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Ecological and Wildlife Risk Assessment of Chemical Use in Vegetation Management on Electric Utility Rights-of-Way

Technical Report

Ecological and Wildlife Risk Assessment of Chemical Use in Vegetation Management on Electric Utility Rights-of-Way

1009445

Interim Report, December 2004

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This report describes research sponsored by EPRI.

The report is a corporate document that should be cited in the literature in the following manner:

Ecological and Wildlife Risk Assessment of Chemical Use in Vegetation Management on Electric Utility Rights-of-Way, EPRI, Palo Alto, CA: 2004. 1009445.

REPORT SUMMARY

The management of vegetation on electric utility rights-of-way (ROWs) is an essential part of managing electrical transmission and distribution systems. A variety of manual, mechanical, and chemical methods, singly or in combination, are used for this purpose. The method or methods selected must be safe for humans and the environment and cost-effective in accomplishing the goals of ROW management. This report reviews environmental and wildlife safety through an assessment of risk to the environment, including air, water, and soil pollution as well as effects on terrestrial, avian, and aquatic wildlife.

Background

ROW management for humans and the environment focuses first and foremost on not causing unacceptable adverse effects. Risk assessment is the process by which the likelihood of unacceptable adverse effects can be determined. Risks to human safety were reviewed in a previous EPRI report, Human Health Risk Assessment of Chemicals Encountered in Vegetation Management on Electric Utility Rights-of-Way (1005367, November 2003). EPRI sponsored this research as part of its program to provide balanced, cost-effective solutions for economic and environmental challenges associated with ROW management.

Objectives

- To familiarize readers with basic concepts and methods used in environmental risk assessment.
- To summarize the behavior of each chemical in the environment, as it relates to environmental and wildlife risk.
- To describe the way each chemical causes adverse affects on wildlife (chemical toxicity).
- To identify the highest known doses of each chemical that causes no adverse effect in the most sensitive test in the most sensitive organism.
- To quantify the exposure to chemicals that broad groups of wildlife species may receive due to the use of herbicides for vegetation management on electric utility ROWs.
- To quantify the wildlife risk due to chemical exposure.

Approach

This report is based on a variety of major reviews developed by or for the U.S. Department of Agriculture (USDA) Forest Service; U.S. Environmental Protection Agency (EPA); U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry; and California Department of Pesticide Registration. In addition, electronic searches identified specific literature or other sources of information that may not have been included in these

reviews. Investigators used these various sources to identify critical toxicity values and to estimate exposure levels for each chemical included in the report. In calculating the level of wildlife and environmental effects, investigators employed toxicity values with the exposure values, as reflected by the margin of safety.

Results

In most cases, the margins of safety are adequate to ensure protection of terrestrial, avian, and aquatic wildlife. For two herbicides (2,4-D and triclopyr esters), maximum exposure and the results of major spills and accidents cause unacceptable levels of risk. Accidental spills are the most serious of these. Picloram has the greatest potential to cause environmental risk because of its persistence and mobility in soil, leading to potential groundwater contamination.

The picture is less clear for some herbicide carriers, bar oil, and hydraulic fluid because of the limited amount of information about these chemicals. In a regulatory framework, they are not accorded the same level of attention as pesticides, which must go through a rigorous EPA registration process.

This report provides suggestions for mitigation of inadequate margins of safety, emphasizing tactics for minimizing exposure. Effective education, supervision, project planning, and execution are essential in minimizing the potential for adverse ecological and wildlife effects.

EPRI Perspective

Today's energy companies – under pressure to reduce O&M costs while increasing power line throughput – need more cost-effective methods for managing existing ROWs. This report evaluates the cost-effectiveness of vegetation management practices, particularly the impact of chemicals on wildlife, along with means of minimizing ecological effects. Utilities will find this information particularly valuable as they work to protect local ecology and wildlife from chemical overexposure.

Keywords

Wildlife Risk
Environmental Risk
Herbicides
Carriers
Bar Oil
Hydraulic Fluid

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INTRODUCTION

Electricity is the most common form of energy used directly in homes, businesses, government, and communities. The effective and reliable transmission and distribution of power from where it is produced to the end users is essential to the orderly conduct of life in this society. Interference by vegetation near the power lines is a common, and a generally preventable cause of power outages. A short circuit may cause the following:

1. Loss of service to customers.
2. Damage to the vegetation and electric utility equipment. The effects of power outages vary from simple annoyances, such as the loss of power for home appliances, to safety-related problems when traffic lights stop working, to serious human health problems when such things as electric respirators may have to shift to local generator power. The longer the power is out, the greater the magnitude of the problem;
3. Fires affecting vegetation (such as forest fires) or structures; and
4. Increased risk of electrocution to humans or animals.

The management of vegetation on electric utility rights-of-way (ROW) is an essential part of managing electrical transmission and distribution systems. A variety of manual, mechanical and chemical methods, singly or in combination, are used for this purpose. No method of vegetation management is appropriate for all sites and sets of conditions. The method or methods selected must be both safe for humans and the environment, and be cost-effective in accomplishing the goals of ROW management. Safety for humans and the environment includes not causing unacceptable adverse effects. In this context, risk assessment is the process by which the likelihood of unacceptable adverse effects from the use of various methods of vegetation management can be determined.

EPRI has undertaken a 2-year effort to identify and evaluate these risks. This effort is in four parts, including the following:

Part 1. Human health risk assessment of chemicals encountered in vegetation management on ROW (completed in December 2003 and issued as EPRI, 2003).

Part 2. Assessment of human injuries from accidents, during vegetation management on ROW.

Part 3. Comparative assessment of human health risk from chemicals and injuries.

Part 4. Environmental risk assessment of chemicals used in vegetation management on ROW.

This report is Part 4 of this effort. Reports on Parts 2 and 3, and an integrated executive summary of all the reports are planned for completion in 2005.

PART 1. ENVIRONMENTAL AND WILDLIFE RISK ASSESSMENT OF CHEMICALS ENCOUNTERED IN VEGETATION MANAGEMENT ON ELECTRIC UTILITY ROW

Introduction

There are a variety of methods commonly used for the management of vegetation within electric utility rights-of-way and substations in the United States (US). Herbicides are a key part of the strategies used for this purpose. Herbicides are often combined with carriers, such as diesel oil, light fuels oil, kerosene, or some non-petroleum based materials. All of these are chemicals and can have environmental and wildlife impacts, depending on their toxicity and behavior in the environment. Mechanical cutting and manual cutting, perhaps surprisingly, also involve chemicals, especially hydraulic fluids used in machinery and bar oil used in chainsaws. Environmental and wildlife risk is also present in connection with the use of manual and mechanical methods, because of the involvement of these chemicals.

There is a well-established method for quantitatively evaluating the risk to human health from exposure to each of these types of material (National Research Council (NRC), 1983). This method is described in detail in EPRI (2003). The same fundamental approach is used in wildlife risk assessment.

Purpose

The purpose of this report is to:

- Familiarize EPRI members with the basic concepts and methods used in environmental and wildlife risk assessment.
- Describe the environmental behavior and toxicity of each chemical.
- Identify no adverse effect exposure levels in the most sensitive test in the most sensitive organism.
- Estimate the exposure to these chemicals that wildlife species may receive due to the vegetation management on electric utility ROW.
- Quantify the risk to the environment and to wildlife, and suggest strategies for mitigation of risks that are not acceptable.

Scope

The scope of this report includes the following:

Methods of Vegetation Management

- Chemical (ground-based applications only)
- High volume broadcast foliar
- Low volume selective foliar
- Low volume selective basal
- Cut surface
- Mechanical cutting
- Manual Cutting

Note that aerial application is not included because it is somewhat limited in scope of use on electric utility ROW in the US, and because the risk assessments done by USDA Forest Service (1989) and others provide a reasonable approximation, as it relates to ROW management with this tool.

Chemicals

- Herbicides, including the eight herbicides most widely used on EPRI member electric utility rights-of-way, based on a May 2003 survey to which 15 EPRI members responded.
- 2,4-D
- Fosamine ammonium
- Glyphosate
- Imazapyr
- Metsulfuron methyl
- Picloram
- Sulfometuron methyl
- Triclopyr
- Herbicide carriers, bar oil and hydraulic fluid

Wildlife Species

- Avian and terrestrial species
- Aquatic species

Organization of This Report and How to Use It

This report is organized into 6 sections and a summary. Section 1, this introduction, introduces the topic, identifies the scope and explains how the report is organized and how it should be

used. Section 2 contains 10 parts, covering the specifics of the risk assessment for each chemical or group of chemicals. Part 1 of Section 2 explains the concepts and practice involved in environmental and wildlife risk analysis.

Users of this report should read Part 1 of Section 2 before reading the specific risk assessment for any one of the chemicals covered in Parts 2 through 10. Any of Parts 2 through 10, when used with Part 1, can serve as stand-alone environmental and wildlife risk assessments. Thus, as long Part 1 is read, it is not necessary to read Part 5 (glyphosate) to be able to read and understand Part 8 (sulfometuron methyl). It may also be helpful to read portions of the human health risk assessment (EPRI, 2003) because it covers these same chemicals, but with an emphasis on human health risk.

Note that readers of this report will be able to download selected Sections and Parts of this document for their individual formatting and use as needed.

Section 3 is a brief discussion of extrapolation of the results of this analysis across the diverse environments of the US.

Section 4 summarizes the environmental and wildlife risks on a chemical by chemical basis and provides guidance on techniques and approaches that can be used to mitigate levels of risk that may not be acceptable.

Section 5 is a listing of the references cited in the text and tables of the report. When a more complete understanding of a specific point is desired, the citations should provide sufficient additional detail.

Section 6 is a glossary of terms found in this report.

How This Investigation Was Conducted

This report has been prepared with liberal use of a variety of major reviews that have been developed by or for the USDA Forest Service, the US Environmental Protection Agency, the USDHHS Agency for Toxic Substances and Disease Registry, and the California Department of Pesticide Registration (please see below). Several thorough reviews have been published in the open scientific literature. The reviews cited have been prepared with access to proprietary data not accessible to the public, except under confidentiality agreements. All of the governmental reviews are publicly available on line and in some libraries.

The review sources mentioned above include significant risk assessments that are relevant to ROW vegetation management. Specifically the following:

- Herbicide Background Statements (USDA Forest Service, 1984). A series of these were prepared for USDA Forest Service and were published in 1984. While perhaps seeming a bit old, these Statements remain relevant today. These documents review the basic chemistry, environmental behavior and fate and the basic toxicology of each herbicide, and estimate

exposure and dose. The primary weakness of the Statements is that the latest literature is not included; however, they remain valuable reference documents.

- Environmental Impact Statements (USDA Forest Service, 1989 and 1997). Both of these EIS documents provide comprehensive reviews of many of the herbicides covered by this report. The 1989 document covers all aspects of Forest Service forest management, including ROW management, in the Southern US.
- The 1997 EIS, Vegetation Management on Electric Utility Rights-of-Way, Allegheny National Forest, was prepared by Logan Norris, Frank Dost and Rufin VanBossuyt. It focuses specifically on vegetation management on electric utility ROW and is site specific for the Allegheny National Forest (NE Pennsylvania).
- Both documents are comprehensive and consistent with current standards for risk assessments.
- The only limitations of the documents is that (a) the literature is not up to date, and (b) the focus is on a limited portion of the US. Most of what the documents include; however, is relevant to ROW management in most settings in the US. There could be unusual situations, for example, extremely hot and dry or cold and wet climates where environmental behavior of the chemicals would be different, but experience has shown these differences make relatively little impact on the outcome of risk assessments.
- Human Health and Ecological Risk Assessment Final Reports. USDA Forest Service commissioned a series of risk assessments that were done by Syracuse Environmental Research Associates, Inc. (SERA). These are very recent, with completion dates ranging from 1998 to 2003. These reports are extremely comprehensive and provide a very useful reference tool for those requiring more detail on risk assessment than is include in this report. The Final Reports are available through the Internet at www.fs.fed.us/foresthealth/pesticide/health. At this web site, look under health and safety and select risk assessments.
- California Department of Pesticide Regulation (CDPR) reviews pesticide registrations and some other chemicals independently of USEPA. These are available on line at www.cdpr.ca.gov/docs/docsums/toxsumlist.

This report frequently cites these sources and incorporates material contained in the documents. In some cases, the material is attributed to these review documents and, as necessary in other cases, to the original literature cited in these other reports. Citation of specific literature in this report reflects acceptance by the investigator that the literature as sound, except as noted.

In addition, to identify specific literature or other sources of information that may not have been included in those reviews, electronic searches of literature using several search engines were conducted. These include Medline and Toxline (The National Library of Medicine), the Federal Register, Agricola, and Environmental Sciences and Pollution Management (which includes Aquatic Sciences and Fisheries Abstracts and Conference Papers Index). Searches of the database of Material Safety Data Sheets and Google searches of commercial sources were also made.

The “literature” on the topics covered in this report is voluminous. The approach is to specifically cite the review documents used, which include the underlying literature, and to cite any specifically relevant literature, including that not found in the cited review documents (publications that appeared after a specific review was published).

2
ENVIRONMENTAL AND WILDLIFE RISK ASSESSMENT

PART 1. CONCEPTS AND PROCESSES OF ENVIRONMENTAL AND WILDLIFE RISK ASSESSMENT

Part 1 of this section provides background information on the concepts and processes used in environmental and wildlife risk assessment. Detailed analysis for each of the chemicals covered by this report are in Parts 2 through 10 of Section 2. Environmental and wildlife risk assessment use different approaches. These are outlined below.

Concepts and Process of Environmental Risk Assessment

The primary emphasis of this environmental risk assessment relates to the direct chemical contamination of key parts of the environment resulting from chemical applications for vegetation management on electric utility rights-of-way. The four key parts of the environment covered by this report are plants, soil, water and air. The approach to risk assessment in this report is to summarize what is known about the magnitude and persistence (and in some cases the movement) of each chemical in Parts 2 through 10 of Section 2 of this report.

Chemical contamination of plants is important because they can serve as a route of chemical exposure for humans and/or wildlife. Exposure can occur because plants are a source of food for humans and wildlife, and they are an environmental surface from which chemical residues could be transferred to the skin. The consequences of such exposure by humans were covered in detail in the report on human health risk assessment (EPRI, 2003). Consequences of wildlife exposure are covered in this report. The most important aspects of plant contamination are the magnitude and persistence of chemical residues, and these are summarized for each of the chemicals in Parts 2 through 10 of Section 2 of this report.

Soil contamination is important because of effects on vegetation or wildlife that live in the soil and the potential for ground water contamination, if the subject chemical persists and leaches to a significant degree. The residue level, persistence and mobility of chemicals in soil are covered in detail for each chemical. The risk of ground water contamination is the most important consequence of soil contamination and is covered in the portions dealing with soil and ground water in Parts 2 through 10. Available information on effects of soil residues on plants and animals is covered under soil contamination for each of the chemicals in Parts 2 through 10. In most instances, there is more literature on ground water pollution potential than there is on the effects of soil contamination on plants or wildlife. For these reasons, ground water contamination risk receives most of the attention in considering soil contamination.

