment spreadsheets compiled for each herbicide (Appendix B of each ERA; ENSR 2005a-j).

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

Unless application information from the herbicide-use label indicates otherwise, exposure concentrations were estimated for a single annual application. However, there might be chemical-specific exceptions to this. This application is evaluated on a case-by-case basis, and reflects a best estimate of maximum possible application rates and frequency at any one location.

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

The AgDRIFT® computer model (Section 4.1) was used to estimate off-site herbicide transport due to spray drift. The GLEAMS computer model (Section 4.2) was used to estimate off-site herbicide transport in surface runoff and root zone groundwater (Knisel and Davis 2000). GLEAMS is a particularly important model in that it calculates soil concentrations at the site of application and the amount of herbicide that might be transported in surface water or contaminated groundwater into adjacent terrestrial areas or aquatic habitats (e.g., ponds, streams). Furthermore, model inputs are sufficiently flexible such that a wide range of exposure scenarios (e.g., number of annual herbicide applications, soil characteristics) can be accommodated (Knisel and Davis 2000). Finally, the model predicts longer-term loading, such as that associated with continued soil erosion. The computer model CALPUFF (Section 4.3) was used to predict the transport and deposition of herbicides sorbed (reversibly or temporarily attached) to wind-blown dust.

AgDRIFT® and CALPUFF simulate one time transport events, and estimate herbicide concentrations based on a single occurrence of herbicide use (e.g., how much herbicide is transported to a pond by off-site drift during a particular application event). In contrast, soil concentrations reach a steady-state over time using GLEAMS. Therefore, GLEAMS models 10 years of annual applications to simulate the impact of herbicide accumulation in the system. Data from the final year of the GLEAMS simulations were used in the ERA. Each model simulation was approached with the intent of predicting the maximum potential herbicide concentration that could result from the given exposure scenario.

2.2.2 Effects Characterization

The ecological effects characterization phase of an ERA entails a compilation and analysis of the stressor-response relationships and any other evidence of adverse impacts from exposure to each herbicide. Available data consisted mostly of the toxicity studies conducted in support of USEPA pesticide registration, which generally included the following (additional studies may be required depending on herbicide use patterns and characteristics (40 Code of Federal Regulations [CFR] 158):

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

Additional tests, if required for a particular herbicide, may include honeybee (*Apis mellifera*) acute toxicity, avian reproduction, non-target plant toxicity tests, and chronic fish life-cycle tests, among others. These data were identified during a literature review phase which consisted of an evaluation of the existing literature, including published manuscripts, unpublished study reports, electronic databases, and information provided by USEPA as part of a Freedom of Information Act (FOIA) request for available toxicity data related to herbicide registrations.

As data were not available for all receptors or for RTE species, extrapolation of risk based on surrogate species data was necessary. Species for which toxicity data were 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. For example, the majority of the terrestrial plant toxicity data is based on crop species (e.g., tomato [*Solanum* spp.], soybean [*Glycine* sp.]) and not rangeland species more likely to occur on BLM lands. However, it is likely that these tested species are at least as sensitive, and likely more sensitive, than the rangeland species. The selected toxicity value for a particular receptor is based on a review of the available data and the selection of the most appropriate sensitive surrogate species.

2.3 Risk Characterization

The final phase of an ERA consists of quantitative estimates of the ecological risks, a description of data used in support of these risk estimates (including data gaps where appropriate), a discussion of uncertainties in this analysis, and an overall interpretation of the potential ecological impacts of each herbicide.

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

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

The RQ approach used in the risk assessment 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., body weight [BW], ingestion rates). The ERA reports will include a discussion of the uncertainties inherent in the RQ methodology.

2.3.1 Toxicity Reference Values

In the majority of cases, toxicological data do not exist for the specific ecological receptors of concern considered in the risk assessment (i.e., American robin [*Turdus americanus*] and mule deer [*Odeocoileus hemionus*]). Consequently, toxicological data for surrogate species were evaluated and used to establish quantitative benchmarks for the ecological receptors of concern. These benchmark values are referred to as TRVs. This section of text briefly describes the process used to derive TRVs. Once developed, TRVs were compared with predicted environmental concentrations to determine the likelihood of adverse effects to ecological receptors.

2.3.1.1 Literature Review

The process for deriving TRVs consisted of assembling relevant literature, evaluating these information sources, and then establishing specific numeric values for each ecological receptor of concern considered in the risk assessment worksheets. The first step to derive ecological TRVs consisted of an evaluation of the existing literature, including published manuscripts, unpublished study reports, and electronic databases. The primary sources of information are summarized in Table 2-1.

Once data from these various sources were compiled, the information was reviewed to determine its acceptability for deriving ecological TRVs for each of the herbicides evaluated. In order to be classified as an "acceptable" study, the research had to be suitable (i.e., the data were relevant to endpoints of interest) and adequate (i.e., the data were of high enough quality).

TABLE 2-1
Summary of Information Sources Used to Derive Toxicological Reference Values

Information Source	Representative Examples
Freedom of Information Act (FOIA) requests	EPA registration studies and related documents (primarily unpublished information)
Peer-reviewed literature	Published manuscripts
Electronic databases	USEPA Pesticide Database; USEPA Integrated Risk Information System (IRIS) database, Hazardous Substance Data Bank; and 12 literature databases (AGRICOLA, ASFA, Biological Sciences, BIOSIS/Biological Abstracts, Chem Abstract/Scifinder Scholar, Environmental Science and Pollution Management, MedLine, Safety Science and Risk, Toxline, Water Resources Abstracts, Web of Science/Science Citation Index, and Zoological Records; searched dates 1970 to 2003)
Internet	Forest Service pesticide review documents; USEPA registration and re-registration documents; and California EPA pesticide registration data

Data Suitability

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

- The material tested was one of the herbicides under consideration
- The test species was in the same guild as an ecological receptor considered in the risk assessment
- The route of exposure matched the primary routes of exposure
- The toxicity endpoint (e.g., mortality, reproductive success, growth) was considered to be ecologically relevant (see Table 2-2)

For the majority of studies, the acute statistical endpoints consisted of LD_{50}, LC_{50}, or EC_{50} (median effect concentration) values. Adverse effect levels in chronic studies were most frequently reported as lowest-observed-adverse-effect levels (LOAELs). Levels at which no effects were noted were generally reported in chronic studies as NOAELs. As discussed in Section 2.1, several additional statistical endpoints were evaluated for terrestrial plants, including EC_{25} and NOAEL (for any relevant endpoint) to address risk due to direct spray, off-site drift, and dust, and the highest and lowest germination- or emergence-based NOAEL to address risk due to surface runoff.

Data Adequacy

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

Information obtained from the USEPA (e.g., unpublished reports and databases) and from other information sources was often incomplete and insufficient to independently evaluate the validity of the study. Nevertheless, if the USEPA had reviewed the study and classified the study as “acceptable,” the study’s findings were considered “acceptable” for development of TRVs for this document.

TABLE 2-2
Overview of Suitable Study Parameters Used to Derive Toxicological Reference Values

Parameter	Suitable	Not Suitable
Test material	Test chemical (technical grade; known active ingredient) and mixtures of any combination of the 10 herbicides	Mixtures including non-BLM herbicides
Test species	Small mammals (rats, rabbits, mice), large mammals (dogs, sheep), small birds (quail), large bird (mallard), honey bee, coldwater and warmwater fishes, aquatic invertebrates, aquatic plants (including algae), and terrestrial plants	Human health effects, bioassays to cells, effects to target species, marine or estuarine species, microorganisms, blue green algae, and terrestrial insects (other than honeybee)
Exposure route	Dietary: drinking water (including oral and gavage) and dermal	Inhalation, injection (intravenous, inter-peritoneal), ocular irritation, skin irritation, and in vitro studies
Toxicity endpoints	Mortality, immobilization, growth, reproduction, germination, emergence	Metabolism, changes in blood parameters, changes in behavior, genotoxic effects, environmental fate and transport, leaching and analytical methods

2.3.1.2 Toxicity Reference Value Development

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

Endpoints for a receptor and routes of exposure were converted to the same units (e.g., milligrams per kilogram [mg/kg] BW) for comparison and selection of the appropriate TRV. Endpoints for aquatic receptors and terrestrial plants were reported based on exposure concentrations (milligrams per liter [mg/L] and pounds per acre [lbs/acre], 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). Table 2-3 summarizes animal BWs and feeding and drinking ingestion rates that were used to convert concentration endpoints to dose-based endpoints using the following general equation:

$$\text{Dose-based endpoint}_{\text{(mg/kg BW/day)}} = [\text{Concentration-based endpoint}_{\text{(mg/kg food)}} \text{ x food ingestion rate}_{\text{(kg food/day)}}]/\text{BW}_{\text{(kg)}}$$

Toxicity Reference Value Derivation

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

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

Use of the Uncertainty Factor

In some cases, TRVs had to be extrapolated from available toxicity data using an uncertainty factor. Based on a review of the application of uncertainty factors (Chapman et al. 1998), an uncertainty factor of 3 was considered to be appropriate for ecological TRV derivation in this document. This value was used to extrapolate TRVs when appropriate TRVs were not identified in the literature.