Contamination of water includes both surface and ground water. Surface water contamination can occur due to wind drift or direct application of chemicals to the water surface, or indirectly through runoff of chemicals from treated areas upslope. Evidence related to both of these is covered for each of the chemicals in Parts 2 through 10 of Section 2 of this report. The consequences of water contamination for human health were covered in the human health risk assessment report (EPRI, 2003). The consequences for wildlife due to exposure to contaminated water are covered in this report.

Contamination of air is important because air is a possible route of exposure for humans and wildlife, and the potential for chemical deposit outside of treated areas. Chemicals can be in the air (a) as vapors due to volatilization from spray droplets or sprayed surfaces or (b) as liquids or solids in spray droplets. Unfortunately, air contamination by the chemicals covered by this report has received almost no direct study. This report uses information on volatility and experience with spray drift are to assess the likelihood of air contamination. The EPRI (2003) report on human health risk concluded that inhalation was not a significant route of human exposure. The consequences of air contamination for wildlife are assessed in this report.

Concepts and Process of Wildlife Risk Assessment

The concepts of wildlife and human health risk assessment are essentially the same. It involves determination of critical values of toxicity, estimation of levels of exposure and a comparison of these to determine the risk of adverse toxic effects. It may be helpful to review Part 1 of Section 3 of the EPRI (2003) report on human health risk.

In this report, the wildlife risk assessment is subdivided to address avian and terrestrial species in one portion and aquatic species in a separate portion in Parts 2 through 10. The report draws heavily on the detailed wildlife risk assessments in USDA Forest Service (1989 and 1997), and the scientific literature published since these USDA reports were prepared.

Avian and Terrestrial Species

In the Southern Region Final Environmental Impact Statement (FEIS) for Vegetation Management in the Coastal Plain/Piedmont (USDA Forest Service, 1989), wildlife exposure was calculated for groups of wildlife species found in forest vegetation. Both typical and maximum exposure estimates were made for representatives of each group based on the three major exposure routes: inhalation, dermal, and ingestion. The same approach is used here.

Since none of the herbicides covered in this report bioaccumulate, or have long-term persistence in food chains, chronic toxic effects were not considered a problem and were not included in this analysis.

Application rates in right-of-way management are harder to estimate than in large-area broadcast sprays, such as those used in some instances in forest management. Typically, all parts of a right-of-way are not treated, because spray materials are usually directed at target vegetation. The amount of area treated or untreated will vary with both the density of the vegetation and the

method of application. The largest proportion of right-of-way will likely be treated when high volume foliar application is made to a right-of-way, with a high density of tall-growing vegetation.

In this analysis, the typical application rate is what is expected on average in the areas that are directly treated. The maximum rates occur when the highest rate of application allowed by the label is used on a broadcast basis on the right-of-way. Residue levels will be lower, with increasing distance from these areas.

Avian and Terrestrial Species Hazard Analysis

Critical toxicity data for avian and terrestrial species are drawn from USDA Forest Service (1989 and 1997), SERA (1998a,b; 1999, 2000, 2002, 2003a,b,c), and, in some cases, USEPA Registration Eligibility Documents (REDs). There is a paucity of specific toxicity testing for most herbicides on wildlife species. As a consequence, surrogates are used to represent them. USDA Forest Service (1989) discusses this concept in detail (see their Appendix, Section 8). Specific critical toxicity data is available for mammals (rat, mouse, dog, rabbit) and birds (bobwhite quail, mallard) as required for pesticide registration by USEPA. These laboratory species are reasonable surrogates for other mammalian and avian wildlife species. Little, if any, testing is done on amphibians or reptiles, but as noted in USDA Forest Service (1989) a reasonable correlation has been reported between avian toxicity and amphibian toxicity, and there is reason to believe a similar correlation exists with reptiles. Consequently, in this report (as in USDA Forest Service, 1989 and 1997), avian species are used to represent amphibians and reptiles in the risk analysis.

Avian and Terrestrial Species Exposure Analysis

Wildlife inhalation exposures were assumed to come from animals breathing in herbicide spray droplets of respirable size from a cloud of those droplets as outlined in USDA Forest Service (1989). This is an insignificant route of exposure. Dermal exposures are believed to come from two sources: 1) directly from herbicide spray at the deposition rate that should occur on vegetation leaf surfaces in the typical case and at the herbicide application rate in the maximum case, and 2) indirectly by contact with contaminated vegetation. The USDA Forest Service exposure estimates are used in this analysis, and are extended to the other herbicides in this report proportional to the rates of application of these materials. Wildlife ingestion exposure can result from consumption of treated vegetation or contaminated water. Each species is assumed to consume a percentage of its daily intake in contaminated food and water items, as described in USDA Forest Service (1989).

For herbicides in this assessment that are not included in USDA Forest Service (1989 or 1997), the investigator estimated exposure based on herbicides with a similar pattern of use, adjusted for their rates of application.

Avian and Terrestrial Species Risk Analysis

Wildlife risk from vegetation management with the chemicals covered in this report is a function of the inherent toxicity (hazard) of each herbicide to different species and of the amount of each chemical (exposure) those species may take in as a result of managing the vegetation on electric utility rights-of-way or substations.

This report uses the criteria used by the US Environmental Protection Agency (USEPA) to judge the risks to the representative avian and terrestrial wildlife species. The USEPA criteria call for a comparison of an estimated environmental concentration (EEC) with a laboratory-determined LD_{50} or LC_{50} for the most closely related laboratory test species. Where the EEC exceeds one-fifth LD_{50} or LC_{50}, USEPA deems it a significant risk that may be mitigated by restricting use of the herbicide. In this risk assessment, a species total estimated dose (rather than an EEC) is compared with the laboratory toxicity level because the dose comes from all exposure routes, not just feeding. Total estimated dose includes both direct and indirect dermal doses, ingestion doses, and inhalation doses.

The wildlife risk assessments in this report tend to overstate the risks because many of the assumptions are quite conservative. For example, no degradation of the herbicide is assumed to occur and all herbicide sprayed is assumed to be biologically available. Animals are assumed to receive direct and indirect dermal doses through the skin, ingestion doses, and inhalation doses, as described in detail in USDA Forest Service (1989).

Aquatic Species Hazard Analysis

Critical toxicity data for aquatic species are drawn from USDA Forest Service (1989 and 1997), SERA (1998a,b; 1999, 2000, 2002, 2003a,b,c) and, in some cases, USEPA Registration Eligibility Documents (REDs).

Aquatic Exposure Estimates

Exposure was assumed to occur for herbicides that drift offsite from high and low volume foliar application (EPRI, 200X). Low volume basal application and cut surface application are not likely to result in drift at a level important to this analysis. Typical and maximum concentrations in water due to foliar application of the herbicides, spills into a pond and accidental direct application to surface water are included in the exposure and risk analyses.

Aquatic Risk Analysis

To estimate the risk of adverse effects occurring, the most sensitive toxicity value was compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). USEPA compared the Q values to the risk criteria defined in Table 2-1.

Table 2-1
Definition of "Q" values

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

Thus, if the critical toxicity value is a LC_{50} of 5.0 ppm in the water, and the expected environmental concentration (EEC) is 0.5 ppm, the Q value would be 0.1, equating to no acute risk.

Concepts and Process of Environmental Risk Assessment

The key parts of the environment are air, plants, soil and water. Some elements of environmental risk assessment are folded into the wildlife risk assessment covered in this report or the human health risk assessment in EPRI (2003). This is the case for chemical residues in plants and surface water and air. There is some potential for off-site damage to plants due to air contamination and this is the perspective used in this report. The environmental risks due to chemical residues in soil are primarily the potential for ground water contamination and the impact on long-term soil productivity through impacts on soil organisms.

Except for human health and wildlife risk, there are no well defined processes by which the environmental risks from chemical residues in air and soil can be assessed. The investigator includes relevant literature and draws subjective conclusions about these risks in this report.

PART 2. RISK ASSESSMENT FOR 2,4-D

Introduction

2,4-D herbicide resembles the chemical nature of a plant hormone and is sold in several commercial products formulated as one of several amine salts or esters. 2,4-D is typically applied to foliage but the ester formulations in oil can be used for basal bark applications. Application rates range from 0.56 kg a.e./ha to 9.5 kg a.e./ha (0.5 to 8.5 lb a.e./acre). 2,4-D is commonly applied in combination with other herbicides such as picloram and dicamba.

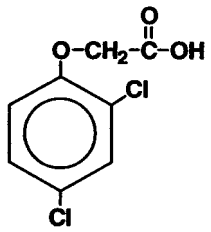
2,4-D has been in use since the early 1940's and early research was summarized in several documents written in the 1970's and 1980's including USDA Forest Service (1984 and 1989). More recent reviews of 2,4-D toxicology and epidemiology have been published including Munro et al. (1992) and Garabrant and Philbert (2002). Other reviews have been concerned with toxicology and environmental behavior (SERA, 1998b) and worker risk associated with use of

2,4-D (Dost, 2003b). Recent regulatory reviews include FAO-WHO (1996), USEPA (1999), California Department of Pesticide Regulation (CDPR, 2002a), and European Commission (2001).These reviews and expert panels have concluded that risks associated with labeled uses of 2,4-D are not significant. These reviews are the primary resources for this risk assessment and are not specifically referenced. Additionally, environmental reviews have been conducted by the FAO and WHO Working Groups (1998) and the WHO Working Group (1984 and 1989). A general description of 2,4-D and environmental facts is also available in at fact sheet at http://infoventures.com/e-hlth/pesticide.24d.html

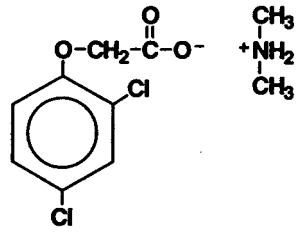
Physical-Chemical Properties and Environmental Behavior

Physical-Chemical Properties

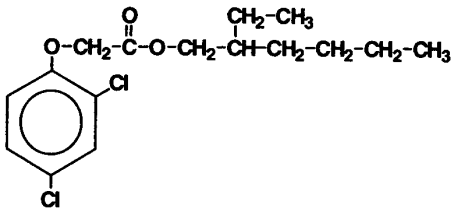
The structure of 2,4-D acid and three common commercial forms are shown below.



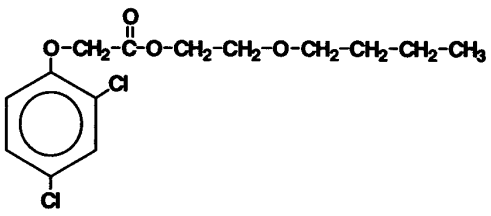
2,4-D Acid



2,4-D Dimethylamine Salt



2,4-D Isooctyl Ester



2,4-D Butoxyethyl Ester

The physical properties most important in determining 2,4-D movement, persistence and fate in the environment are solubility in water (influences leaching, adsorption in soil and accumulation by animal tissues), vapor pressure (influences tendency to volatilize), K_{oc} (influences adsorption in soil), and K_{ow} (influences accumulation in animal tissues). These are listed in Table 2-2.

Table 2-2
Selected physical and chemical properties of 2,4-d acid and commercially significant salts and esters

Chemical	2,4-D (acid)	2,4-D Dimethylamine	2,4-D Butoxyethyl ester	2,4-D Isooctyl esters
Water solubility (mg/L)	23180 (25C, pH7)	3 x 10+6 (20C)	12 (25C)	0.07 (1-octyl ester of 2,4-D)
Vapor pressure (mm Hg)	1.42 x 10-7 (25C)	8 x 10-10 (38C)	4.50 x 10-6 (25C)	Probably similar to butoxyethyl ester
Log K_{ow}	-0.75 (pH7)	0.65	4.10 (estimated)	6.73 (estimated)
Soil adsorption K_{oc} (L/kg)	20-109 in various soils	72 - 136 (avg. of 109 in three soils)	6607-6900, and 1100 in different soils	25000 – 68000 in different soils

From Table 2-3 SERA (1998b)

Environmental Behavior

2,4-D is probably the most extensively and intensively studied herbicide used on ROW today. This reflects the 60 years it has been available for study and use, the remarkable extensiveness of its use, especially in agriculture, and to some degree the controversies that have surrounded its use. National Research Council of Canada (1978), USDA Forest Service (1984), Norris (1981) and Norris (1991) provide extensive reviews of the environmental behavior of 2,4-D in the environment. SERA (1998b) provides a quite recent and thorough examination of the subject from a risk assessment perspective. The following section briefly summarizes information from these and other sources.

Behavior in Air

The tendency of 2,4-D to volatilize depends on the chemical form used. 2,4-D acid, inorganic salt, amines, and long-chain esters have low vapor pressures. The oil-soluble amines are least volatile (Information Ventures, 1995). 2,4-D esters have a higher level of volatility as reflected in their greater vapor pressure, but even these are not overly volatile. In addition, studies have documented the rapid rate at which the ester is hydrolyzed on contact with the environment, including in soil, water and on plants. The product of hydrolysis forms either a salt or acid depending on the pH of the medium.

Studies in Canada have reported 2,4-D present in air, surface water, spring runoff from snowmelt, and bulk atmospheric deposition samples (Waite et al., 2002). Waite et al. (2002)

reported maximum air concentrations during two summers in Saskatchewan to be 3.90 ng/m^3; bulk atmospheric concentrations (wet and dry) were 3550 ng/m^2/d; water film concentrations were 332 ng/m^2; and surface pond water concentrations were 290 ng/L. In northern Manitoba, Rawn et al. (1999a) reported maximum concentrations of 2,4-D in ambient air during summer months over a four-year period as 29, 24, 20, and 3500 pg/m^3 in the vapor phase and 31,26, 18, and 41 pg/m^3 in the particle phase. 2,4-D present in precipitation were directly deposited to surface creek flows in the study area. Hill et al. (2002) reported that 2,4-D present in Alberta rainfall was affected by location, use, and season. Highest concentrations (17-53 μg/L) were found in a small rainfall (0.1–0.2 mm) occurring during the agricultural spray season. Lowest concentrations (21–96 μg/m^2) were found in areas where no spraying occurred within 20 km and highest concentrations occurred in farming areas (44–315 μg/m^2; Hill et al., 2002).

Behavior in Plants

Uptake of 2,4-D occurs through leaves, stems, and roots. Translocation of 2,4-D, once absorbed in the plant, depends on a variety of factors including the point of entry into the plant, environmental conditions such as moisture and temperature, and the physiological status of the plant. In general, foliar-applied 2,4-D translocates readily and is carried with photosynthate to growth sites in roots, shoots, flowers, and fruits. Foliar-applied 2,4-D in dark-grown plants is not translocated until sugar is available for transport in the phloem. Translocation is strongly affected by soil moisture and atmospheric humidity, with low soil moisture resulting in reduced translocation from leaves to roots and high atmospheric humidity stimulating downward translocation of 2,4-D. Increased temperature in the 68 to 86°F (20 to 30°C) range results in increased rates of translocation of 2,4-D. Translocation upward by root-applied 2,4-D is restricted and takes place primarily in the transpiration stream of xylem (Loos, 1975). Studies with small aspen plants (*Populus tremula*, approximately 1 foot high), however, indicated that translocation of foliar-applied 2,4-D occurred mainly upward to the growing shoot tip. Very little additional herbicide ended up in the roots when injected via cut stems (E1iasson and Hallmen, 1973). These data indicated a ready transfer of 2,4-D from phloem to xylem in the plant.

2,4-D tends to persist in the plant tissues for a longer period than the natural plant hormone (Loos, 1975 and Mullison, 1982). Plants metabolize 2,4-D readily by a variety of pathways to various degradation products. These pathways and processes are discussed in detail in Loos (1975). Derivatives of 2,4-D, such as esters, appear not to function as plant growth regulators until they are first converted to the acid form (Loos, 1975). This conversion occurs rapidly, with most esters converted to the acid form within approximately 30 minutes (Scifres, 1977).

Several studies have addressed the persistence of 2,4-D residues following application to various types of vegetation in both laboratory and field situations. Residues of 2,4-D are generally relatively nonpersistent in plants. For example, in data from studies referenced in USDA Forest Service (1984), the following percentage of the initial residue level occurred in 30 days after application to foliage: poplar foliage 48%, chamise foliage 73%, grass and forbs 92%, grass 62%.

Several wild fruits were analyzed for 2,4-D residues from sites along right-of-ways, conifer site preparation or release spray blocks, and one farm. Frank et al. (1983) reported that application

rates varied from 0.8 to 6.0 kg a.e./ha and spraying occurred between June and September to plants in flowering to ripe fruit stages. Spray applied to flowering wild red raspberries resulted in negligible residues on fruits; applied to immature fruits resulted in 0.2 mg/kg residues on ripe fruits; applied to ripe fruit initial residues ranged from 2.6 to 31 mg/kg and declined to 0.1 to 3.3 mg/kg over a 2- to 5-week period. However, residues on ripe fruit did not decline when sprayed on ripe blueberries (0.8 to 3.0 kg/ha) or immature pin cherries (3–5 mg/kg residues remained on ripe fruit over a 32-day period). Residues declined rapidly on wild strawberries on two sites and slowly at one site.

Based on this type of data, USDA Forest Service (1989) and SERA (1998b) reported an exposure scenario involving the consumption of fruit, such as berries, consumed by humans shortly after application of 2,4-D. The concentration of herbicide on these materials is considered to be 125 mg/kg per lb a.i./acre on leaves and leafy crops; 1.5 to 12 mg/kg for fruits, grains, and seed pods and up to 30 mg/kg for berries. It is reasonable to assume similar exposure levels for wildlife species that might consume the same vegetation. These assumptions are incorporated into the wildlife risk assessment.

Behavior in Soil and Groundwater

Soil

There is an enormous amount of information on the behavior of 2,4-D in its various forms in the soil. SERA (1998b) provides a detailed coverage of this literature. The essence of the soil behavior is reported briefly in WSSA (1994).

2,4-D does not persist in soil because of its rapid degradation (WHO Working Group, 1989). 2,4-D persistence and kinetic of its degradation in soils are presumed to be associated with the population dynamics of 2,4-D degrading microorganisms (Sandmann et al., 1988). Sandmann et al. (1988) discuss microbial degradation in soils in depth and found that the rate of microbial degradation is affected by the rate of application, the formulation of 2,4-D, soil type, adsorption and availability of 2,4-D, the concentration of the degrading microorganisms, soil moisture, temperature, pH, and oxygen, and soil amendments. The ester or amine salts undergo rapid hydrolysis or dissociation in most soils, while the isooctyl ester can persist from several days to several weeks.

Laboratory soil tests of 2,4-D degradation were conducted on two Philippine clay soils (Maahas and Luisiana) obtained from paddy fields (Yoshida and Castro, 1975). Half of the soil samples were maintained at submerged conditions while the other half were maintained at 80% of field moisture capacity (upland conditions). 2,4-D was applied at a concentration of 20 ppm; residues were determined 0, 2, 4, and 6 weeks after incubation. In the Maahas soil, little 2,4-D was found under upland conditions 2 weeks after incubation while under the submerged conditions after a 2 week lag time, 2,4-D degraded rapidly and little remained after 4 weeks. Degradation was slower in the Luisiana soils and less than 40% of the 2,4-D was recovered in both soil conditions the sixth week of incubation. 2,4-D applied to rice paddies at rate of 0.43 kg a.i./ha after emergence dissipated rapidly with initial dissipation of 50% of the initial concentration (DT_{50S}) in paddy

water, dryland-rice soil, and bare ground soil being 10 days or less (Johnson et al., 1995). 2,4-D was not detected in water 28 days after application and was slightly more persistent in soil than water.

The investigator concludes that the average persistence of phytotoxic levels is 1 to 4 weeks in warm moist soils, and that the average field half-life is 10 days. The mechanism of dissipation is primarily microbial degradation as demonstrated by the much longer persistence found in sterile soils maintained under sterile conditions. The rate of microbial degradation is influenced by the same factors that influence the population level of microbes and the rate of microbial metabolism. Decomposition is faster in warm, moist soils with higher levels of organic matter. Metabolism is also more rapid at higher pH levels. Photo degradation can also occur but it is relatively minor in importance in field settings.

WSSA (1994) notes that based on the water solubility of 2,4-D it is potentially mobile in soil, but as a practical matter this is minimized by the rapid rate of decomposition occurring in the soil and the rapidity of uptake by roots of plants. In essence, there is little there to be mobilized. Detectable residues below 15 cm in the soil will be unusual. There are reports that both the ester and amine of 2,4-D leached into the soil profile from irrigation and rainfall, as the soil surface dried the herbicides moved toward the soil surface by upward-flowing capillary flow. Both formulations moved in similar patterns in the soil profile.

Soil persistence and lateral movement of 2,4-D was examined following brush control treatments applied at 4.8 kg/ha to power line ROWs (Meru et al., 1990). Soil samples and runoff water was collected at time increments from 0.14 to 48 weeks after treatment. Only 3 of 85 down slope soil samples contained detectable residues of 2,4-D and only 11 of 56 water runoff samples contained detectable residues. The furthest distance from the treatment site that 2,4-D residues were found in runoff water was 20 m. No residues were found in soil or water samples at either 15 or 48 weeks after treatment.

Groundwater and Aquifers

There are very few reports in which 2,4-D has been found in groundwater. While it may be physically possible for this to occur, it is considered highly unlikely, and limited to very unusual circumstances. Two studies have reported the fate of the herbicide in aquifers. Madsen et al. (2000) reported that 2,4-D had low sorption to low organic carbon sediments (total organic carbon content below 1 g/kg) in 10 Danish aquifers. The mean sorption values (K_d) for 2,4-D were 0.06 (L/kg) and ranged from 0.01 to 0.22 L/kg; further analysis showed that at pH above 7.7 sorption is low for sandy sediments and when pH decreases the sorption increases. Albrechtsen et al. (2001) reported that 2,4-D was degraded microbially under aerobic conditions in shallow, sandy Danish aquifers under experimental trials.

Behavior in Surface Waters

Drift and direct application are the primary possible routes by which 2,4-D may enter surface waters. While runoff is possible from surfaces that are highly compacted and have little or no

covering of vegetation or plant litter, this should be a mechanism of little importance. Norris (1981), Norris (1991) and National Research council of Canada (1978) provide extensive reviews of the entry and fate of 2,4-D in water, especially as it relates to the use of this herbicide in forestry operations.

SERA (1998b) estimated the ambient level of 2,4-D in water based on a monitoring study in which 2,4-D was monitored in ground water, pond water, and runoff in a watershed in which a known amount of 2,4-D had been applied over a 3-year period. Based on this study, they estimate a concentration of 2,4-D at 0.002 (0.001-0.004) mg/L per lb a.e. of 2,4-D applied per acre.

Studies by Norris and others have usually found that entry to water is brief and while concentrations of 0.1 mg/L may be observed in some instances during or shortly after application, concentrations even as low as 0.002 do not persist for more than 24 hours. Extensive research and monitoring on electric utility rights-of-way and managed forests in the Pacific Northwest has shown little or no residues of 2,4-D or other herbicides in surface waters near the ROW. Detected residues are attributed to drift or accidental direct application to the water. With attention to the details of application, such residues are nearly entirely avoidable and should result in little or no environmental risk (Norris and Charlton, 1995; Norris 1985; Norris 1970)[1]. Contemporary ROW applications give greater attention to buffers and care in application than that used in some of these earlier studies, and the investigator concludes that water contamination with 2,4-D on or near a ROW will be rare and quite limited.

For purposes of the aquatic species risk analysis the 2,4-D concentrations in water is estimated at 0.001 mg/L for typical and 0.01 mg/L maximum rates of application, and 0.01 mg/L for typical and 0.1 mg/L for maximum accidental direct application to surface water. This analysis adopts the USDA Forest Service (1989) value of 1.7 mg/L as the concentration from a spill into a pond.

Residues in Animals and Fish

Although animals that are exposed to 2,4-D will take up some of the chemical, the bioaccumulation ratios (amount in tissue compared to ambient levels) will be low and the small amounts accumulated will be rapidly eliminated once exposure ceases. The data supporting this conclusion is provided in detail in EPRI (2003) and is not repeated here.

Wildlife Risk Analysis

2,4-D is undergoing re-registration by the USEPA. Recent data from a re-registration study for 2,4-D has been published in peer-reviewed literature and is publicly available. Because of its widespread use and the length of time it has been used, there is a more extensive scientific base for 2,4-D than any other herbicide. A wide body of literature is publicly available from laboratory, field and epidemiologic research by academic and governmental researchers is also

[1] Results reported in several unpublished reports to the Bonneville Power Administration and in a BPA Environmental Seminar delivered April 23, 1970 by Logan Norris.

available because of the wide use of 2,4-D. Several expert panels and workshops have also been assembled over past 15 years to examine the data (Munro et al., 1992; Garabrant and Philbert, 2002; USEPA, 1999a; and Dost, 2003b).

2,4-D acid and amine are relatively low in toxicity to most organisms, but 2,4-D esters are significantly more (200 to 1000 times) toxic to aquatic species. As with the environmental behavior of 2,4-D, its toxicity to wildlife and fish has been extensively studied and reviewed. National Research Council of Canada (1978), USDA (1984), Norris (1981) and Norris (1991) provide extensive reviews of the environmental behavior of 2,4-D in the environment. SERA (1998b) provides a quite recent and thorough examination of the subject from a risk assessment perspective. The following sections briefly summarize key elements of toxicity from these and other sources.

Avian and Terrestrial Species

Toxicology values from WSSA (1994) are in Tables 2-3, 2-4 and 2-5.

Table 2-3
Toxicity of 2,4-D acid

Species	Type of Exposure	Response Measured	Value Reported
Rat	Oral	LD_{50}	639 mg/kg
Rabbit	Dermal	LD_{50}	>2000 mg/kg
Rat, Mouse	Chronic Oral	NOEL	5 mg/kg/day
Dog	Chronic Oral	NOEL	1mg/kg/day
Bobwhite Quail	Oral	LD_{50}	500 mg/kg
Bobwhite Quail	8-day Dietary	LC_{50}	> 5620 ppm
Mallard Duck	8-day Dietary	LC_{50}	> 5620 ppm
Earthworm	In Soil	No Effect Level	> 2 mg/kg
Honey Bee	Acute Contact	LD_{50}	1 ug/bee

From WSSA (1994)

Table 2-4
Toxicity of 2,4-D dimethyl amine

Species	Type of Exposure	Response Measured	Value Reported
Rat	Oral	LD_{50}	>1000 mg/kg
Rabbit	Dermal	LD_{50}	909 mg/kg
Bobwhite Quail	Oral	LD_{50}	500 mg/kg
Bobwhite Quail	8-day Dietary	LC_{50}	> 5620 ppm
Mallard Duck	8-day Dietary	LC_{50}	5620 ppm

From WSSA (1994)

Table 2-5
Toxicity of 2,4-D isooctyl ester

Species	Type of Exposure	Response Measured	Value Reported
Rat	Oral	LD_{50}	1045 mg/kg
Rat	Dermal	LD_{50}	>5000 mg/kg
Bobwhite Quail	8-day Dietary	LC_{50}	> 5620 ppm
Mallard Duck	Oral	LD_{50}	663 mg/kg
Mallard Duck	8-day Dietary	LC_{50}	> 5620 ppm

From WSSA (1994)

The toxicity of 2,4-D to amphibians has not been investigated as thoroughly as the toxicity of 2,4-D to fish but the available data suggest the 2,4-D may be more toxic to some amphibians than to fish. In general, the acute LC_{50} values for amphibians are comparable although at the lower range of the LC_{50} values for fish (i.e. about 200 mg/L). An earlier report, however, suggests that at least some species of amphibians (i.e., *Bufo melanostictus*) have LC_{50} values for 2,4-D acid that are much lower (i.e., a 96-hour LC_{50} of about 8 mg/L). No data have been encountered on the toxicity of 2,4-D esters to amphibians. By analogy to the data on fish, the acute toxicity of 2,4-D ester is likely to be greater than the acute toxicity of 2,4-D acid or salts.

The wildlife risk analysis is in Table 2-6.

Table 2-6
Wildlife risk analysis for 2,4-D

Species Group	Dose Estimate		Critical Toxicity Values		Representative Laboratory Species[1]
	Typical	Maximum	$1/5LD_{50}$	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	22	200	60	300	chukar
Mammals	36	405	75	375	rat
Amphibians	22	199	60	300	chukar
Reptiles	13	120	60	300	chukar

[1] The common laboratory test species used to represent the group of wildlife species.

This analysis shows that typical exposure levels are below the critical toxicity value of 1/5 LD_{50} values in all cases, although in each case the maximum exposure level exceeds the 1/5 LD_{50} values in all cases, and in the case of mammals as represented by the rat exceeds the LD_{50} value as well.

Aquatic Species

The aquatic toxicity of 2,4-D varies widely between acid, amine and ester forms. The ester forms are significantly more (typically 100 times more) toxic than the acid or amine forms, but the ester hydrolyzes rapidly to acid or salt forms, reducing its actual toxicity in field settings. Average 96-hour LC_{50} values range from 0.5 mg/L to 10 mg/L.

Aquatic toxicology values from WSSA (1994) and USDA Forest Service (1989) are in Tables 2-7, 2-8 and 2-9.