For example, a chronic LOAEL (e.g., 100 mg/kg BW) could be divided by an uncertainty factor of three to obtain an extrapolated chronic NOAEL TRV (e.g., 33 mg/kg BW).

2.3.2 Levels of Concern

To facilitate the translation of RQs into readily applicable estimates of risk, the calculated RQs were compared against LOCs used by the USEPA in screening the potential risk of pesticides (Table 2-4). These LOCs are used by the USEPA's OPP to analyze potential risk to non-target organisms and the need to consider regulatory action. Distinct USEPA LOCs are currently defined for the following risk presumption categories:

- **Acute high risk** – the potential for acute risk is high.

- **Acute restricted use -** the potential for acute risk is high, but may be mitigated.
- **Acute RTE species –** there is potential for risk to RTE species.
- **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. A "chronic RTE species" risk presumption category for aquatic animals was added for the 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 (less sensitive) LOCs were not necessary.

TABLE 2-3
Parameters Used to Convert Concentration-based Data to Dose-based Endpoints

Organism	Body Weight (kg)	Food Ingestion Rate ($kg_{\text{food wet weight}}$/day)	Water Intake Rate (L/day)
Mouse (*Peromyscus* spp.)	0.03	0.0055	0.0075
Rat	0.35	0.028	0.046
Rabbit	3.8	0.135	0.268
Dog (*Canis domesticus*)	12.7	0.301	0.652
Chicken	1.6	0.11	0.081
Mallard (*Anas platyrhynchos*)	1	0.1	0.059
Japanese quail (*Coturnix japonica*)	0.15	0.0169	0.017
Bobwhite quail (*Colinus virginianus*)	0.154	0.093	0.130
Ring-necked pheasant (*Phasianus colchicus*)	1	0.058	0.059

2.3.3 Rare, Threatened, and Endangered Species

To specifically address potential impacts to RTE species, two types of RQ evaluations were conducted. For RTE terrestrial plant species, the RQs for 'typical' and RTE species were calculated using different toxicity endpoints but keeping the same LOC (set at 1) for all scenarios. The plant toxicity endpoints were selected to provide extra protection to the RTE species. In the direct spray, spray drift, and wind erosion scenarios, the selected toxicity endpoints were an EC_{25} for "typical" species and a NOAEL for RTE species based on any relevant toxicity endpoint. In runoff scenarios, the highest and lowest germination based NOAECs (No Observed Adverse Effect Concentrations) were selected to evaluate exposure for typical and RTE species, respectively. Emergence data were used to address risks due to surface runoff when germination data were unavailable.

The evaluation of RTE terrestrial animals and aquatic species was addressed using a second type of RQ evaluation. The same toxicity endpoint was used for both typical and RTE species in all scenarios, but the LOC was lowered for RTE species (see Table 2-4).

2.4 Uncertainty Analysis

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

- *Limited toxicity data available for a given herbicide.* For some herbicides, the only toxicity data available may be those studies conducted as part of the USEPA pesticide registration process. In this case, chronic toxicity data may be limited or non-existent, and may not include sublethal studies of importance in particular to AE *4—Adverse indirect effects on the survival, growth, or reproduction of salmonids* (see Section 2.1). Where relevant studies existed, then this type of uncertainty was limited; however, where they did not exist, the uncertainties were discussed as thoroughly as possible.

- *The potential indirect effects of herbicides on RTE salmonids.* Unless actual field studies were identified for a given herbicide, this discussion was limited to only qualitative estimates of potential indirect impacts to salmonid populations and communities. Such qualitative estimates were limited to a general evaluation of the potential risks to food (typically represented by acute and/or chronic toxicity to aquatic invertebrates) and cover (typically represented by potential for destruction of riparian vegetation, or aquatic vegetation if appropriate). The USEPA OPP is using similar approaches for RTE species effects determinations and consultations.[5]

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

- *Ecological risks of inerts, adjuvants, and mixtures.* From an ecological point of view, it is desirable to estimate risks not just from the a.i. of an herbicide, but from the cumulative risks of all potentially harmful ingredients. However, deterministic risk calculations (e.g., exposure modeling, effects assessment, and RQ calculations) can only be conducted for the a.i. using currently available models (e.g., GLEAMS). An attempt was made to qualitatively estimate the potential additional risks (if any) posed by chemicals added to the a.i. of an herbicide, such as inerts (anything other than the a.i. in a product; not having active properties though still potentially toxic), adjuvants (chemicals used to enhance the pharmacological or toxic agent effect of the a.i.), and surfactants (a surface active agent that may be used to increase solubility; usually an organic compound whose molecules contain a hydrophilic [chemically attracted to water] group at one end and a lipophilic [chemically attracted to fats] group at the other), and degradates (chemicals created during the natural breakdown or decomposition of another chemical). The BLM received Confidential Business Information (CBI) clearance from the USEPA and reviewed the CBI data on all inert compounds. The toxicity evaluation is fully disclosed in each risk assessment report, although the names of specific chemicals cannot be disclosed.

Evaluating the potential additional/cumulative risks from mixtures of pesticides is substantially more difficult. While many pesticides are present in the natural environment along with other pesticides and toxic chemicals, it is extremely difficult to estimate the potential cumulative risks of such mixtures, particularly at the level of a programmatic EIS. The composition of such mixtures is highly site-specific, and thus nearly impossible to address at the programmatic level. However, the label information from each of the 10 herbicides mentions that most can be "tank mixed" with