Table 2-7
Toxicity of 2,4-D acid

Species	Type of Exposure	Response Measured	Value Reported
Daphnia	In Water	48-hour LC_{50}	25 mg/L
Bluegill Sunfish	In Water	96-hour LC_{50}	263 mg/L
Rainbow Trout	In Water	96-hour LC_{50}	377 mg/L

From WSSA (1994) and USDA (1989)

Table 2-8
Toxicity of 2,4-D dimethyl (or other) amine

Species	Type of Exposure	Response Measured	Value Reported
Chinook Salmon	In Water	96-hour LC_{50}	> 100 mg/L
Smallmouth Bass	In Water	96-hour LC_{50}	236 mg/L
Channel Catfish	In Water	96-hour LC_{50}	119 mg/L
Crayfish	In Water	48-hour LC_{50}	> 100 mg/L
Eastern Oyster	In Water	NOEL at 95 hours	>2 mg/L
Daphnia	In Water	48-hour LC_{50}	184 mg/L
Bluegill Sunfish	In Water	96-hour LC_{50}	168 mg/L
Rainbow Trout	In Water	96-hour LC_{50}	250 mg/L

From WSSA (1994) and USDA (1989)

Table 2-9
Toxicity of 2,4-D isooctyl (butoxy ethyl) ester

Species	Type of Exposure	Response Measured	Value Reported
Stonefly Nymph	In Water	96-hour LC_{50}	1.6 mg/L
Stonefly Adult	In Water	96-hour LC_{50}	> 1000 mg/L
Crayfish	In Water	48-hour LC_{50}	> 100 mg/L
Eastern Oyster	In Water	96-hour EC_{50}	3.75 mg/L
Daphnia	In Water	48-hour LC_{50}	5.2 mg/L
Bluegill Sunfish	In Water	96-hour LC_{50}	1.2 mg/L
Rainbow Trout	In Water	96-hour LC_{50}	1.5 mg/L

From WSSA (1994) and USDA (1989)

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by USEPA where the risks of adverse effects to fish or invertebrates as described in Table 2-10.

Table 2-10
Definition of "Q" value

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic risk analysis is in Table 2-11.

Table 2-11
Aquatic species risk analysis for 2,4-D

		Off-Site Drift			Accidental Direct Spray	
Herbicide	**Critical LC_{50}**	**Typical**	**Maximum**	**Spill in Pond**	**Typical**	**Maximum**
PPM ----------Q Value----------						
2,4-D Acid	25	<0.1	<0.1	<0.1	<0.1	<0.1
2,4-D Amine	119	<0.1	<0.1	<0.1	<0.1	<0.1
2,4-D Isooctyl Ester	1.6	<0.1	<0.1	1.06	<0.1	<0.1

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to aquatic species for typical and maximum exposures resulting from off-site drift, or accidental direct spray to surface water. All Q values are less than 0.1. A spill into a pond results in an exposure level that is likely to produce toxic effects on sensitive aquatic species.

Other Aquatic Species

Phytotoxicity of technical 2,4-D to an aquatic macrophyte, *Myriophyllum sibiricum,* was studied under laboratory conditions to determine inhibitory concentration 25 and 50 [IC_{25} and IC_{50}, concentrations (mg a.i./L) that inhibit an endpoint parameter by 25 and 50%, respectively. Roshon (1997, as cited by Roshon et al., 1999) used the maximum label rate for technical 2,4-D of 5.1 kg/ha (expected environmental concentration of 3.4 mg a.i./L). IC_{25} and IC_{50} for shoot growth were 0.41 and >1.47, respectively; IC_{25} and IC_{50} for number of roots were 0.008 and 0.018, respectively; and IC_{25} and IC_{50} for root length were 0.005 and 0.013, respectively. Roshon (1997) reported phytotoxic effects on roots and shoots at concentrations below the expected environmental concentration.

Peterson et al. (1994) also determined the phytotoxicity of 2,4-D to aquatic organisms when applied at a the label rate of 4.375 kg/ha (expected environmental concentration of 2.917 mg/L) and reported that 2,4-D at the expected environmental concentration had low toxicity to algal species and caused <50% inhibition of growth in duckweed (*Lemna minor*). For *L. minor*, Fairchild et al. (1997) reported a 96-hr EC_{50} value of >100,000/L and could not determine a NOEL because concentration exceeded highest levels tested. For a single-celled green alga (*Selenastrum capricornutum*), Fairchild et al. (1997) reported a 96-hr EC_{50} value of 41,772/L and an NOEL of 25,000.

Based on these toxicity values and the exposures expected in aquatic systems, it is clear that there should be little or no environmental impact on other aquatic organisms.

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

Virtually all data on 2,4-D in air comes from areas where extensive areas (such as large grain fields in Canada, and the mid west and western US) are treated with this herbicide. While there are no studies of 2,4-D in air in connection with right-of-way spraying, it is highly unlikely that biologically important residues would occur in air if normal precautions about drift are used. Consequently, as it is used on electric utility ROW, the investigator concludes that 2,4-D in air is not a significant environmental risk assessment.

Soil and Groundwater

Given the limited persistence and mobility in soil and the susceptibility of 2,4-D to microbial degradation under aerobic conditions, the investigator finds that the use of 2,4-D on electric utility rights-of-way, 2,4-D does not present a risk to groundwater quality.

Mårtensson (1993) reported that N-fixation by free-living soil bacteria was significantly reduced in the laboratory when exposed to 6 times the recommended application rate (0.5-3.5 kg/ha) of 2,4-D. The adverse effects were independent of soil pH, and C and N content of the soil. Ectomycorrhizal fungi often benefit the host plant through symbiotic associations with the roots of vascular plants, and are particularly important for the growth of conifer trees. Estok et al. (1989) found in lab studies using an agar media, that 2,4-D significantly reduced three species of ectomycorrhizal fungus at concentrations ≥ 1000 ppm and growth was completely inhibited at concentrations ≥ 5000 ppm.

The toxicity values reported here are far in excess of soil contamination values that might occur due to use of 2,4-D on electric utility rights-of-way, and the investigator concludes that there should be no substantive direct impact of such use on long-term soil productivity.

Based on this information the investigator concludes that due to its brief persistence and limited mobility in soil and low level of toxicity to soil organisms 2,4-D as it is used on electric utility rights-of-way presents no soil-related environmental risk of any significance.

PART 3. RISK ASSESSMENT FOR FOSAMINE AMMONIUM

Introduction

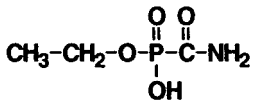
Fosamine ammonium is a post-emergence herbicide used to control woody plants and is usually applied when leaves are fully developed but before leaves begin to turn color in the fall. Commercial products containing fosamine ammonium include Krenite® S and Krenite® UT and are applied at rates ranging from 6 to 24 lb a.i./acre for brush control, or in a spray-to-wet treatment using 6 – 12 lb a.i./100 gal of water. An oil adjuvant may be added to improve penetration of foliage and stems (WSSA, 1994).

The principal information used in this report includes USDA Forest Service (1984, 1989 and 1997) and Norris et al. (1991) and are not specifically quoted. This current review and assessment is not intended to repeat the analyses of those reports but rather to provide a brief discussion of environmental and wildlife risk associated with ROW vegetation management and, where appropriate, to include recent work that may not be discussed in the major reviews.

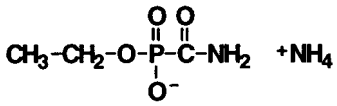
Physical-Chemical Properties and Environmental Behavior

Physical-Chemical Properties

Fosamine ammonium is the common name for ethyl hydrogen (aminocarbonyl) phosphonate and is the active ingredient in Krenite® S and Krenite® UT (both contain 41.5%; 4 lb a.i./gal technical fosamine ammonium as the ammonium salt). The inert ingredients used in the formulation are considered proprietary and are not publicly available but have been identified and evaluated by USEPA.



Fosamine Acid



Fosamine Ammonium Salt

Following are important physical properties for fosamine ammonium (WSSA, 1994):

Water Solubility: 1.7 kg/L @ 25C
Vapor Pressure: 4 X 10^{-6} mm Hg @ 25C
K_{ow}: Not available
K_{oc}: 150 L/kg

Environmental Behavior of Fosamine Ammonium

Fosamine ammonium is an herbicide for post-emergence control of woody plants. Fosamine ammonium tends to be non-toxic to nontarget plant species. It is rapidly adsorbed to soil particles, is rapidly degraded in soil, is relatively stable in water at neutral pH's, and has a very low toxicity to all species of animals in which it has been tested. It is absorbed by plant stems, buds, and foliage. Absorption through young stems appears to occur more readily than through foliage. Laboratory studies indicate that fosamine ammonium is translocated throughout the plant; however, in practice, effective action requires complete coverage of all parts of woody plants. Fosamine ammonium has a short half-life in plants and is metabolized to products, which in turn have a relatively short half-life. It does not bioaccumulate in animals (Ghassemi et al., 1982; Norris, 1991; and WSSA, 1983 and 1994).

Behavior in Air

Fosamine has a very low vapor pressure (4×10^{-6} mm Hg) meaning its vaporization will be negligible. A laboratory study of the loss of fosamine from soil shows volatilization to the atmosphere does not occur at a significant rate (Spar, 1992). Fosamine should not appear in the air except as spray droplets during application.

Behavior in Plants

Plants absorb fosamine ammonium through stems, buds, and foliage and absorption appears to occur more readily through young steam than through foliage. The effects of herbicide treatment are usually not evident until the following spring after application when buds fail to develop, or develop into miniature spindly leaves that do not provide adequate photosynthesis. Consequently, the plant dies. In some species of plants, such as pines and bindweed (nondeciduous), leaves may turn brown immediately after application (Ghassemi et al., 1981; WSSA, 1983). Laboratory studies indicate that fosamine ammonium is translocated throughout the plant; however, in practice, effective action requires complete coverage of all parts of woody plants. Fosamine ammonium has a short half-life in plants and is metabolized to products that have a relatively short half-life.

Based on available studies, the expected half-life of fosamine ammonium in plants is 10 days or less. In greenhouse studies, ^{14}C-labeled fosamine ammonium was applied to apple seedlings. When the material was applied to leaf surfaces, slower penetration was observed than when it was applied to the stems. The material applied to the leaf surface had a half-life of 2 to 3 weeks (WSSA, 1983 and Ghassemi et al., 1981 and 1982).

Metabolism of fosamine ammonium in plants was studied in the field in a pasture area in which small pin oak were surrounded by grass and clover (Chzranowski, 1983). When applied at a rate of 12 lbs a.i/acre (13.4 kg/ha), ^{14}C-labeled fosamine ammonium had an average half-life of 7 days in the pasture flora. Two metabolites, carbamoylphosphonic acid and carboxyphosphonic acid, were detected in the flora and reached peak concentrations 2 weeks after application of the

parent material and then rapidly declined. One year after treatment, neither fosamine ammonium nor the two metabolites were detected in the pasture at a detection limit <0.05 ppm.

Behavior in Soil and Groundwater

There are only limited data available on the behavior of fosamine ammonium in soil. These data, provided by Han (1979a), as cited by Ghassemi et al. (1981 and 1982), by the WSSA (1983), and by Norris (1991).

Fosamine ammonium is degraded relatively rapidly by soil microorganisms according to studies reported by Han (1979a in Ghassemi et al., 1982 and in WSSA, 1983). Radiolabeled fosamine ammonium was incorporated into sandy loam and into silt loam to a concentration of 4 and 20 pm each in laboratory flasks. Within 90 days, between 45 and 75 percent of the applied radiolabeled carbon had been given off as $^{14}CO_2$ when the material had not been sterilized. Degradation in sterile flasks, however, was negligible for the first 20 to 30 days, and minimal $^{14}CO_2$ evolution observed subsequently was attributed to loss of sterility. Laboratory biometer flask studies show microbial decomposition of [^{14}C] fosamine ammonium to $^{14}CO_2$ to be 45 to 75 percent complete after 90 days in the dark.

The same reports also reported hydrolysis of fosamine ammonium to carbamoylphosphonic acid in soil. In the field, the metabolite carbamoylphosphonic acid (CPA) was found several days after initial soil treatment, but all [^{14}C] fosamine ammonium and [^{14}C] CPA had disappeared completely by 3 to 6 months after application. The environmental conditions of temperature, moisture, and pH that favor this reaction have not been reported, however.

Fosamine ammonium is a very low mobility herbicide and is not readily leached from soil or carried away in precipitation runoff, despite its high water solubility, because it is readily and strongly adsorbed to soil particles. Because of this strong adsorption of fosamine ammonium to soil particles, there is little tendency for groundwater contamination or for surface waters to become contaminated without direct application of the material. Laboratory and field studies, though limited, indicate that photodecomposition and volatilization are not significant mechanisms for the loss of fosamine activity in soil.

Field studies have been done with ^{14}C-labeled fosamine ammonium applied to surface soils at a rate of 10.1 lb/acre (11.3 kg/ha) (Han, 1979a in Harvey, 1983). When applied to silt loam in Delaware, very little movement of material was detected in the soil column and, after 6 months with over 36 inches (91.7 cm) of rainfall, more than half of the original applied radioactivity (58 percent) was located in the top 2 inches (5 cm) of soil. In fine sand in Florida, however, the radioactive material penetrated more rapidly despite less rainfall (40.4 cm), with only 6.4 percent of the original applied radioactivity in the top 2 inches (5 cm) of soil after 6 months. In this study, radioactivity lost from soil was assumed to be as $^{14}CO_2$ because both fosamine ammonium and its metabolite, carbamoylphosphonic acid, are essentially nonvolatile. Carbamoylphosphonic acid was detected in these soils as the fosamine ammonium metabolite but was lost from the Delaware soil within 6 months and from the Florida soil within 2 months. After 6 months in both soils, almost the entire residual radioactivity was identified as reincorporation products utilizing

the radiolabeled carbon from the parent fosamine ammonium. A half-life on the order of 1 week to 10 days in field and greenhouse studies, respectively, is expected.

Eble (1989) summarized the results of three field studies (Delaware, Tennessee and North Carolina) of fosamine movement and persistence in soil. The level of fosamine declined to less than the limit of detection in soil at the Tennessee site in 1 day and to this level in three days at the New Jersey site. In North Carolina, the rate was slower, with a half-life of 2 days. While laboratory studies indicate fosamine is highly mobile in soil, it is decomposed so quickly that movement does not occur. Fosamine ammonium persistence and movement have been determined in soils and under conditions which are reasonably similar to those on forest based ROW.

Based on this research the investigator expects fosamine ammonium to have a half-life of 7 to 10 days. It will show little or no leaching in the forest-based soils on electric utility rights-of-way, making contamination of groundwater quite unlikely.