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

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

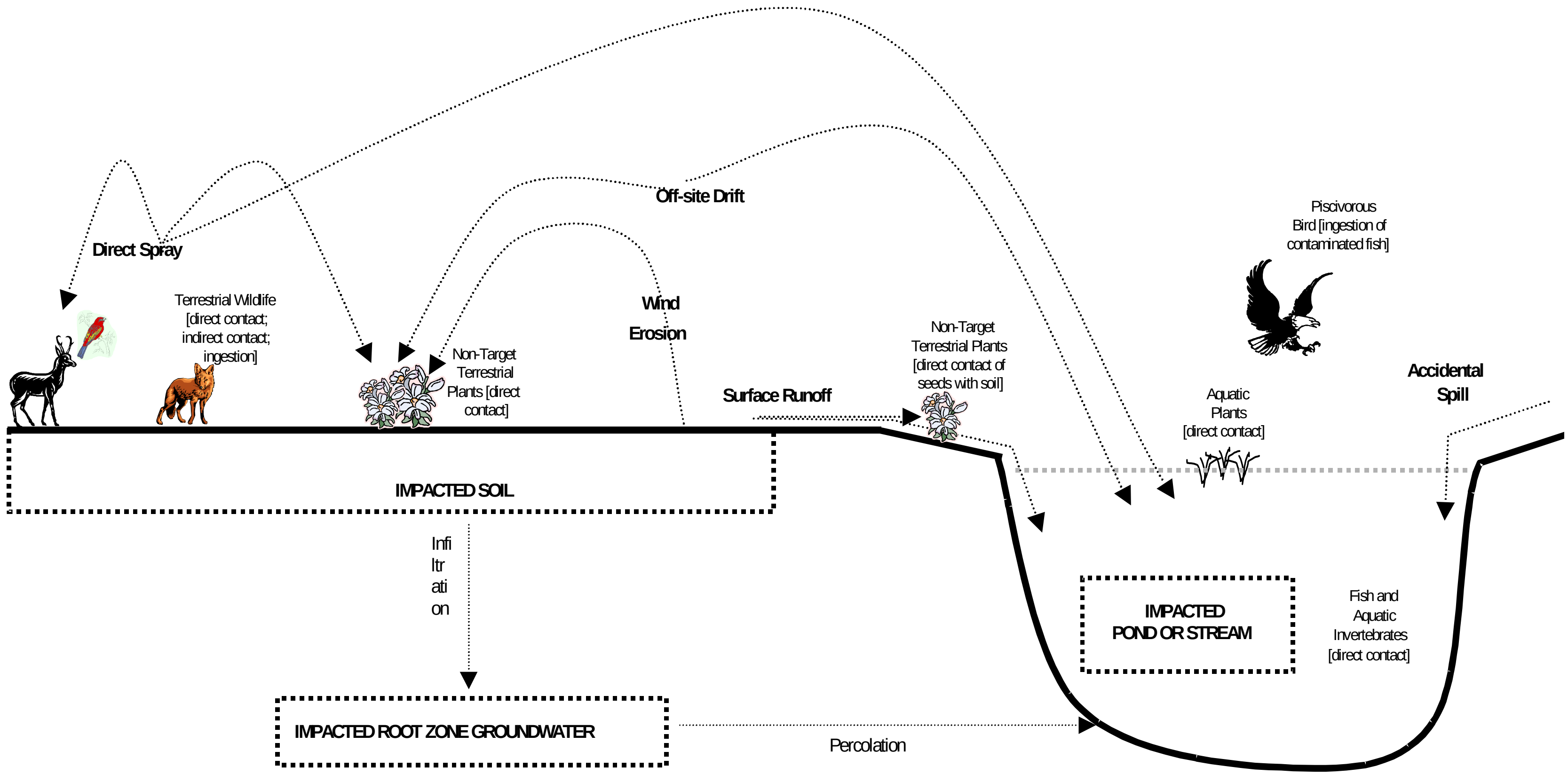
TABLE 2-4
Levels of Concern

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

[1] Estimated Environmental Concentration has units of mg $_{\text{prey wet weight}}$/kg $_{\text{BW}}$ for acute scenarios and mg $_{\text{prey wet weight}}$/kg $_{\text{BW}}$/day for chronic scenarios.
[2] Estimated Environmental Concentration has units of mg/L
[3] Estimated Environmental Concentration has units of lbs. a.i./acre

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

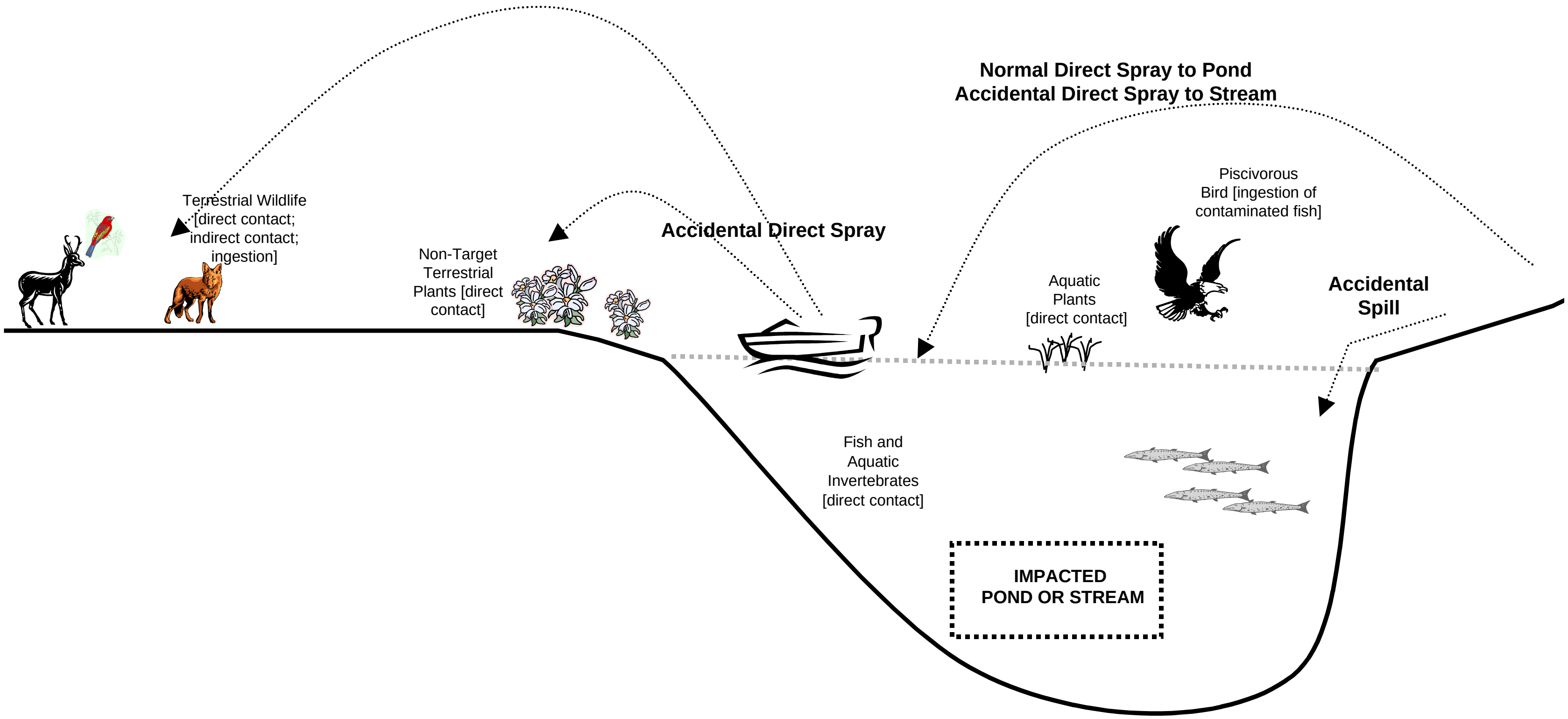
Figure 2-1 Conceptual Model – Terrestrial Herbicides



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

See Figure 2-3 for simplified food web & evaluated receptors.

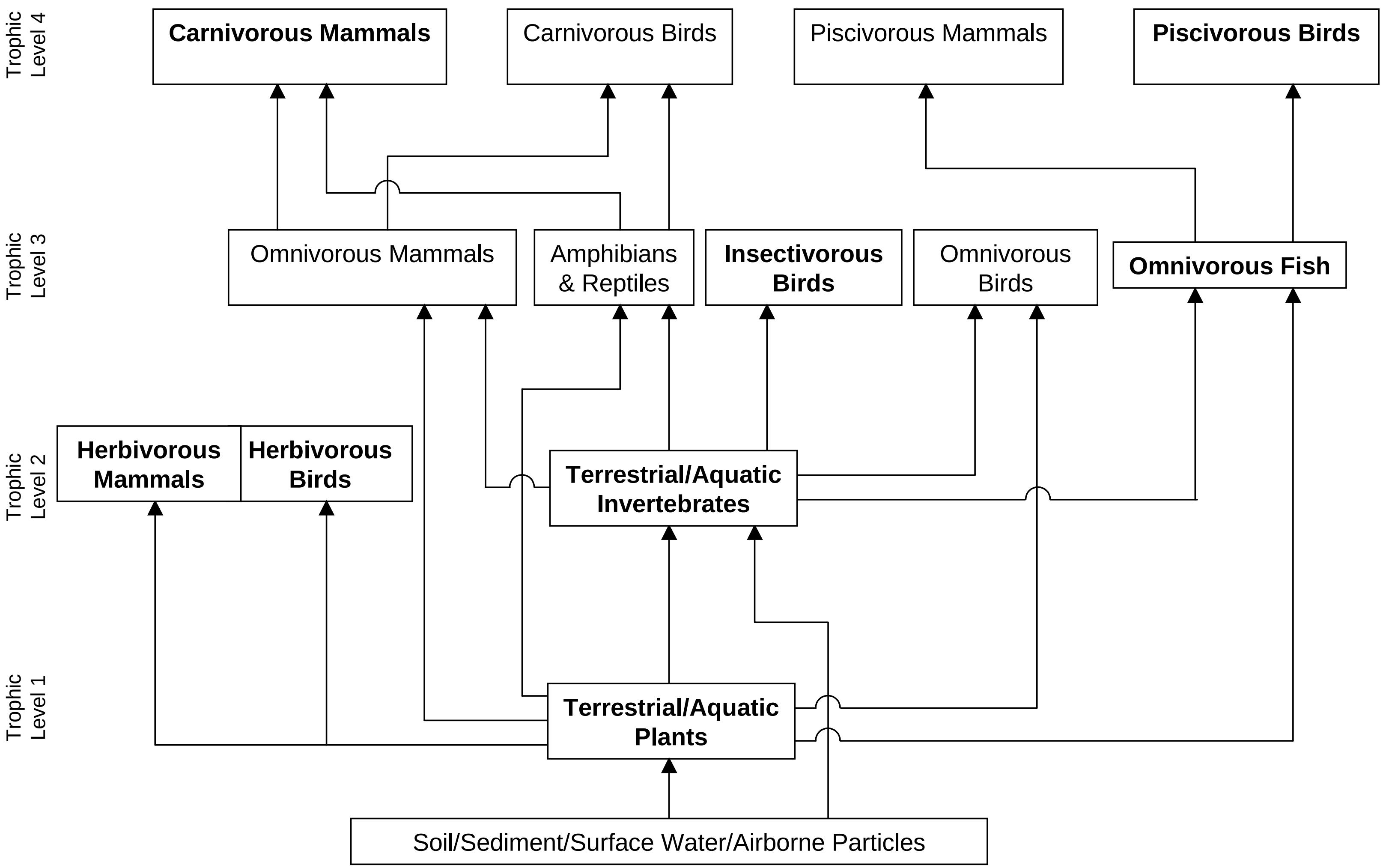
Figure 2-2 Conceptual Model – Aquatic Herbicides



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

See Figure 2-3 for simplified food web & evaluated receptors.

Figure 2-3 Trophic Levels Evaluated – Terrestrial and Aquatic Herbicides



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

3.0 EXPOSURE PATHWAYS AND DOSE CALCULATIONS

As described in Section 2.2.1, a number of exposure scenarios were developed to address potential acute and chronic impacts to receptors under a variety of exposure conditions that may occur within public lands. These exposure conditions include normal application situations, accidental spills, and associated off-site transport via spray drift, windblown dust, or surface runoff and root zone groundwater. In general, the exposure scenarios describe how a particular receptor group (e.g., terrestrial animals, terrestrial plants, aquatic plants) may be exposed to an herbicide in a complete exposure pathway. Exposure scenarios vary significantly between terrestrial and aquatic herbicides because off-site transport via drift, surface runoff/root zone groundwater discharge, and wind erosion are not realistic scenarios for the aquatic herbicides and were, therefore, not considered in the aquatic herbicide risk assessments. The selected pathways and relevant dose calculations are described in more detail in the following sections.

Ecological receptors, including terrestrial and aquatic species, were selected to address the potential risks due to unintended exposure to herbicides. A variety of generic terrestrial receptors were selected to cover the range of species and feeding guilds that might be found on public lands. Unless otherwise noted, receptor BWs were selected from the *Wildlife Exposure Factors Handbook* (USEPA 1993).This list represents general surrogate species; not all species will be present within each actual application area. The terrestrial animal receptors include the following:

- A pollinating insect with a BW of 0.093 grams (g; 0.003 ounces). The honeybee was selected as the surrogate species to represent pollinating insects. This BW was based on the estimated weight of receptors required for testing in 40 CFR 158.590.
- A small mammal with a BW of 20 g (0.7 ounce) that feeds on fruit (e.g., berries). The deer mouse (*Peromyscus maniculatus*) was selected as the surrogate species to represent small mammalian frugivores (i.e., fruit eaters).
- A large mammal with a BW of 70 kilograms (kg; 154 lbs) that feeds on grasses. The mule deer 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 (26.4 lbs) 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 (2.8 ounces) that feeds on insects. The American robin was selected as the surrogate species to represent small avian insectivores.
- A large bird with a BW of approximately 3.5 kg (7.7 lbs) that feeds on vegetation. The Canada goose (*Branta canadensis*) was selected as the surrogate species to represent large avian herbivores.
- A piscivorous bird with a BW of approximately 5 kg (11 lbs) feeding on fish in the pond. 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[6]).

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

In addition, potential impacts to non-target terrestrial plants were considered by evaluating two non-target plant receptors: the “typical” (i.e., non-RTE or common) species and the RTE species. Aquatic exposure pathways were evaluated using fish, aquatic invertebrates, and non-target aquatic plants for two types of generic aquatic habitat: 1) a small pond (¼-acre pond of 1 meter [m; 3.3 feet] depth, resulting in a volume of 1,011,715 L [267,295 gallons]); 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 (6.6 feet) wide and 0.2 m (0.66-foot) deep with a mean water velocity of approximately 0.3 m (1 foot) per second, resulting in a base flow discharge of 0.12 cubic meters (0.16 cubic yards) per second (cms).

3.1 Direct Spray

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

3.1.1 Exposure Scenarios Within the Application Area

These scenarios address potential impacts to non-target organisms within the area where the herbicide is being applied.

3.1.1.1 Direct Spray of Terrestrial Herbicide on Terrestrial Wildlife

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

The extent of exposure from direct spray of a receptor is based on three variables: the herbicide application rate; the surface area of the receptor species; and the rate of dermal absorption. Both typical and maximum herbicide application rates were evaluated for each herbicide for both the small mammal and the honeybee, or other more sensitive species. For each receptor it was assumed, based professional judgement, that exposure occurred over ½ the body surface (i.e., ½ the organism was covered by direct spray of the herbicide). The surface area calculation was obtained from the *Wildlife Exposure Factors Handbook* (USEPA 1993).

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

The dose calculation for 100% absorption over 24 hours for the small mammal and the honeybee is a function of the portion of the body that is sprayed (set at ½), a conversion factor (CF) of 0.01121 (mg/cm^2 [centimeters] per lb/acre), the application rate (R), the surface area of the receptor (A), and the body weight of the receptor (BW).

$$\text{Dose estimate}_{\,(mg/kg)} = [0.5 \times CF_{\,((mg/cm^2)/(lb/acre))} \times R_{\,(lb\ a.i./acre)} \times A_{\,(cm^2)}] \,/\, BW_{\,(kg)}$$

The first-order dermal absorption calculation was based on an evaluation presented in support of herbicide ERAs for the USDA Forest Service (SERA 2000). This evaluation found that first-order absorption rate coefficients (k_a) were best estimated based on both molecular weight (MW) and the log K_{ow} (partition coefficient):

$$\log k_a = 0.233255(\log K_{ow}) - 0.005657(MW) - 1.49615$$

The dose assuming first-order dermal absorption is a function of the portion of the body that is sprayed (set at ½), a conversion factor (CF) of 0.01121 (mg/cm^2 per lb/acre), the application rate (R), the surface area of the receptor (A), the proportion of herbicide that is absorbed (k_a), the duration of the exposure (T), and the body weight of the receptor (BW).

$$\text{Dose estimate}_{\,(mg/kg)} = [0.5 \times CF_{\,((mg/cm^2)/(lb/acre))} \times R_{\,(lb\ a.i./acre)} \times A_{\,(cm^2)} \times 1\text{-}e^{\,(-ka \times T\ (hours))}] / BW_{\,(kg)}$$

3.1.1.2 Indirect Contact with Foliage After Direct Spray of Terrestrial Herbicide

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

3.1.1.3 Ingestion of Prey Items Contaminated by Direct Spray of Terrestrial Herbicide

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

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

Herbivores and Insectivores

The estimated dose at Day 0 is a function of the amount of food consumed (F), application rate (R), the residue rate of the herbicide on the vegetation or insect (rr), and the body weight of the receptor (BW).

$$\text{Dose estimate}_{\,(mg/kg\ bw)} = [F_{\,(kg/day)} \times R_{\,(lb\ a.i./acre)} \times rr_{\,((mg/kg\ veg)\ /\ (lb\ a.i./acre))}] / BW_{\,(kg)}$$

The estimated dose at Day 90 is a function of the initial concentration of herbicide on the food item and the amount of herbicide remaining on the food item after 90 days. The initial concentration of herbicide (C_0) is a function of the application rate (R), the residue rate of the herbicide on the vegetation or insect (rr), and the proportion of the application rate that drifts onto the food item (D).

$$C_{0\ (mg/kg)} = R_{\ (lb\ a.i./acre)} \times rr_{\ ((mg/kg)\ /\ (lb\ a.i./acre))} \times D_{\ (unitless)}$$

A variable for decay (k) based on the foliar half-life of the herbicide is used to estimate the concentration of herbicide at Day 90.

$$k_{\ (days\text{-}1)} = \ln(2) \div t^{50}{}_{\ (days)}$$

The chronic exposure scenario uses a time-weighted average concentration (CTWA) to account for the decay of the herbicide over 90 days. The CTWA on vegetation or insects is a function of the initial concentration on the vegetation or insect (C_0), the decay coefficient (k; e.g., based on the foliar half-life), and the duration of exposure (T).

$$CTWA_{\ (mg/kg)} = C_{0\ (mg/kg)} \times (1\text{-}\exp(-\ k_{\ (days\text{-}1)} \times T_{\ (days)})) / (k_{\ (days\text{-}1)} \times T_{\ (days)})$$

The estimated dose at Day 90 is a function of the same variables described for the previous scenario, plus the surface area of the receptor (A) and the proportion of the diet that is contaminated (P).

$$\text{Dose estimate}_{\ (mg/kg\ bw)} = [CTWA_{\ (mg/kg)} \times A_{\ (kg/day)} \times P] / BW_{\ (kg)}$$

The drift proportion (D) and the contaminated diet proportion (P) were both conservatively assumed to be 100%.