Behavior in Surface Water

Drift and direct application are the primary possible routes by which fosamine ammonium may enter surface waters. While runoff is possible from surfaces that are highly compacted and have little or no covering of vegetation or plant litter, this should be a mechanism of little importance on rights-of-way.

Fosamine ammonium is not readily degraded in water under neutral to mildly alkaline pH conditions in laboratory environments. The relative importance of various environmental factors studied on the rate of fosamine ammonium decomposition in water exclusive of microbial action appears to be pH, temperature, and UV light, in descending order. (Han, 1979a in Ghassemi, 1981). Under actual field conditions, microbial decomposition rates would be high (Schneider, 1984). Fosamine ammonium is strongly adsorbed onto soil particles and is not readily leached from soils into streams and ponds. Fosamine ammonium can enter aquatic systems as direct spray, as spray drift, or bound to soil runoff. Once in aquatic systems, it is likely that fosamine ammonium will adsorb onto bottom sediments or suspended sediments (Norris, 1991).

At 5 ppm concentration in the dark, aqueous solutions of fosamine ammonium are stable (with less than 3 percent decomposition) for 4 weeks at neutral pH (pH 7) and under slightly basic (pH 9) conditions. Under slightly acidic conditions (pH 5), however, it hydrolyzes rapidly (half-life of 10 days) to carbamoylphosphonic acid. At a higher concentration of 7,200 ppm, however, less than 3 percent decomposition occurred at all pH's observed (Han, 1979a in Ghassemi et al., 1981 and 1982).

Data reported from photodecomposition studies with aqueous solutions of fosamine ammonium indicate that photodecomposition is very slow in the laboratory and should not play a significant role in loss of fosamine ammonium in natural aquatic systems. Han (1979a in Ghassemi et al., 1981 and 1982) reported only 2 percent decomposition of fosamine ammonium in a 5 ppm solution exposed for 8 weeks to a light source equivalent to half of the sun's intensity at noon. A

5 ppm solution at pH 5, exposed to sunlight for 4 weeks in Wilmington, Delaware, was only slightly degraded.

Under anaerobic aquatic conditions with water and sediment, fosamine showed a half-life of 4 days. When this substrate was sterilized, no decomposition of fosamine occurred, indicating its dissipation is biologically mediated (Spar, 1991).

There is no specific field monitoring data on the entry and fate of this herbicide in water that the investigator knows of, but given the pattern of use and mechanisms of entry, the investigator concludes that data for 2,4-D provides a reasonable surrogate. Therefore, for this risk analysis the concentrations in water is estimated at 0.001 mg/L for typical and 0.01 mg/L maximum rates of application, 0.01 mg/L for typical and 0.1 mg/L for maximum accidental direct application to surface water, and 1.7 mg/L as the concentration from spill of a 5-gallon container of concentrate into a pond.

Residues in Animals and Fish

Fosamine ammonium does not bioaccumulate in animals and any residues in animal tissues will be low and rapidly dissipated. Once absorbed, fosamine ammonium is rapidly metabolized to carbamoylphosphonic acid and is eliminated in feces and urine along with any parent compound not metabolized.

Rats administered 57 mg/kg of fosamine ammonium eliminated the entire dose within 72 hours (Chrzanowski et al., 1979). Approximately 87 percent of the dose was excreted in the feces and 13 percent in the urine. Thirteen percent of the eliminated dose had metabolized to carbamoylphosphonate acid, while the remainder was excreted unchanged. No toxicity studies with carbamoylphosphonate acid are available.

Han (1979) exposed channel catfish for 4 weeks to water containing 1.1 ppm radiolabeled fosamine ammonium or to water over soil pretreated with radiolabeled fosamine ammonium at 15 ppm and aged 30 days prior to flooding. In both studies, the fish reached peak body tissue levels of fosamine ammonium after 2 to 3 weeks exposure. Maximum tissue concentrations were lower than those of the surrounding water. Concentrations of radioactivity in the fish tissue were reduced by 50 to 90 percent in both cases after 2 weeks depuration (exposure to water without the radiolabeled fosamine ammonium). Fosamine did not accumulate in bluegill exposed to the herbicide at 1 ppm in water over a 28-day study period (Blasberg, Hicks and Stuerman, 1992).

Wildlife Risk Analysis

Most research concerning the toxicology of fosamine ammonium was conducted by the registrant and is not publicly available. The registration data was reviewed by USDA Forest Service (1984), USEPA (1995a), and USDA Forest Service (1997). The human risk analysis reported in EPRI (2003) contains extensive review of the basic toxicology of fosamine ammonium.

Avian and Terrestrial Species

Based on acute oral LD_{50} values of 24,400 mg/kg in rats, 7,380 mg/kg in guinea pigs, and greater than 15,000 mg/kg in dogs for the Krenite® formulation (41.5 percent active ingredient), fosamine ammonium is very slightly toxic to mammals (DuPont, 1983a; USDA, 1984). The acute oral LD_{50} of the Krenite® S formulation (Krenite® with surfactant added) is greater than 5,000 mg/kg in rats (DuPont, 1983a). Sheep given Krenite® in the diet for 90 days showed no adverse effects at doses of up to 2,500 ppm, the highest dose tested (Schneider and Kaplan, 1983, as cited in USDA Forest Service, 1984). Unformulated fosamine ammonium is very slightly toxic to birds based on acute oral LD_{50}'s of greater than 5,000 mg/kg in mallard ducks and bobwhite quail (Schneider and Kaplan, 1983, as cited in USDA Forest Service, 1984). The 8-day dietary LC_{50} of unformulated fosamine ammonium is greater than 10,000 ppm in mallards and bobwhite quail (Schneider and Kaplan, 1983, as cited in USDA Forest Service, 1984). The acute oral LD_{50} of formulated fosamine ammonium is greater than 10,000 mg/kg in bobwhite quail and mallard ducks (DuPont, 1983a).

According to a study by Lutz-Ostertag (1983), the ammonium salt of fosamine ammonium (solutions of 1 to 5 percent) is teratogenic when sprayed directly onto fertilized eggs of quail and chickens; quail eggs are more frequently and severely affected. Teratogenic effects in the quail and chick embryos included slight to severe malformations. Embryo toxicity to these species was considered low (Lutz-Ostertag, 1983).

Dr. D. Hoffman of the U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, reported that fertile bobwhite quail and mallard duck eggs submerged in 1.5-, 6.5-, and 30-percent fosamine ammonium solutions showed no teratogenic effects. Embryo toxicity was observed at the higher concentrations. However, because the exposure method (submersion) and test concentrations greatly exaggerate the likely field exposures, fosamine ammonium is not considered hazardous to avian species (O'Neal, 1987, as cited in USDA Forest Service, 1989).

Based on effects observed in honey bees, fosamine ammonium appears to be only slightly toxic to insects (USDA Forest Service, 1984). The contact LC_{50} was greater than 10,000 ppm when bees were sprayed with a 42-percent formulation of fosamine ammonium as the ammonium salt (Schneider and Kaplan, 1983, as cited in USDA Forest Service, 1984). The LD_{50} was greater than 200 ug/bee when fosamine ammonium was dissolved in solvent and applied directly to bees (O'Neal, 1987, as cited in USDA Forest Service, 1989).

The wildlife risk analysis is in Table 2-12.

Table 2-12
Wildlife risk analysis for fosamine ammonium

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	14	244	1000	5000	bobwhite
Mammals	25	517	1476	7380	guinea pig
Amphibians	14	242	1000	5000	bobwhite
Reptiles	19	310	1000	5000	bobwhite

[1] The common laboratory test species used to represent the group of wildlife species.

The risk analysis shows there is a wide margin of safety for wildlife species.

Aquatic Species

Fosamine ammonium is considered practically nontoxic to fish and invertebrates. All acute LC_{50} values are greater than 100 ppm.

Yolk-sac fry, fingerlings, and eggs of salmonids are not acutely sensitive to fosamine ammonium (USDA, 1984). Ninety-six-hour EC_{50}'s based on avoidance behavior and white blood cell counts in coho salmon also are greater than 100 ppm (USDA, 1984). McLeay and Gordon (1980) worked extensively with fosamine ammonium and coho salmon and rainbow trout at several life stages. They reported yolk-sac fry were the most sensitive with LC_{50} values of 618 mg/liter for coho and 367 mg/liter for rainbow trout. Lorz et al. (1979) found a no-effect level for yearling coho of 2,000 mg/liter. The 48-hour LC_{50} for Daphnia is 1524 mg/liter according to research cited by Ghassemi et al. (1982).

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by EPA where the risks of adverse effects to fish or invertebrates as shown in Table 2-13.

Table 2-13
Definition of "Q" value

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-14.

Table 2-14
Aquatic species risk analysis for fosamine ammonium

		Off-Site Drift			Accidental Direct Spray	
	Critical LC_{50}	Typical	Maximum	Spill in Pond	Typical	Maximum
PPM ----------Q Value----------						
Fosamine ammonium	367	<0.1	<0.1	<0.1	<0.1	<0.1

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to any of the representative aquatic species for typical and maximum exposures to fosamine ammonium resulting from off-site drift, spill in a pond or accidental direct application to surface water. All Q values are less than 0.1.

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

Fosamine's low vapor pressure largely eliminates vaporization, meaning it should not appear in the air except as spray droplets during application. The investigator concludes there is no environmental risk from fosamine in air.

Soil and Groundwater

Fosamine ammonium is a very low mobility herbicide and is not readily leached from soil. Because of this there is little likelihood of groundwater contamination. There is no data suggesting environmental concentrations of fosamine ammonium cause adverse effects on soil

organisms, hence there should be no impact on long-term soil productivity. The investigator concludes there is no environmental risk from fosamine ammonium in soil.

PART 4. RISK ASSESSMENT FOR GLYPHOSATE

Introduction

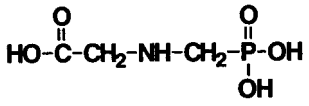
Glyphosate is a non-selective herbicide that controls most annual and perennial weeds and is very effective in controlling annual grasses. Currently, there are about 35 commercial formulations including Roundup®, Accord®, and Rodeo®. Formulations range in concentration from 147 g a.e./L (1.23 lb a.e./gal) to 570 g a.e./L (4.75 lb a.e./gal). Glyphosate formulations are commonly applied to foliage by backpack for spot spraying or by boom or hydraulic spray equipment for broadcast spraying. Glyphosate is also a cut-stem application on woody plants. Application rates range from 1.4 to 6 kg a.e./ha (1.3 to 5.4 lb a.e./acre), or at 0.5% to 5% v/v or 360 g/L in a spray-to-wet application for general vegetation control in many non-crop sites, such as industrial sites.

There is an extensive literature on glyphosate as it relates to human health and environmental risk. It has been reviewed and summarized in several documents including USDA Forest Service (1984, 1989, and 1997), ExToxNet (1996), Geisy et al. (2000), Williams et al. (2000), USEPA (1993 and 2002a), and SERA (2002 and 2003a). The principal sources for the information in this report are SERA (2003a), USEPA (1993), USDA Forest Service (1989 and 1997), Geisy et al. (2000), and Williams et al. (2000). It is not the purpose of this review and assessment to repeat the analyses of these reports but rather to provide a brief discussion of environmental and wildlife risk associated with ROW vegetation management, and where appropriate to include recent work that may not be discussed in the major reviews.

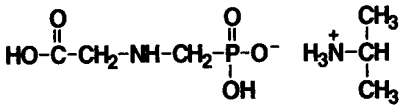
Physical-Chemical Properties and Environmental Behavior

Physical-Chemical Properties

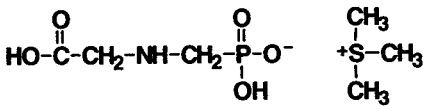
Glyphosate is the common name for N-(phosphonomethyl) glycine. Except for Touchdown® produced by Zeneca, which is formulated as the trimethylsulfonium salt, all other commercial products are formulated as the isopropylamine salt of glyphosate.



Glyphosate Acid



Glyphosate Isopropylamine Salt



Glyphosate Trimethylsulfonium Salt

Table 2-15
Physical and chemical properties of glyphosate

Property	Value
Water solubility	12 g/L @ 25C, acid; 900 g/L, amine salt
Vapor pressure	< 7 x 10^{-9} mm Hg @ 25C
Log K_{ow}	-3.39 @ pH 1.77; -4.38 @ pH 4.61; -4.85 @ pH 6.86; -4.14 @ pH 9.00
K_{oc}	554–34,000 L/kg

From SERA (2003a)

Some formulations contain surfactants that are more toxic to aquatic animals than glyphosate itself. The concentration of surfactants in glyphosate formulations tends to range from about 1% to 11%. The most common class of surfactants is characterized as polyoxyethyleneamines (POEA). Technical grade glyphosate contains an impurity, N-nitrosoglyphosate (NNG), but the amount of this impurity in glyphosate is less than one part per million and has been classified as toxicologically insignificant by the USEPA (SERA, 2003a). 1,4-dioxane can also be an impurity. The upper limit of this compound in Roundup® is about 10 ppm. The specific identity of the surfactants, other inerts, contaminants, and impurities has been disclosed to the USEPA as part of the registration process and this information was reviewed in the SERA (2003a) risk assessment. The toxicological implications of these constituents and impurities of formulation are discussed in the hazard analysis below.

Environmental Behavior of Glyphosate

Glyphosate is an effective herbicide for a broad-spectrum of plant species, yet it has short persistence in soil and water and has relatively low toxicity to animal species. In a recent review, SERA (1996b) summarizes relevant information on glyphosate relevant to the risk assessment process. Glyphosate is taken up by the plant primarily through the foliage and is then translocated through the plant to roots and other structures. Of the glyphosate in the soil, less than 1 percent is absorbed through the roots, primarily because glyphosate is quickly and tightly adsorbed to soil particles. Glyphosate is not metabolized in plants. Complete and rapid degradation occurs in both soil and water by microbiological activity but not by chemical activity (Rueppel et al., 1977; Ghassemi et al., 1981).

Newton et al. (1994) summarized the results from studies of the movement, persistence and fate of glyphosate and AMPA in three forests (Oregon, Georgia and Michigan). They reported eight-

hectare residual stands of low-quality hardwoods were treated with 4.12 kg/ha glyphosate ae applied aerially in late summer. Residues were highest in upper crown foliage. Overstory reduced exposure of understory vegetation and streams. Residues in streams decreased to the detection limit or were undetectable in 3-14 days. Residues in soils were highest where cover was sparse and where litter was removed. No residues were detectable in soil 409 days after treatment; movement below 15 cm was negligible. AMPA appeared at low levels in all degrading matrices, including sediments, soon after deposition of glyphosate. In pond sediments, both glyphosate and AMPA remained bound and inactive. Residue concentrations in foliage, water, and soil were below levels known to be biologically active in nontarget fauna.