Carnivores

The estimated dose at Day 0 is a function of the amount of food consumed (F), application rate (R), the amount of the herbicide deposited on the prey item (Amnt), and the body weight of the receptor (BW).

$$\text{Dose estimate}_{\ (mg/kg\ bw)} = [F_{\ (kg/day)} \times R_{\ (lb\ a.i./acre)} \times Amnt_{\ (mg\ a.i.)}] / BW_{\ (kg)}$$

The estimated dose at Day 90 is a function of the initial concentration of herbicide absorbed by the small mammal assuming 100% absorption over 24 hours ([Ci] see Section 3.1.1.1), the proportion absorbed by the prey item assuming first order absorption rate (PropAb), the amount of food consumed, and the body weight of the receptor (BW).

$$\text{Dose estimate}_{\ (mg/kg\ bw)} = [F_{\ (kg/day)} \times Ci_{\ (mg\ a.i.)} \times PropAb] / BW_{\ (kg)}$$

3.1.1.4 Direct Spray of Terrestrial Herbicide on Non-target Terrestrial Plants

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

$$\text{Dose estimate}_{\ (lb\ a.i./acre)} = R_{\ (lb\ a.i./acre)}$$

3.1.1.5 Direct Spray of Aquatic Herbicide onto Pond

The normal application of aquatic herbicides to a pond was considered to evaluate potential impacts to aquatic receptors other than the target plant species. For this scenario, the typical and maximum application rates (R) of the herbicides were applied directly to the pond, and the associated instantaneous water concentration was calculated based on the pond area, pond volume (VL), and a conversion factor (CF).

$$\text{Dose estimate}_{\ (mg\ a.i./L)} = [R_{\ (lb\ a.i./acre)} \times \text{Pond Area}_{\ (acre)} \times CF_{\ (mg/lb)}] / VL_{\ (L)}$$

Neither degradation nor sorption of the herbicide to sediments, aquatic vegetation, or suspended solids was considered, so this represents a conservative estimation of the concentration of herbicide in pond water. The pond water concentrations were compared against appropriate acute and chronic toxicity endpoints to evaluate potential impacts to fish, aquatic invertebrates, and non-target aquatic plants.

3.1.2 Exposure Scenarios Outside the Application Area

These scenarios evaluated the accidental direct spray of an herbicide outside of the original intended application area. Such impacts outside of the intended application area are accidental exposures that are not typical of BLM application practices.

3.1.2.1 Accidental Direct Spray

It is possible that terrestrial receptors may be accidentally sprayed during the normal application of the aquatic herbicide to ponds. The doses received by terrestrial receptors were calculated in the same manner as direct spray of terrestrial animals by a terrestrial herbicide (see Section 3.1.1.1).

3.1.2.2 Indirect Contact with Foliage After Direct Spray of Aquatic Herbicide

Indirect contact with foliage accidentally directly sprayed with an aquatic herbicide may also impact terrestrial animals. The doses received by terrestrial receptors were calculated in the same manner as indirect contact with foliage after direct spray of a terrestrial herbicide (see Section 3.1.1.2).

3.1.2.3 Ingestion of Prey Items Contaminated by Accidental Direct Spray of Aquatic Herbicide

Impacts to terrestrial wildlife receptors were also evaluated for the accidental direct spray of prey items during the normal application of an aquatic herbicide to ponds. The doses received by terrestrial receptors were calculated in the same manner as ingestion of prey items contaminated by direct spray of a terrestrial herbicide (see Section 3.1.1.3).

3.1.2.4 Accidental Direct Spray of Aquatic Herbicide on Non-target Terrestrial Plants

Non-target terrestrial plants may be accidentally sprayed during normal application of aquatic herbicide on ponds. The doses received by terrestrial receptors were calculated in the same manner as direct spray of a non-target terrestrial plant by a terrestrial herbicide (see Section 3.1.1.4).

3.1.2.5 Accidental Direct Spray of Terrestrial Herbicide over Pond

A pond surface may be accidentally sprayed during the normal application of a terrestrial herbicide. The typical and maximum application rates (R) of the herbicides were applied directly to the pond, and the associated instantaneous water concentrations were calculated based on the pond area, pond volume (VL), and a conversion factor (CF).

$$\text{Dose estimate}_{\text{(mg a.i./L)}} = [R_{\text{(lb a.i./acre)}} \times \text{Pond Area}_{\text{(acre)}} \times CF_{\text{(mg/lb)}}] / VL_{\text{(L)}}$$

Neither degradation nor sorption of the herbicide to sediments, aquatic vegetation, or suspended solids was considered, so this represents a conservative calculation of the pond water concentration. The pond water concentrations were compared against appropriate acute and chronic toxicity endpoints to evaluate potential impacts to fish, aquatic invertebrates, and non-target aquatic plants.

3.1.2.6 Accidental Direct Spray of Terrestrial or Aquatic Herbicide over Stream

Aquatic and terrestrial herbicides may be accidentally directly sprayed onto the surface of a stream. The typical and maximum application rates of the herbicides were applied directly to the stream segment (assumed to be adjacent to a 100 acre application area), and the associated water concentrations were calculated. Degradation and sorption of the herbicide and transport from flow of the stream were not considered, so this represents a conservative calculation of the stream water concentration (essentially an instantaneous concentration). The stream concentrations were compared against appropriate acute and chronic toxicity endpoints to evaluate potential impacts to fish, aquatic invertebrates, and non-target aquatic plants.

$$\text{Dose estimate}_{\text{(mg a.i./L)}} = [R_{\text{(lb a.i./acre)}} \times \text{Waterbody Area}_{\text{(acre)}} \times CF_{\text{(mg/lb)}}] / VL_{\text{(L)}}$$

3.2 Off-site Drift

During normal application of herbicides, it is possible for a portion of the herbicide to drift outside of the treatment area and deposit onto non-target receptors. To simulate off-site herbicide transport as spray drift, the AgDRIFT® model was used to evaluate a number of possible scenarios (see Section 4.1 and 4.2 of this document).

Ground applications were modeled using a low- or high-placed boom and aerial applications were modeled from either a helicopter or a plane. In addition, aerial applications were modeled at two different heights to mimic conditions that depend on whether the land being treated is forested or non-forested. Actual model runs depended on the specific BLM uses of each herbicide. Drift depositions were modeled at 25, 100, and 900 feet from the application area for each scenario. The AgDRIFT® model determined the fraction of the application rate that would be deposited on the off-site location without considering herbicide degradation.

3.2.1 Off-site Drift of Terrestrial Herbicide onto Plants

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

3.2.2 Off-site Drift of Terrestrial Herbicide onto Pond

As described previously, during normal application, is possible for a portion of the terrestrial herbicide to drift outside of the treatment area. This off-site drift may eventually reach a pond and contaminate the waterbody. AgDRIFT® was used to calculate pond water concentrations of the herbicide in the various application scenarios using the variables described in Section 3.2. The pond concentration calculated by AgDRIFT® does not consider herbicide degradation, sorption, or dissipation, and likely overestimates actual concentrations. AgDRIFT® does consider the dilution of the herbicide in the volume of the pond.

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

3.2.3 Off-site Drift of Terrestrial Herbicide onto Stream

As described previously, during normal application, is possible for a portion of the terrestrial herbicide to drift outside of the treatment area. This off-site drift may eventually reach a stream and contaminate the waterbody. AgDRIFT® was used to calculate in-stream concentrations of the herbicide in the various application scenarios using the variables described in Section 3.1. The stream concentration calculated by AgDRIFT® does not consider herbicide degradation sorption, or dissipation, and likely overestimates actual concentrations. The rate of deposition estimated by AgDRIFT® was diluted into the stream based on the flow rate (0.12 cms) described in Section 3.0.

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

3.2.4 Consumption of Fish from Pond Contaminated by Drift of Terrestrial Herbicide

Off-site drift of herbicide may eventually reach the off-site ponds and contaminate the resident fish population, which may be consumed by piscivorous bird species. In this scenario, impacted pond water is modeled using the AgDRIFT®

inputs described above. The pond concentration calculated by AgDRIFT® does not consider herbicide degradation, sorption, or dissipation, and likely overestimates actual concentrations. Exposure for the piscivorous bird was evaluated by modeling fish tissue concentrations from pond surface water (C_{pond}) using bioconcentration factors (BCFs) and food chain multipliers (FCMs) for different TLs. Food chain multipliers assumed a TL3 for fish and a TL2 for the prey of fish (e.g., aquatic invertebrates). Food chain multipliers were obtained from USEPA (1995a).

$$C_{\text{fish (mg a.i./kg fish)}} = C_{\text{pond (mg a.i./L)}} \times BCF_{\text{(L/kg fish)}} \times FCM_{TL2} \times FCM_{TL3}$$

The calculated dose to the piscivorous bird is a function of the concentration in the fish tissue (C_{fish}), the food ingestion rate in wet weight (FIR_{ww}), the proportion of the diet that is contaminated ([P] assumed to be 100%), and the body weight of the bird (BW).

$$\text{Dose estimate}_{\text{(mg a.i./kg-day)}} = [C_{\text{fish (mg a.i./kg fish)}} \times FIR_{\text{ww (kg ww/day)}} \times P] \,/\, BW_{\text{(kg)}}$$

The dose estimate to the piscivorous bird was compared to appropriate chronic toxicity values.

3.3 Surface Runoff

Precipitation may result in the transport of herbicide applied to soils from the application area via surface runoff and root-zone groundwater flow. This transport to off-site soils or waterbodies was modeled using GLEAMS software (see Section 4.2 and Appendix B for GLEAMS details and assumptions). Model variables include soil type, annual precipitation, size of application area, hydraulic slope, surface roughness, and vegetation type. The range of values for each variable used in the GLEAMS simulations was selected to bracket a large possible range of conditions, and values were not selected to predict concentrations at any particular site. These variables were altered to predict soil concentrations of the herbicides in various watershed types at both the typical and maximum application rates.

The surface runoff scenarios are not considered relevant for the aquatic herbicides.

3.3.1 Surface Runoff of Terrestrial Herbicide to Off-site Soils

To evaluate the potential impact of surface runoff on off-site non-target plants for each herbicide, the GLEAMS model was configured to provide 52, 7-day average loadings from the application area for the final year of the GLEAMS run (when the model reaches a quasi-steady state). The maximum of the 7-day average loadings calculated by GLEAMS was assumed to affect a soil area immediately downslope of the application area. The loading was expressed as a proportion of the total herbicide loading to the application area. For example, if 30% of the applied herbicide was found to run off, the soil concentration off-site was predicted to be 30% of that in the application area. These off-site soil concentrations were compared against appropriate toxicity endpoints to determine potential impacts to typical and RTE non-target plants. This particular exposure pathway (exposure in the root zone rather than foliar deposition) may impact seed germination. Toxicity data relevant to seed germination, a sensitive endpoint, were used for evaluation.

3.3.2 Surface Runoff of Terrestrial Herbicide to Off-site Pond

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

The GLEAMS exports were used to calculate two pond water concentrations for comparison against acute and chronic toxicity endpoints for the aquatic receptor species. To estimate potential acute exposure, the maximum 3-day average herbicide export rate from the final year of the GLEAMS run was used (the year when the model reaches a quasi-steady state). To estimate potential chronic exposure, the overall average herbicide export rate from the final year of the GLEAMS run was used. The surface water concentrations in the pond calculated from these two export rates were compared against the appropriate acute and chronic toxicity endpoints for each of the three aquatic receptors.

3.3.3 Surface Runoff of Terrestrial Herbicide to Off-site Stream

As described previously, precipitation may result in the transport of terrestrial herbicides via surface runoff and root-zone groundwater flow. This flow of herbicide bound to soil in runoff and root zone groundwater may eventually reach an off-site stream resulting in the contamination of the waterbody. The GLEAMS model daily predictions of herbicide export rates were used to calculate stream water concentrations of the herbicide in the various watershed scenarios.

The GLEAMS exports were used to calculate two stream concentrations for comparison against acute and chronic toxicity endpoints for the aquatic receptor species. To estimate potential acute exposure, the maximum 3-day average herbicide export rate from the final year of the GLEAMS run was used (the year when the model reaches a quasi-steady state). To estimate potential chronic exposure, the overall average herbicide export rate from the final year of the GLEAMS model run was used. The stream water concentrations calculated from these two export rates were compared against the appropriate acute and chronic toxicity endpoints for each of the three aquatic receptors.

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

Surface runoff containing herbicide bound to soil may eventually reach an off-site pond, and resident fish may accumulate herbicide. The fish may in turn be consumed by piscivorous bird species. In this scenario, impacted pond water was modeled using the GLEAMS model described above. Because bioaccumulation is a long-term process, the chronic exposure concentration (i.e., the overall average concentration from the final year of the GLEAMS run) was used to predict fish tissue concentrations. A BCF and FCM for different TLs were included in the estimate. Food chain multipliers assumed a TL3 for fish and a TL2 for the prey of fish (e.g., aquatic invertebrates). Food chain multipliers were obtained from USEPA (1995a).

$$C_{\text{ fish (mg a.i./kg fish)}} = C_{\text{ pond (mg a.i./L)}} \times BCF_{\text{ (L/kg fish)}} \times FCM_{TL3} \times FCM_{TL2}$$

The calculated dose to the piscivorous bird is a function of the concentration in the fish tissue (C_{fish}), the food ingestion rate in wet weight (FIR_{ww}), the proportion of the diet that is contaminated ([P] assumed to be 100%), and the body weight of the bird (BW).

$$\text{Dose estimate}_{\text{ (mg a.i./kg-day)}} = [C_{\text{ fish (mg a.i./kg fish)}} \times FIR_{ww\text{ (kg ww/day)}} \times P] / BW_{\text{ (kg)}}$$

3.4 Wind Erosion and Transport Off-site of Terrestrial Herbicide

Dry conditions and wind may also allow transport of herbicide from the application area as wind-blown dust onto non-target plants some distance away. This transport due to wind erosion of the surface soil was modeled using CALPUFF software (see Section 4.3 for CALPUFF details and assumptions). Five distinct watersheds were modeled using CALPUFF to determine herbicide concentrations in dust assumed to deposit on plants. The concentrations were modeled after a wind event, with dust deposition estimates calculated at distances ranging from 1.5 to 100 km (0.9 to 62 miles) from the application area. At each radius considered, the maximum predicted rate of herbicide deposition in a given wind event was calculated. The dust estimates calculated within the model were then compared against the appropriate non-target plant toxicity values.

This scenario was considered relevant for the terrestrial herbicides only. The dust exposure scenario was not considered for aquatic herbicides.

3.5 Accidental Spill of Terrestrial Herbicide to Pond

Two spill scenarios were modeled to represent worst-case potential impacts to the pond. The scenarios included a helicopter or a truck spilling entire loads (140-gallon spill and 200-gallon spill, respectively) of herbicide mixed for the maximum application rate into the ¼ acre, 1 m (3.3 feet) deep pond. These volumes represent the typical load sizes used on BLM lands. To represent an acute exposure event for the three types of aquatic receptors, the pond concentration was compared against the appropriate toxicity endpoint for the fish, aquatic invertebrates, and non-target aquatic plants.

The concentration of herbicide in the pond water is based on the concentration in the spilled solution (C), the volume spilled (VS), and the volume of the pond (VL), and a conversion factor (CF) assuming instantaneous mixing.

$$\text{Concentration in pond (mg/L)} = [VS_{\text{(gal)}} \times CF_{\text{(L/gal)}} \times C_{\text{(mg/L)}}] / VL_{\text{(liters)}}$$

4.0 MODELS USED TO PREDICT MEDIA CONCENTRATIONS OF HERBICIDE

As described in Section 2.2.1, exposure characterization depends on the selection of appropriate fate and transport models for predicting herbicide concentrations in various environmental media (e.g., soils, water). Three computer models were selected to calculate the more complex processes. Concentrations used in aerial deposition calculations were predicted by AgDRIFT®, herbicide transport in surface runoff and root-zone groundwater flow was predicted by GLEAMS, and transport of herbicide sorbed to windblown dust was predicted by CALPUFF.

The following sections present general overviews of the three models; more detailed information is presented in Appendix A for AgDRIFT, Appendix B for GLEAMS, and in the *Vegetation Treatment Programmatic EIS Air Quality Impact Assessment Protocol* (ENSR 2004) for CALPUFF.

4.1 AgDRIFT®

Off-site spray drift and resulting terrestrial deposition rates and waterbody (pond and stream) concentrations were predicted using the computer model, AgDRIFT® (Spray Drift Task Force [SDTF] 2002).