Behavior in Air

Glyphosate has a low vapor pressure so it does not volatilize into the atmosphere (Ghassemi et al., 1981; U.S. Department of Energy, 1983). Rueppel et al. (1977) report that only a small amount of glyphosate should be expected to volatilize from either soil or water.

Behavior in Plants

Glyphosate is an effective herbicide for a broad spectrum of plant species. It is taken up by the plant primarily through the foliage and is then translocated throughout the plant, including through underground structures such as roots and rhizomes where it inhibits further growth and sprouting. Depending upon soil type and conditions, some root uptake may occur, but typically of the glyphosate in the soil, less than 1 percent is absorbed through the roots, primarily because glyphosate is quickly and tightly adsorbed to soil particles (USDA Forest Service, 1997). Generally, the half life of glyphosate in vegetation is expected to be 13 days or less, and for there to be levels of AMPA which are less than 0.1 ppm, and are non-detectable (less than 0.01 PPM) in less than 30 days.

Glyphosate is stable within the plant tissue and is not metabolized by plants. Residue levels of glyphosate and AMPA in fruits of wild blueberry and red raspberry were determined in field tests in Ontario, Canada, by Roy et al. (1989b). Glyphosate was applied using a hand held spray boom to achieve glyphosate application rates which were approximately 2 kg/ha close to the time when fruit would be approaching maturity and be ready for harvest. Ripe fruits were collected for chemical analysis at various times after application. Blueberry absorbed about 14 % of the glyphosate that fell on the fruits, while red raspberry absorbed about 9%. Washing the fruit with water removed the balance (Table 2-16).

Table 2-16
Residue levels of glyphosate and ampa in blueberry and red raspberry fruits collected from boreal forests of Ontario

		Glyphosate (mg/kg)	AMPA (mg/kg)
	Days postspray[a]	**Mean +SD**	**Mean +SD**
Blueberry	0[a]	7.94+0.678	ND1
	1	6.60+0.708	ND
	2	5.66+1.210	0.055+0.017
	13	3.73+0.535	0.051+0.009
	20	2.50+0.524	0.031+0.010
	33	1.23+0.248	ND
	61	0.19+0.035	ND
Raspberry	0	19.49+2.110	ND
	1	18.25+2.570	ND
	2	17.12+0.990	0.102+0.024
	13	5.55+0.880	0.089+0.031
	20	3.39+0.420	0.033+0.0088
	33	1.22+0.122	0.024+0.004
	61	NA2	NA

Note: Adapted from Table 5, Roy et al. (1989) Data corrected for glyphosate and AMPA recovery efficiencies.
[1]ND, not detectable; limits of detection for glyphosate and AMPA were 0.025 and 0.01 ppm, respectively.
[2]NA, not available; no berries on the plantation.
[a]Zero days post spray was August 8, 1985.

As expected, highest residue levels occurred the day of application when glyphosate residue level in the washed fruit was approximately 8 mg/kg in blueberry, and 19.5 mg/kg in red raspberry. Residue levels declined with time. Residue levels of AMPA were small and transitory.

These data probably provide an overestimate of the exposure of organisms that might consume the fruits similar to these from treated portions of rights-of-way, because berry plants are considered desirable vegetation on rights-of-way, and are not normally treated. However, it is possible that some treatment could occur in connection with the application of herbicide to nearby tall growing vegetation, and the pattern of dissipation observed in this study is relevant for this risk assessment. The risk of this is greatest with the high volume foliar application methods. Low volume foliar applications would result in less exposure to such plants and the basal and cut surface treatment methods should prevent such exposure of berry plants.

In British Columbia where glyphosate was aerially applied for reforestation purposes, the residue level of glyphosate on foliage of two species immediately after application was 261 and 448

mg/kg. Residue levels were not followed in the foliage over time, except as the foliage senesced and fell from the plant. The initial residue levels found in this material (i.e., in the first foliage to fall from the plants) was 12 to 20 ppm, declining to less than 1 ppm in 45 days. In an Oregon forest following aerial application of glyphosate, initial herbicide levels in foliage ranged from 80 to 489 ppm in the foliage most exposed to the spray material. In understory vegetation (that least likely to be directly sprayed on the right-of-way), the residue level immediately after treatments ranged from 20 to 1818 ppm. Levels of AMPA did not exceed 2 ppm at any time. The residue levels declined rapidly with time in the foliage, regardless of the initial herbicide residue levels. The half-life values were reported to range from 10 to 27 days in plants.

Behavior in Soil and Groundwater

Soil

When glyphosate reaches the soil (the majority of the herbicide lands on vegetation and is absorbed by the plant tissue), it is completely and rapidly degraded by microbiological activity. Within the soil environment, it is resistant to chemical degradation, stable to sunlight, is relatively non-leachable, has no tendency to leach through the soil or to run off, is strongly adsorbed to soil particles, has negligible volatility, and has a minimal effect on soil microflora. (Rueppel et al., 1977; WSSA, 1983).It does not accumulate in the soil (USDA Forest Service 1989 - Vol. II, pp. 4-25 through 4-27). Soil microflora degrade glyphosate to aminomethyl phosphonic acid, which is somewhat more stable than glyphosate. Principal decomposition end products are carbon dioxide, water, nitrogen and phosphate. Decomposition occurs under both aerobic and anaerobic conditions. Glyphosate has an average half-life of 60 days in the soil. The half-life is shorter than average in silt loam soils and longer than average in sandy soils.

Sacher (1978) compared $^{14}CO_2$ evolution over a period of seven days in sterile and non-sterile conditions with radiolabeled glyphosate and with radiolabeled sucrose (sugar) as a control. Minimal amounts of $^{14}CO_2$ evolved, along with lack of significant changes in composition of the radiolabeled glyphosate compounds under sterile conditions, demonstrated that chemical degradation is not a major means of glyphosate degradation. The importance of biodegradation by soil microflora was indicated in the non-sterilized conditions, in which as much as 55 percent of the $^{14-}$C-labeled glyphosate was given off as $^{14}CO_2$ within 4 weeks using Lintonia Sandy Loam soil. Microbial degradation of glyphosate can occur under both aerobic and anaerobic conditions and the same general distribution of glyphosate metabolites is found under both aerobic and anaerobic conditions (Rueppel et al., 1977).

The primary metabolite of glyphosate is aminomethyl phosphonic acid (AMPA). Studies with radiolabeled AMPA in silt loam and silty clay loam soils resulted in 34.8 and 16.1 percent of the applied ^{14}C being given off as $^{14}CO_2$ within 63 days, respectively (Rueppel et al., 1977). Degradation of AMPA is generally slower than that of glyphosate, possibly because AMPA may adsorb onto soil particles more strongly than glyphosate and/or because it may have a lower permeability through the cell walls or membranes of soil microorganisms.

Glyphosate is relatively immobile in most soil environments as a result of its strong adsorption to soil particles. This tendency of glyphosate to adsorb to soil particles serves as the initial stage in the inactivation of glyphosate with respect to plant uptake--the adsorbed glyphosate is unavailable for uptake by plant roots.

Adsorption of glyphosate to soil particles begins immediately after application, and binding occurs with particular rapidity to kaolinite, illite, and bentonite clays as well as to muck. Organic matter and clays saturated with Fe^{+++} and Al^{+++} tend to adsorb more glyphosate than do organic matter and clays saturated with Na^{+} or CA^{++}. The prime factor in determining the amount of glyphosate adsorbed to soil particles is the soil phosphate level, and it appears that glyphosate is bound to soil through the phosphonic acid moiety (Sprankle et al., 1975a).

The effect of both pH and phosphate on glyphosate adsorption was examined by Sprankle et al. (1975b) based on plant yield as an indicator of glyphosate unavailability due to adsorption. They found no significant difference in plant yield as an indicator of glyphosate unavailability due to adsorption. However, there was a decrease in glyphosate adsorption as the pH increased and the level of phosphate increased. Salazar and Appleby (1982), however, found that in some high-organic soils, glyphosate applied to soil at a rate of 3 lb/acre (3.4 kg/ha) reduced bentgrass seedling growth for seeds germinating as long as 5 days after glyphosate application. Brewster and Appleby (1972 in Salazar and Appleby, 1982) found similar reduction in wheat seedling growth with pre-emergence application of glyphosate that appeared to be related to increased levels of moisture in the soil.

In general, glyphosate dissipates relatively rapidly when applied to most soils. As a result of greenhouse soil dissipation studies with three different soil types, Rueppel et al. (1977) calculated half lives from 3 to 130 days, the former for a silty clay loam with 6 percent organic content and the latter for a sandy loam with 1 percent organic content. These data have been substantiated by field studies, in which glyphosate had an average half-life of 2 months in 11 soils and by other studies reporting half-lives of 17 to 19 weeks in sandy soil and 3 weeks in silt loam (USEPA data reported in Ghassemi et al., 1981).

In a field study in British Columbia, Feng and Thompson (1990b) determined the persistence and movement of glyphosate in the soil after aerial application. They reported glyphosate and AMPA (a primary metabolite in the pathway of glyphosate decomposition) were retained primarily in the upper organic layers of the profile, with more than 90% of the total glyphosate residue in the 0 - 15 cm layer. They conclude there is a low propensity for the leaching probably as a result of the strong adsorption and the tendency for decomposition. Glyphosate soil residues dissipated as function of time, with a half-life of 45 to 60 days. After 360 days, total soil residue levels were 6 to 18% of the initial levels. In an Oregon forest ecosystem, Newton et al. (1994) reported the half life of glyphosate ranged from 10 to 27 days in foliage and forest floor material, and roughly twice as long in the soil.

In a Canadian boreal forest, Roy et al. (1989a) studied the degradation, persistence and mobility of glyphosate under field conditions after application of herbicide (2 kg/ha, active ingredient) to areas with either a sand soil or a clay soil. The sand site was used to study leaching and persistence, and the clay site was used to assess lateral mobility due to runoff. Except on one

sampling date, 100% of the herbicide was found in the upper organic layer throughout the 335 days in which detectable residues were found. On one date, 95% was in the organic layer, and 5% was in the next layer down. The herbicide level in the soil decreased 50% in 24 days and 90% in 78 days. There was no detectable lateral movement down the 8 degree slope at any time after application (762 day sampling period).

Groundwater

The Monsanto Company (USEPA data reported in Ghassemi et al., 1981) conducted soil column leaching studies in which glyphosate was aged for 30 days in soil columns and then eluted for 454 days with 1/2 acre-inch water. Leaching of parent compound was found to be insignificant.

Well water was monitored at three electrical substations in Newfoundland, Canada to determine the extent of glyphosate off-target movement after gravel platforms around substations were sprayed with 2% solution of glyphosate. Smith et al. (1996) reported that the levels of glyphosate at two substations were below the limits of detection throughout the duration of the study (32 weeks). Both substations had fragipan constricting layers below the surface, restricting water movement from the surface. The third substation was on a rapid to well-drained site and glyphosate levels were detected in well water. Levels peaked two weeks post-spray at 0.25 mg/l and at 37 weeks were at 0.13 mg/l. Albrechtsen et al. (2001) examined degradation of herbicides in shallow aquifers and reported that glyphosate showed little degradation under anaerobic conditions in water, but as a practical matter there is expected to be little or no glyphosate entering aquifers because of the tendency to bind to soil organic matter and its subsequent rapid microbial degradation.

The behavior of glyphosate in soil has been tested in a wide range of environmental conditions, bracketing those found on typical ROW sites. Based on these studies, the investigator expects the soil half-life of glyphosate on such sites to be less than 60 days, with half-life persistence in forest floor estimated to less than 30 days. Glyphosate has not been found to leach in soil, and is not expected to do so on typical ROW sites, thus there is no likelihood of groundwater contamination.

Behavior in Surface Water

In aquatic systems, glyphosate is strongly adsorbed to both organic and mineral matter and is degraded primarily by microorganisms. However, the rate of degradation of glyphosate in water is generally slower than it is in most soils because there are fewer microorganisms in water than in most soils (Ghassemi et al., 1981).

Stream samples of flowing canal water were taken by Comes et al. (1976) following metering of Roundup® at a rate of 150 ppb of glyphosate into the canals. Sampling 1 mile (1.6 km) downstream accounted for only 70 to 72 percent of the total glyphosate introduced into the stream. Subsequent downstream disappearance of glyphosate diminished and 5 to 9 miles (8 to 14.4 km) downstream, only 57 to 58 percent of the glyphosate remained in the water flow. These figures are for herbicide metered directly into flowing water. Sacher (1978) reports that when

glyphosate is applied to ditch banks at 150 ppb concentration, maxima of only 10 and 3 ppb glyphosate could be expected in streamflow 1 and 5 miles (1.6 and 8 km) downstream, respectively.

In a pond at Fort Lauderdale, Florida, Sacher (1978) reported a half-life for glyphosate of approximately 12 days. Studies by Monsanto Company (USEPA data reported in Ghassemi et al., 1981) on glyphosate persistence in natural water bodies found a half-life of 7 weeks in Sphagnum bogs at pH 4.23; of 9 weeks in cattail swamps at pH 6.25; and of 10 weeks in pond water at pH 7.33.

Leaching of residues from irrigation canal banks treated with glyphosate was investigated by Comes et al. (1976). Neither glyphosate nor its metabolite, aminomethyl phosphonic acid, were detected in the first flow of water through canals that had been dry for 23 weeks after glyphosate had been sprayed on the ditch banks at a rate of 5 lb/acre (5.6 kg/ha). Soil samples collected from the canal bed the day before the canals were filled indicated residual levels of both glyphosate and aminomethyl phosphonic acid (0.35 and 0.78 ppm, respectively) in the top 10 centimeters of the soil along the ditch banks. The authors concluded that glyphosate could be applied to ditch bank vegetation in the fall after draining canals, with little to no chance of glyphosate residues contaminating irrigation water the following spring.

Inclined beds of three different soils were used to evaluate the runoff potential of glyphosate (Rueppel et al., 1977). Radiolabeled glyphosate was applied uniformly at a rate of 1.0 lb/acre (1.12 kg/ha) to the upper third of the soil bed and the surface then subjected to artificial rainfalls at days 1, 3, and 7 after treatment. The artificial rainfall was continued long enough for the collection of two consecutive 50 ml samples of runoff water and sediment. The ^{14}C activity of runoff water and sediment was measured. The results indicated a maximum runoff rate of less than 1.8×10^{-4} lb/acre (2×10^{-4}kg/ha), confirming that glyphosate binds tightly to soil particles.