AgDRIFT® Version 2.0.05 (SDTF 2002) is a computer model that is a product of the Cooperative Research and Development Agreement between the USEPA's Office of Research and Development and the SDTF (a coalition of pesticide registrants). It is based on, and represents an enhancement of, the computer program for agricultural dispersion (AGDISP [agricultural dispersion model]). AGDISP was developed by the National Aeronautics and Space Administration, the USDA Forest Service, and the U.S. Army. AgDRIFT® was developed for use in regulatory assessments of off-target drift associated with agricultural use of pesticides through aerial, ground, or orchard/airblast applications. AgDRIFT® is based on the idea that pesticide or herbicide drift is primarily a function of application technique (e.g., droplet size, release height), environmental conditions, and physical properties of the spray solution, and is not a function of the chemical properties of the a.i. itself. The computational approach employed by AgDRIFT® is based on a method that has evolved over a period of more than 20 years and yields high correlation with field measurement data sets. The model was selected for use in this risk assessment because of its existing use in regulatory assessments of off-target drift and its suitability to this particular application.

AgDRIFT® enables the user to take a tiered approach to the modeling of drift by allowing the user to choose between three tiers of increasingly complex evaluations of off-target drift and deposition. The basic difference between the three tiers (I, II, and III) is the number of model input variables the users can change. Further, Tier I supports the evaluation of aerial and ground application scenarios, whereas Tiers II and III support the evaluation of only aerial application scenarios (for agricultural and forestry applications).

Tier I is based on a set of standard "Good Application Practices," requires little knowledge of the actual application conditions or herbicide properties, and allows the user to modify a small number of model variables. Tiers II and III are based on the same set of "Good Application Practices" as Tier I. However, to implement either Tier II or III the user must have a progressively greater knowledge of the specific conditions under which herbicides will be applied. These tiers allow the user to modify variables to make the scenario evaluated representative of these conditions. Tier I was used in the PEIS to evaluate off-target drift associated with ground application scenarios. Ground applications may be conducted 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), and deposition rates vary by the height of the boom (the greater the height of the spray boom, the greater the off-target drift). Tier II was used to evaluate off-target drift associated with agriculture-like (e.g., rangeland) and forestry application scenarios in the PEIS risk assessment. The implementation of the Tier I ground and Tier II aerial application model and the model input variables (including the variables specific to the application method and environmental setting and specific to the herbicide being evaluated) are discussed and presented in Appendix A.

4.1.1 Terrestrial and Pond Herbicide Concentrations

The concentrations resulting from the terrestrial deposition rates predicted using the AgDRIFT® Tier II model were used to evaluate the potential risk to off-site non-target terrestrial plants. The concentrations resulting from the deposition onto the pond were calculated using the deposition rates predicted using the AgDRIFT® Tier II model and the pond volume.

4.1.2 Instream Dilution of Herbicide

The stream herbicide concentrations predicted using AgDRIFT® represent static conditions (similar to pond scenarios). Exposure in the stream, however, will change instantaneously due to downstream flow and movement of the herbicide. For modeling purposes, the length of the immediately affected stream reach is 636 m (2,087 feet). The width and depth of the stream channel are 2 m (6.6 feet) and 0.2 m (0.7 feet), respectively; the flow rate is 0.12 m^3 (0.16 cubic yard)/sec and the flow velocity is 0.3 m (1 foot)/sec, as indicated in the risk assessment protocol. Because the airspeed of the plane spraying the herbicide is assumed to be 120 mph, covering 20 passes takes approximately 4 minutes and for all intents and purposes is assumed to occur instantaneously. Therefore, the volume of water that receives the herbicide deposition can be approximated as the spray length multiplied by the product of the stream width and depth. This results in a stream volume of 254 m^3 (332 cubic yards) that, at a velocity of 0.3 m (1 foot)/sec, would pass by a downstream receptor over approximately 2,120 seconds (0.58 hours).

$$\text{Volume of Stream Receiving Herbicide from Drift} = 636_{\ (m)} \times 2_{\ (m)} \times 0.2_{\ (m)} = 254_{\ (m^3)}$$

$$\text{Amount of Time for Water to Flow Through Affected Area of the Stream} = 636_{\ (m)} / 0.3_{\ (m/sec)} = 2{,}120_{\ (sec)} = 0.58_{\ (hrs)}$$

Realistically, an instream organism will have an exposure duration greater than 0.58 hours. Therefore, the maximum concentration was averaged over a volume of water equivalent to the amount that would pass by a downstream receptor in 3 hours (a conservative estimate of acute exposure duration of an instream organism). To determine the dilution offered by this increased volume, the volume over 3 hours was divided by the initial slug volume, resulting in a net dilution ratio of 5.1.

$$\text{Dilution} = \text{Volume}_{\ (\text{exposure time})} / \text{Volume}_{\ (\text{Stream Water})}$$

or

$$\text{Dilution} = (\text{Flow Rate} \times 3_{\ (hrs)}) / (\text{Length}_{\ (m)} \times \text{Width}_{\ (m)} \times \text{Depth}_{\ (m)})$$

or

$$\text{Dilution} = (0.12\ (m^3/sec) \times 3_{\ (hrs)} \times 3600_{\ (sec/hr)}) / (636_{\ (m)} \times 2_{\ (m)} \times 0.2_{\ (m)}) = 5.1$$

4.2 GLEAMS

GLEAMS was used in this risk assessment to calculate soil concentrations at the site of application, transport of herbicides to adjacent soils, and the amount of herbicide that might runoff into aquatic habitats (e.g., ponds, streams). One benefit of GLEAMS is the ability to estimate a wide range of potential herbicide exposure concentrations as a function of site-specific parameters, such as soil characteristics and annual precipitation. The following subsections present a general overview of the GLEAMS model and the calculation of media concentrations in a variety of different exposure scenarios. A more detailed discussion is presented in Appendix B. Each herbicide risk assessment contains an herbicide-specific description of the model outputs.

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

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

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

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

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

4.2.1 Data Requirements

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

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

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

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

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

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

4.2.2 GLEAMS Model Scenarios

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

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

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

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

The GLEAMS model predicts daily herbicide export rates. As a result of the conservative assumptions used in the model, it is likely that the export rates predicted by GLEAMS are higher than actual rates. This hypothesis of over-prediction is substantiated by the comparison of the GLEAMS export rates modeled here to measured data presented by Lerch and Blanchard (2003). GLEAMS-predicted export rates were higher than measured rates. Details of this comparison are presented in Appendix B.

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

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

4.3 CALPUFF

One of the exposure scenarios in the ERA is the potential migration of the herbicide from the area of application by windblown soil (fugitive dust). The USEPA's guideline air quality California Puff (CALPUFF) air pollutant dispersion model, (referenced in Appendix W of 40 CFR Part 51) was used to predict potential impacts of herbicide particulate matter (i.e., total suspended particulates ranging between 0.1 and 50 micrometers (μm) in diameter [TSP], particulate matter 2.5 μm in diameter and smaller [$PM_{2.5}$], and particulate matter 10 μm in diameter and smaller [PM_{10}]) on receptors located between 1.5 and 100 kilometers (km; 0.9 and 62 miles) from the assumed emission locations (the center of the application site). CALPUFF "lite" version 5.7 was selected because of its ability to screen potential air quality impacts within and beyond 50 km (31 miles) and its ability to simulate plume trajectory over several hours of transport based on limited meteorological data. The details of the CALPUFF model are further described in the *Vegetation Treatment Programmatic EIS Air Quality Impact Assessment Protocol* (ENSR 2004).

4.3.1 Source Characterization

A high wind event and associated wind erosion may cause the surface soil (with the herbicide) to migrate from the application area. It was assumed that all of the applied herbicide was adsorbed by the top 1 mm (0.039 inch) of soil.

This assumption determines the rate of herbicide deposition as a function of the rate of dust deposition at the downstream receptor location. The depth of 1 mm is believed to be conservative, and this depth is less that that assumed by others (e.g., SERA [2003] assumed 1 cm [0.39 inch]). The modeling assumed a square, flat area of 1,000 acres was treated with herbicide applied from the air using a fixed-wing aircraft. For suspended particulate matter, modeled impacts were directly proportional to the modeled emission rate. Therefore, the modeling assumed a unit rate of chemical application/deposition (1.0 g per m^2 [0.035 ounces per 1.2 square yards). For varying application rates to bare, undisturbed soil (if converted to the same units), the model results can be scaled directly. The modeling results were expressed as the fractional downwind deposition based on this initial application.