Studies of the runoff of Roundup® applied to agricultural soils in Ohio were conducted by Edwards et al. (1980). Roundup® was applied as a pre-seeding treatment in early spring at rates of 1, 3, and 8 lb/acre (1.10, 3.36, and 8.96 kg/ha) and levels of glyphosate in the runoff water from natural rainfall were measured. The highest concentration of residual glyphosate, 5.2 ppm, was contained in runoff 1 day following treatment at the highest application rate. Glyphosate levels up to 2 ppb were detected in runoff from the highest application rate for up to 4 months following treatment. For watersheds treated at lower application rates, the highest level of glyphosate detected in runoff was 100 ppb for rainfall events 9 to 10 days after treatment. Two months following treatment, residual glyphosate levels in runoff had decreased to 2 ppb. Of the glyphosate applied to the soil, a maximum of 1.85 percent was removed by runoff transport and, of this amount, 99 percent was removed during the first rainfall event after herbicide application. Most of this loss is associated with soil erosion in agricultural watersheds. Because the forest floor and litter layer remain (usually) intact on herbicide-treated ROW, little or no soil erosion from herbicide treated ROW is expected to occur.

In British Columbia in connection with a forestry application (aerial), the fate of glyphosate in a forested watershed was determined. In some cases streams in the area were intentionally over-sprayed (no buffers), in others a 10 meter buffer was used (although the investigators, Feng,

Thompson and Reynolds (1990a) noted that the vegetation was dense around these streams and the buffers were indistinct from the air, suggesting some unintentional direct application to the stream also occurred). Buffered streams had very low residue levels (2 to 4 micrograms per liter [parts per billion, ppb]). Highest residues occurred in the stream directly and intentionally over-sprayed. The peak concentration measured was 162 micrograms per liter, which dissipated to less than 1 ppb in 96 hours. Water sampling conducted during the first storm after application (about 20 hours after application), showed peak herbicide concentration in the stream as less than 150 micrograms per liter, with residues declining to near 1 microgram per liter or less in less than 100 hours. Seven significant storm events were monitored subsequently; no quantifiable residues of glyphosate were detected in the next 150 days.

Newton et al. (1984) provided no buffer in an Oregon forest and aerially sprayed directly across a small stream. The peak herbicide residue level found in the stream was 270 micrograms per liter during the application, and declined rapidly thereafter. Residues were at or near the detection limit in less than 10 days, with this pattern continuing for the 55-day sampling period. Herbicide was adsorbed by stream sediments. The peak concentration in sediment was about 0.5 mg/liter, occurring about 14 days after application. While still measurable at 55 days, the residue level was decreasing. Adsorption on sediment and dilution with water movement likely produce the reduction in concentration in water observed with time in this study. There was no evidence of subsequent input to the stream due to leaching or surface runoff, despite the absence of a buffer.

Ramwell et al. (2002) studied the loss of glyphosate in surface runoff after being applied to roadsides. Glyphosate was applied at a rate of 1800 g a.i./ha to the road curb edge. After 5 mm of accumulated rainfall 28% (accumulated value) of glyphosate was lost from the roadside; after 25 mm of accumulated rainfall 35% (accumulated) had been lost. Ramwell et al. (2002) estimated that the maximum predicted surface water concentration for glyphosate was 51.80 µg/l.

Glyphosate dissipates rapidly from standing water, as in a pond. Goldsborough and Beck (1989) measured the concentration of glyphosate and AMPA in four small ponds in the boreal forest in Manitoba, Canada. They found a half life of 1.5 to 3.5 days. In microcosms containing only water, glyphosate persisted for longer periods. Microcosms with sediment and water showed rapid dissipation of the herbicide, suggesting adsorption is a major route of dissipation from water - reflecting the behavior of glyphosate in soil, where adsorption is also a dominant factor in its dissipation. Goldsborough and Brown (1993) returned a second year to reapply glyphosate to two of the ponds, and to apply it to a third pond that had not been treated previously. They found that glyphosate dissipated rapidly from the surface waters of all ponds (dissipation half-lives of 3.5-11.2 days). AMPA residues were detected in water samples during the first 14 days after treatment, suggesting that herbicide degradation was occurring in the water column. However, not all applied herbicide was accountable in residues in the water. Glyphosate and AMPA increased in sediment samples to day 36, indicating that sediment adsorption was a major sink for the herbicide.

Glyphosate in river water was found to rapidly degrade in river water with free and colloidal sediments (Zaranyika and Nyandoro, 1993) suggesting microbial degradation. The authors reported that over a study period of 72 days, there was an immediate 35% loss of glyphosate

(mixed solution was approximately 150 ppm glyphosate) due to adsorption to suspended sediment particles and deposition to the bottom sediment.

When buffer strips are used, glyphosate residues in water are expected to be less than 0.05 ppm and to persist in measurable amounts for less than 1 week. During storms occurring within the first 30 days after application, the maximum concentration of glyphosate expected in perennial stream water is less than 0.02 ppm, with persistence of less than 1 week.

Residues in Animals and Fish

Glyphosate taken into the body does not accumulate in any tissues and is excreted rapidly, unchanged. Game or fish taken immediately after exposure may contain very low residues, but possible intake is orders of magnitude below levels of concern.

In bluegill fish exposed to 0.612 ppm radiolabeled glyphosate for 28 days, there was an accumulation of residue in edible portions of the fish with a bioconcentration factor of 1.6. Maximum residue levels in channel catfish of 0.55 ppm were found 7 days after exposure to 10 ppm glyphosate for 14 days. In the same study, largemouth bass and rainbow trout had maximum residue levels of 0.12 and 0.11 ppm, respectively. Rainbow trout exposed to varying concentrations of glyphosate for 12 hours had no detectable residues of glyphosate or its primary metabolite, aminomethyl phosphonic acid. However, trout exposed to Roundup® at 2.0 ppm for 12 hours were reported to have 80 ppb of glyphosate in fillets and 60 ppb of glyphosate in eggs (Folmar et al., 1979). Note: In the original publication the tissue concentration was reported as 80 ppm, which was a misprint, later corrected. There were no detectable residues of glyphosate in midge larvae exposed to 2.0 ppm. The bioconcentration factors are believed to be between 0.1 and 0.3 (USDA Forest Service, 1997).

Bioconcentration factors (BCF) for carp and tilapia fish were reported by Wang et al. (1994) using radioactively labeled and unlabeled glyphosate over a 14 day period. BCF factors for carp exposed to 0.5 and 0.05 ppm of glyphosate ranged from 11.3 (lowest on day 3) to 42.3 (peaked at day 7) and 10.0 (day 1) to 33.6 (peaked at day 7), respectively; for tilapia exposed to 0.5 and 0.05 ppm of glyphosate BFCs ranged from 12.9 (lowest on day 2) to 65.5 (peaked at day 5) and 12.0 (day 0.5) to 35.4 (peaked at day 3), respectively.

In an Oregon forest, the highest concentration (5 ppm) was found one day after application in the viscera of deer mice, an omnivore. The concentration fell rapidly with time, to 0.37 ppm at one week, 0.17 ppm at 2 weeks and not detectable at 4 and 7.8 weeks. No resides of glyphosate exceeded 0.5 ppm in the body (minus the viscera) of any animal in this study. Fish from the stream, which was directly over-sprayed in this study, did not contain detectable residues of glyphosate or AMPA, indicating it has little tendency to bioaccumulate despite its detectable presence in the water for 3 days, and a source of glyphosate and AMPA in the sediment for 55 days (USDA Forest Service, 1997).

Wildlife Hazard Analysis

There is an extensive regulatory and published literature describing the toxicology of glyphosate, its behavior in the body and other information that relates to wildlife risks. This mass of information has been reviewed in several recent documents, including the human health risk analysis (EPRI, 2003).

Avian and Terrestrial Species

Glyphosate is generally recognized to be of a low toxicity in the environment (USDA Forest Service, 1984; SERA, 1996b). Acute oral LD_{50}'s are >5,000 mg/kg for the rat and 3,800 mg/kg for the rabbit (USEPA, 1984a; USDA Forest Service, 1984; Monsanto, 1983). Based on these values, glyphosate can be considered slightly toxic.

Oral LD_{50} values for the Roundup® and Rodeo® formulations in rats are 5,400 mg/kg and greater than 5,000 mg/kg, respectively (Monsanto, 1983, 1985a,b). Glyphosate, The NOELs derived from chronic feeding studies in rats are 362 mg/kg/day for males and 457 mg/kg/day for females. In a 1-year oral study with dogs, a NOEL of 500 mg/kg/day (HDT) was determined (USEPA, 1987). Studies conducted on black-tailed deer in pens in the Pacific Northwest showed no gross adverse health effects caused by the use of glyphosate for vegetation management (Sullivan, 1985). Glyphosate-treated browse and commercial chow were as acceptable for consumption by deer as untreated food. Likewise, glyphosate-induced weed and shrub control did not adversely affect deer use of treated habitat areas for at least the first year after treatment. Moose were found to browse preferentially in untreated areas of clearcuts treated with glyphosate (Santillio, 1994). This would likely minimize their exposure to glyphosate.

In a study to evaluate the direct effects of glyphosate on small mammals, no adverse effects on reproduction, growth, or survival were observed in populations of deer mice during the year following treatment (Sullivan, 1985).

Glyphosate is slightly toxic to birds based on the acute oral LD_{50} of greater than 4,640 mg/kg in bobwhite quail and mallard duck (Monsanto, 1983). The 8-day dietary LC_{50} is more than 4,000 ppm for both mallard ducks and bobwhite quail (USEPA, 1986b). Avian reproduction studies yielded no reproductive effects at dietary exposure levels of up to 1,000 ppm (USEPA, 1986b).

Perkins et al. (2000) used teratogenic responses in testing the toxicity of glyphosate to the frog. They found that Rodeo® (glyphosate IPA amine salt with no surfactant) was the least toxic with LC_{5} and LC_{50} values of 3,779 ppm and 5,407 ppm respectively. This compares to LC_{5} and $LC_{50\ for}$ values of 6.4 ppm and 9.4 ppm for Roundup® (glyphosate IPA salt plus a surfactant). These results reflect a similar pattern of toxicity reported in fish. Following a direct application at the highest permitted application rate to 15 cm-deep water (calculated concentration was 2.8 ppm) they calculated the margins of safety to be 2 and 1,312 for Roundup® and Rodeo®, respectively.

Yokoyama and Pritchard (1984) evaluated the effect of glyphosate on the mortality, fecundity and egg viability of an insect that is a common predator of other insects. They found no detectable effect, and concluded deleterious long-term effects on predator populations would not

occur if the insects survived the initial application (this latter comment reflecting the fact they were also testing some common insecticides). Glyphosate is relatively nontoxic to insects based on the 48-hour acute toxicity of greater than 100 ug/bee in honey bees (USEPA, 1986b).

Acute toxicity survival tests for frog tadpoles native to Argentina (*Scinax nasicus*) was done using a glyphosate formulation (48% glyphosate as the isopropylamine salt plus inerts) at concentrations of 3.07, 3.84, 4.8, 6 and 7.5 mg/L. The 24-h LC_{50} was 4.78; 48-h LC_{50} was 3.62; 72-h LC_{50} was 3.23; and the 96-h LC_{50} was 2.64 (Lajmanovich et al., 2003). Additionally, larval development aberrations (craniofacial and mouth deformities, eye abnormalities, and bent curved tails) occurred in all tests and increased with time and concentration (Lajmanovich et al., 2003).

The wildlife risk analysis is in Table 2-17.

Table 2-17
Wildlife risk analysis for glyphosate

Species Group	Dose Estimate		Critical Toxicity Values		Representative Laboratory Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	19	230	928	4640	quail
Mammals	32	458	760	3800	rabbit
Amphibians	19	230	928	4640	quail
Reptiles	25	292	928	4640	quail

[1] The common laboratory test species used to represent the group of wildlife species.

This analysis shows exposure levels that are far less than the critical toxicity value of 1/5 $LD_{50.}$ The risk analysis shows there is a wide margin of safety for wildlife species.

Aquatic Species

Technical glyphosate is only slightly to practically nontoxic to fish and invertebrates (Table 6-12 in USDA Forest Service, 1989). Studies with channel catfish, bluegill, rainbow trout, and largemouth bass indicate that glyphosate does not bioaccumulate in fish to any significant degree, as expected form its high solubility in water and low solubility in fat solvents. The toxicity of glyphosate or glyphosate-formulations to amphibians has not been reported in the literature. An maximum acceptable toxicant concentration (MATC) of greater than 257 ppm has been reported in a long-term study with fathead minnows. A 21-day study with *Daphina magna* determined a NOEL of 50 ppm based on decreased reproduction (Monsanto 1982, 1983).

Toxicity of glyphosate to three salmonid species (juveniles) in three water types was determined by Wan et al. (1991). For soft, intermediate, and hard waters, the 96-h LC_{50} for coho salmon was

30, 31, and 13 mg/L, respectively; for pink salmon 31, 10, and 14 mg/L, respectively; and for rainbow trout 31, 31, and 14 mg/L, respectively. Wan et al. (1989) reported similar values for soft, intermediate, and hard waters, the 96-h LC_{50} for coho salmon was 32, 27, and 13 mg/L, respectively; for pink salmon 33, 19, and 14 mg/L, respectively; and for rainbow trout 33, 18, and 14 mg/L, respectively.

Hildebrand et al. (1982) reported 96-hr LC_{50} values for rainbow trout exposed to glyphosate in the in the laboratory and field to be similar: 54.8 and 52.0 mg/L, respectively. Avoidance-preference data collected by the authors indicated that fish would avoid lethal levels of glyphosate. Operational application of glyphosate at the recommended field dose or 2.2 kg a.e./ha, as well as 10× and 100× field dose did not result in trout mortality in field streams.

Mitchell et al. (1987) reported the 96-hr LC_{50} for rainbow trout exposed to the isopropylamine salt of glyphosate (Roundup®) as 7.4 mg/L in lake water and 12 mg/L in dechlorinated municipal water. In dechlorinated municipal water, 96-hr LC_{50} for chinook and coho salmon were 9.6 and 11 mg/L, respectively. For the isopropylamine salt of glyphosate (Rodeo®) with a surfactant, the values for rainbow trout, chinook, and coho salmon in dechlorinated municipal water were 130, 140, and, 120 mg/L, respectively.

Servizi et al. (1987) reported the 96-hr LC_{50} for sockeye salmon fingerlings and fry exposed to glyphosate isopropylamine salt plus surfactant as 26.7/27.7 and 28.8 mg/L, respectively; coho salmon fry, 42.0 mg/L; and rainbow trout fry, 25.5/28.0 mg/L.

Abdelghani et al. (1997) reported the adjusted LC_{50} (values adjusted for density and % a.i.) for channel catfish as be 5.5 mg/L at 48-hours and 4.9 mg/L at 96-hours; the adjusted LC_{50} for bluegill sunfish was determined to be 4.5 and 4.4mg/L, respectively; and the adjusted LC_{50} for crawfish was 323650.0 and 21632.8 mg/L, respectively.