4.3.2 Determination of Wind Erosion Event

The CALPUFF model was used to estimate acute exposures. The maximum 1-hour and 24-hour, as well as annual average deposition rates from a conservative impact migration event (i.e., an event modeled using very conservative properties such that the potential for dust to move is high) were computed for the distance ranges being modeled (1.5 to 100 km [0.9 to 62 miles] from the center of the 1,000 acre application site). Although a given area would be sprayed with herbicide only once per year, a full year was modeled to consider a large range of meteorological conditions that could influence the herbicide migration potential for that single event. The highest impact was considered to represent a "reasonable, but conservative" impact under the range of the meteorological conditions tested. The modeling conservatively assumed that the herbicide application (and subsequent migration) could occur any day of the year, should meteorological conditions trigger a wind erosion event (except as noted below).

The herbicide was assumed to adhere to undisturbed surface soil, which can be picked up and transported by sufficiently high winds. The threshold wind speed for such an event is linked to the "friction velocity," which is a measure of the mechanical turbulence at the soil-atmosphere interface, and thus is a good gauge of the ability of the wind to pick up surface particles. The friction velocity is the square root of the surface shearing stress divided by the air density (a quantity with units of wind speed), and the surface shearing stress is related to the vertical transfer of momentum from the air to the earth's surface. Shearing stress, and therefore friction velocity, increases with increasing wind speed and surface roughness. Shearing stress and friction velocity are also nearly constant with height near zero (roughly lower than the height of the surface roughness elements). Generally, for areas with average roughness, the friction velocity is on the order of one-tenth of the 10-m wind speed.

Threshold friction velocities for undisturbed soils were determined from Gillette (1988), as described in the *Vegetation Treatments Programmatic EIS Air Quality Impact Assessment Protocol* (ENSR 2004). The BLM (Ypsilantis 2003) identified appropriate soil types for each of the "example" modeling analysis locations, as discussed in more details in this protocol. When the friction velocity (provided as an output from PCRAMMET) is above the threshold friction velocity, off-site migration of surface soil particles to which herbicide has adhered could occur, and is modeled to occur for a potential migration event.

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

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

- Wet soil adjustment – assume no hourly particulate matter emissions when there is measurable hourly precipitation (at least 0.01 inch).

- Frozen soil adjustment – assume no hourly particulate matter emissions when the hourly ambient temperature is at/below 28 ºFahrenheit.

- Snow cover adjustment – assume no hourly particulate matter emissions when the hourly snow depth is at least 1 inch.

- Operational adjustment – assume only one application of chemical herbicide per given 1,000-acre location in the same year.

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

4.3.3 Determination of Herbicide Emission Rates

The initial incorporation depth of herbicide, applied as an equally distributed aerosol over an application area, was assumed to be 1 mm (0.039 inch); the applied herbicide was therefore assumed to adhere to the upper 1 mm of soil. This depth determines the concentration of herbicide on eroded dust and defines the depth of erosion at which the mass of herbicide would be exhausted. This incorporation depth (also known as a mixing depth) is based on fast-acting physical processes and does not include leaching of herbicide into the soil as a result of precipitation. The mixed layer depth is estimated to account for three processes:

- Settling of the applied herbicide at different depths relative to a given elevation as a result of an uneven soil surface

- Preferential erosion of fine-grained soils by the wind resulting in segregation of soil particles and essentially mixing of the surface layer

- Physical infiltration of the herbicide into the soil (this is likely a minor factor as little herbicide volume is available to drive infiltration)

A 2-mm (0.079 inch) incorporation depth has been previously assumed for herbicides that have high adsorptivity and low leachability (Haney et al. 2002). For groundwater modeling applications, the GLEAMS model assumes a 1-cm (0.37 inch) surface-soil-layer thickness or mixing depth (Leonard et al. 1987). Because thinner affected soil depths result in elevated herbicide emissions during fugitive dust events, assuming an incorporation/mixing depth of 1 mm is highly conservative.

It was also assumed that there is an even distribution of the herbicide across the soils, and that the mass of the herbicide is negligible compared to the mass of the soil. Given a typical soil density of 1 g/cm^3 (0.035 ounce/0.06 in^3) the mass of a 1-mm depth of soil occupying a m^2 is 1,000 g (35 ounces). This represents the total mass of soil per m^2 that may be removed by the wind before all of the herbicide has been re-suspended. Using the meteorological data for each site and emission source equation, the mass of soil removed by the wind was calculated for every hour that herbicide re-suspension is possible. The fraction of the herbicide applied to the area that could be released was determined by dividing the mass of soil removed per m^2 by 1,000 g (per m^2) for each hour. This percentage was applied at each herbicide's maximum application rate. The resulting value represented the amount of herbicide potentially released each hour, which was assigned to each of the three particle sizes ($PM_{2.5}$, PM_{10}, and TSP) to estimate potential herbicide deposition.

4.3.4 Calculation of Herbicide Deposition

Air quality modeling focuses upon the emissions and subsequent concentrations of soil particulate matter deposited as a result of high wind events. Particle size distribution affects deposition velocities; therefore, CALPUFF predicted deposition of windblown particulate matter to compute the total potential soil particulate-matter deposition at each downwind receptor.

Both the hourly particulate-matter emission rate, and the particle-size-mass distribution at each location followed the guidance provided in the USEPA's AP-42 (USEPA 1995b), including the aerodynamic-particle-size multipliers (adjustment factors): TSP = 1.0, PM_{10} = 0.5, and $PM_{2.5}$ = 0.2. The actual mass distribution of adsorbed pesticide material is related to the surface area of each particle size category, with small particles having a larger surface area relative to their mass than large particles, and therefore, carrying the majority of the herbicide mass, as shown in Table 4-1.

The deposition algorithm in CALPUFF simulated the effective mass distribution of the adsorbed herbicide, based on particulate matter size. The model accounted for the effect of deposition in removing particulate matter from the plume as it moves downwind. Dispersion modeling estimated the maximum 1-day and 30-day deposition values at each receptor distance. The results were scaled for typical and maximum application rates and were applied to the risk assessment analysis.

TABLE 4-1
Mass Distribution of Particles for Adsorbed Herbicide

Parameter	Fine Particles	Small Particles	Large Particles
Size category (diameter)	$PM_{2.5}$	PM_{10}	TSP
Assumed mean diameter (micrometer [µm])	2	5	20
Mass (or volume) fraction	0.2	0.3	0.5
Assumed surface area per particle (μm^2)	13	79	1,257
Mass (microgram [µg]) per particle (assuming density of 1 g/ cm^3)	$4.2E^{-6}$	$6.5E^{-5}$	$4.2E^{-3}$
Number of particles per µg of suspended soil (accounting for mass fraction)	$4.8E^{+4}$	$4.6E^{+3}$	119
Total surface area (μm^2) available per µg of suspended soil	$6.0E^{+5}$	$3.6E^{+5}$	$1.5E^{+5}$
Assumed chemical herbicide deposition fraction of total modeled deposition	0.54	0.32	0.14

4.3.5 Watershed Evaluated

Three watersheds were used in the simulation:

- 1990 MT94008 Glasgow/International Airport (Glasgow, Montana)
- 1990 OR24225 Medford/Jackson County Airport (Medford, Oregon)
- 1990 WY24021 Lander/Hunt Field (Lander, Wyoming)

These locations were selected as representative of various regions of the western states addressed by the *Vegetation Treatments Programmatic EIS*. Because it is assumed that no chemical treatment will be applied to vegetation in Alaska under each of the five treatment alternatives, Alaska was not included in the CALPUFF modeling analysis (ENSR 2004).

For each location included in the model, 1 year of surface meteorological data from the Solar and Meteorological Surface Observation Network (SAMSON) data set that has been produced by National Climatic Data Center was used.[7] After a review of available data capture, the most recent SAMSON year with complete surface and mixing height data was selected for each station. The SAMSON data set is particularly applicable for CALPUFF modeling because it contains hourly values of relative humidity and solar radiation, which are needed for chemical transformation calculations. Mixing height data for these sites were obtained from the USEPA's "Technology Transfer Network Support Center for Regulatory Air Models." [8] The highest impact was considered to represent a "reasonable, but conservative" impact under the range of meteorological conditions tested.

The CALPUFF model was run in a screening mode, where meteorological conditions are assumed to vary from hour-to-hour, but are uniform throughout the modeling domain within each hour. In addition, because specific treatment locations are unknown, the terrain is assumed to have no meteorological influences (i.e., it is assumed to be "flat").

Further details about the CALPUFF model inputs and assumptions can be found in the *Vegetation Treatments Programmatic EIS Air Quality Impact Assessment Protocol* (ENSR 2004).

[7] http://ols.nndc.noaa.gov/plolstore/plsql/olstore.prodspecific?prodnum=c00066-CDR-S0001

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

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