There are three different commonly used formulations of glyphosate: Roundup®, Rodeo®, and Accord® (see SERA, 1996b for a review of these three formulations). Because of the content of its surfactant, Roundup® is more toxic to aquatic organisms than the other two formulations. Therefore, the risks of different formulations are discussed separately. The discussion includes toxicity information for both fish and other aquatic species

Roundup®

The toxicity of the Roundup® formulation (41 percent isopropylamine (IPA) salt of glyphosate, 15 percent surfactant, and 44 percent water) to aquatic organisms is summarized in Table 6-12 of USDA Forest Service, 1989. Roundup® is moderately to slightly toxic; most 96-hour LC_{50} values range from 2 to 18 ppm. The acute toxicity of Roundup® is greater at pH 7.5 than pH 6.5, and toxicity also increases with increasing temperature (Folmar et al., 1979). Rainbow trout did not exhibit avoidance behavior at concentrations up to 10 ppm, whereas mayfly nymphs showed avoidance behavior at this level (Folmar et al., 1979).

Grande, Anderson and Berge (1994) reported on the toxicity of glyphosate (Roundup®) to fish. They found the 96 hour LC_{50} for brown trout was 4.5 ppm. Their graphic data shows a no mortality effect level at about 3 ppm.

Rainbow trout were exposed for 12 hours to 0.02, 0.2, and 2.0 ppm of formulated Roundup® (Folmar et al., 1979). No effects were observed on fecundity or maturation of gonads after being held in freshwater for 30 days. Midge larvae also were exposed to 0.02, 0.2, and 2.0 ppm of Roundup®. Significant increases in stream drift of the larvae were observed at the highest concentration.

Rodeo® and Accord®

The Rodeo® formulation (53.5 percent isopropylamine salt of the active ingredient N-phosphonomethyl glycine and 46.5 percent water) of glyphosate is practically nontoxic to aquatic organisms (Table 6-12 in USDA Forest Service, 1989). The 96-hour LC_{50}'s for fish are all greater than 1,000 ppm, and the 48-hour LC_{50} for *Daphnia magna* is 930 ppm (Monsanto, 1983). The Rodeo® formulation of glyphosate had a 24 hour EC_{50} of 5900 and a 48-hour EC_{50} of 5600 ppm to midge larvae (Buhl and Faerber, 1989). The toxicity of the Accord® formulation (41.5 percent IPA salt and 58.5 percent water) is expected to be similar to Rodeo® because both of the products have the same active ingredient and have water as the only inert ingredient.

Aquatic Risk Assessment

Solomon and Thompson (2003) did an exhaustive analysis of risk to aquatic organisms based on three model exposure scenarios, including ponds, flowing streams and estuaries where tidal action occurs. They included Roundup® (includes surfactant) and Vision® (equivalent to Roundup® used in forestry in Canada) at application rates of 1 kg/ha to 8 kg/ha. They concluded that even with over-water applications (for instance in wetlands) the risk to aquatic organisms is negligible or small at application rates less than 4 kg/ha.

The California Dept. of Fish and Game (CDPR, 1997) published a preliminary hazard assessment for glyphosate to aquatic organisms. Their report is also a good compilation of toxicity but is based on a hypothetical concentration in water ranging from 0.18 ppm to 1.84 ppm following direct application of Rodeo® to surface water. They conclude that Rodeo® should not pose a hazard to aquatic organisms when used according to the label. Roundup was found to be moderately to slightly toxic to fish and invertebrates and its use near aquatic sites would be less desirable.

The risks of adverse effects from exposure to glyphosate were estimated for representative aquatic species. In cases where no acute toxicity reference value was available for a representative species, a value was selected from the table of the most closely related species.

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria

proposed by U.S. EPA where the risks of adverse effects to fish or invertebrates are described in Table 2-18.

Table 2-18
Definition of "Q" value

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-19.

Table 2-19
Aquatic species risk analysis for glyphosate

		Off-Site Drift			Accidental Direct Spray	
Herbicide	Critical LC_{50}	Typical	Maximum	Spill in Pond	Typical	Maximum
PPM ----------Q Value----------						
Glyphosate (Accord®)	600	<0.1	<0.1	<0.1	<0.1	<0.1

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to aquatic species for typical and maximum exposures resulting from off-site drift, spill in a pond or accidental direct spray on surface water. All Q values are less than 0.1.

Effects on other aquatic organisms

Gardner and Grue (1996) reported that glyphosate applied to wetlands in Washington state was not associated with significant decreases in survival or growth of bioassay organisms, with the exception that growth of duckweed (*Lemna gibba*) was reduced 48 hours after exposure.

Peterson et al. (1994) also determined the phytotoxicity of glyphosate to aquatic organisms when applied at a the label rate of 4.272 kg/ha (expected environmental concentration of 2.848 mg/L) and reported that glyphosate at the expected environmental concentration was highly toxic to two diatom algal species and the nitrogen-fixing cyanobacterium species, but was relatively non-toxic to the other algal species and duckweed (*Lemna minor*).

Phytotoxocity of technical glyphosate to an aquatic macrophyte, *Myriophyllum sibiricum,* was studied under laboratory conditions to determine inhibitory concentration 25 and 50 ppm [IC_{25}

and IC_{50}, concentrations (mg a.i./L) that inhibit an endpoint parameter by 25 and 50%, respectively]. Roshon (1997 as cited by Roshon et al., 1999) used the maximum label rate for technical glyphosate of 2.14 kg/ha (expected environmental concentration of 1.43 mg a.i./L). IC_{25} and IC_{50} for shoot growth were 1.28 and >2.99 ppm, respectively; IC_{25} and IC_{50} for number of roots were 2.355 and 2.798 ppm, respectively; and IC_{25} and IC_{50} for root length were 0.599 and 0.844 ppm, respectively. Roshon (1997) reported phytotoxic effects on roots at concentrations below the expected environmental concentration.

Two *Daphnia* species (freshwater zooplankton) were used to assess acute toxicity of a glyphosate formulation (isopropylamine salt and a surfactant).Dilutions of the glyphosate formulation used were: 18, 32, 54, 90, 150, and 250 mg a.i./l. All organisms tested presented immobility at the highest concentration (250 mg/l) at 24 hours of exposure; at 48 hours of exposure immobility of all organisms occurred above 150 mg/l (Alberdi et al., 1996). The EC50 values for *D. spinulata* were 94.87 ppm at 24 hours and 66.18 ppm at 48 hours; for *D. magna* values were 95.6 ppm at 24 hours and 61.72 ppm at 48 hours. The 96-hr EC_{50} for *D. pulex* exposed to the isopropylamine salt of glyphosate plus surfactant was 25.5 mg/L (Servizi et al., 1987).

See additional information about other aquatic species in the portion above that relates to specific formulations containing glyphosate.

It is clear that the possible concentration in surface water will be far less than the levels exhibiting toxic effects on other aquatic species. Therefore the investigator concludes there is no risk to such species.

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

Glyphosate has a low vapor pressure so it does not volatilize into the atmosphere (Ghassemi et al., 1981; U.S. Department of Energy, 1983). Rueppel et al. (1977) report that only a small amount of glyphosate should be expected to volatilize from either soil or water.

Soil and Groundwater

Several studies have shown that glyphosate does not adversely affect soil microorganisms or their metabolic processes. Muller et al. (1981) studied the effect of glyphosate application at 2.3 lb/acre (2.6 kg/ha) in a loam soil and a fine silt soil in Finland during actual agricultural applications. They concluded that glyphosate was degraded over the winter months even at the low prevailing temperatures and that glyphosate did not adversely affect nitrogen fixation, nitrification, or denitrification activity in these soils. The relatively benign nature of glyphosate

with respect to soil microorganisms is further confirmed by tests conducted by Monsanto personnel who compared microbial plate counts on untreated soil and soil treated with glyphosate (Sacher, 1978).

Mårtensson (1993) reported that N-fixation by free-living soil bacteria was significantly reduced in the laboratory when exposed to 100 times the recommended application rate (1-4 kg/ha) of glyphosate. The adverse effects were independent of soil pH, and C and N content of the soil. Nitrogen fixation by cyanobacteria was inhibited by 800 ppm of glyphosate, which corresponded to 200 times the recommended rate. The adverse effects on cyanobacteria N-fixation was independent of C and N content of the soil.

Ectomycorrhizal fungi often benefit the host plant through symbiotic associations with the roots of vascular plants, and are particularly important for the growth of conifer trees. Estok et al. (1989) found in lab studies using an agar media, that glyphosate significantly reduced three species of ectomycorrhizal fungus at concentrations ≥ 1000 ppm and growth was completely inhibited at concentrations ≥ 5000 ppm.

The behavior of glyphosate in soil has been tested in a wide range of environmental conditions, bracketing those found on typical ROW sites. Based on these studies, the investigator expects the soil half-life of glyphosate on such sites to be less than 60 days, with half-life persistence in forest floor estimated to less than 30 days. Glyphosate has not been found to leach in soil, and is not expected to do so on typical ROW sites, thus there is no likelihood of groundwater contamination. The concentrations required to produce adverse effects in these tests will not occur in soils associated with electric utility rights-of-way, therefore the investigator concludes there is no substantive risk to long-term soil productivity.

PART 5. RISK ASSESSMENT FOR IMAZAPYR

Introduction

Imazapyr is an imidazolinone chemical that is used as a broad spectrum, non-selective post-emergence herbicide (American Cyanamid Company, 1985a and 1985b). It is the active ingredient in Assault®, Arsenal AC®, Chopper® and Stalker® herbicides, which have demonstrated excellent activity, with residual control of a wide variety of annual and perennial grasses and broad-leaved weeds, as well as brush and deciduous tree species. Application rates range from 0.08 lb a.e./acre to 4 lb a.e./acre with a common rate being 0.15 lb a.e./acre.

Although imazapyr may be applied as an individual herbicide, it is often used in a mixture with triclopyr. While imazapyr can be used in pre-emergence applications, post-emergent are most effective especially when applications are made to actively growing plants. Common application methods include cut-surface, backpack (low volume foliar) and boom and hose and gun hydraulic spray (high volume foliar).

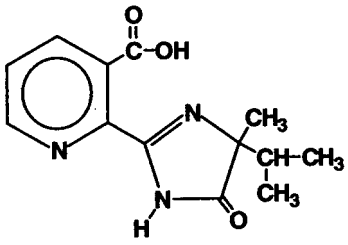
The principal information sources used in this report are: USDA Forest Service (1989); USDA Forest Service (1997); SERA (1999); and USEPA (2003). The sources are not specifically

quoted in the following discussion. Information on inert ingredients in imazapyr formulation is considered proprietary under the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) but was included in the risk assessment done by SERA (1999). It is not the purpose of this review and assessment to repeat the analyses of these reports, but to provide a brief discussion of environmental and wildlife risk associated with ROW vegetation management and, where appropriate, to include recent work that may not be discussed in the major reviews.

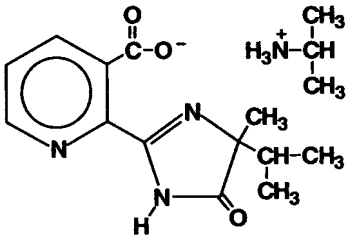
Physical-Chemical Properties and Environmental Behavior

Physical-Chemical Properties

The Arsenal®, Chopper®, and Stalker® formulations contain imazapyr at 2 lb a.e./gal and Arsenal® AC contains (4 lbs a.e./gal). All are formulated as the isopropylamine salt of imazapyr. Important physical properties of imazapyr are in Table 2-20. The herbicide also contains water, surfactant and other inert ingredients (American Cyanamid Company, 1984 and 1987a). The Arsenal® formulations are mixed with water, and Chopper® and Stalker® are emulsifiable concentrates to be mixed with an "oil" carrier and/or water.



Imazapyr Acid



Imazapyr Isopropylamine Salt

Important physical properties of imazapyr are in Table 2-20.

Table 2-20
Identification and physical/chemical properties of imazapyr and the isopropylamine salt of imazapyr

Property	Value
Solubility in Water	13.1 grams/L @ 25C
Solubility in Acetone	33.9 grams/L @ 25C
Solubility in Toluene	1.8 grams/L @ 25C
Vapor Pressure	$<10^{-7}$ mm Hg @ 45C
K_{ow} (acid)	1.3
K_{oc} (ml/g)	46 - 100

From SERA (1999) and WSSA (1994)

Environmental Behavior

Arsenal® (2-[4,5-dihydro-4-methyl-4-(l-methylethyl)-5-oxo-1H-imadazol-2-y1]-3-pyridinecarboxylic acid with 2-propanamine (1:1) salt) is an imidazolinone chemical that is used as a broad spectrum, post emergence herbicide (American Cyanamid Company, 1985a). Arsenal® has demonstrated excellent activity, with residual control of a wide variety of annual and perennial grass and broad-leaved weeds and many brush and deciduous tree species (American Cyanamid Company, 1985a, 1985b). It is registered for use in the United States by USEPA in non-cropland situations, such as industrial sites, fence rows, non-irrigation ditch banks, and rights-of-way.

Behavior in Air

The volatility of imazapyr is virtually nil, with a vapor pressure of less than 2 X 10^{-7} mm Hg at 45 degrees C. This means spray droplets are the primary source of imazapyr in air. Once droplets settle out, there should be no measurable residues in air.

Behavior in Plants

Imazapyr is a potent inhibitor of plant growth. At 63 g/ha (.06 lb/ac) Arsenal® severely inhibited the growth or killed all species tested. Differences in the tolerance of these nontarget species were noted, with sugar beets being the most susceptible and soybeans the most tolerant. Applications of Arsenal® at commercial rates result in reduced vegetative vigor of both target and non-target plant species found in the area. Seedling growth in these areas will be affected until the imazapyr has dissipated sufficiently (American Cyanamid Company, 1987b).

Arsenal® is readily absorbed through foliage and roots and is translocated rapidly throughout the plant in the xylem and phloem, with accumulation in the meristematic regions (American Cyanamid Company, 1987a; USEPA, 1985). Treated plants stop growing soon after spray

application. Chlorosis (loss of chlorophyll) appears first in the newest leaves, and necrosis (localized death of living tissue) spreads from this point (American Cyanamid Company, 1987a,b).

In perennials, Arsenal® is translocated into and kills underground storage organs, thus preventing regrowth. Chlorosis and tissue necrosis may not be apparent in some plant species until 2 weeks after application; complete kill of plants may not occur for several weeks (American Cyanamid Company, 1987b). The isopropylamine salt of imazapyr kills plants by reducing the levels of the amino acids valine, leucine, and isoleucine, through the inhibition of activity of acetohydroxy acid synthase, an enzyme common to the biosynthetic pathway of these amino acids (this pathway is unique to plants). This inhibition leads to interference in DNA synthesis and cell growth (American Cyanamid Company, 1987a).

Michael (1986) reported that the rate of dissipation of the isopropylamine salt of imazapyr from plant tissue followed a logarithmic curve over time. Initial large decreases of the residues from plants occurred as a function of mechanical loss by wind and then later from washoff by precipitation; subsequent losses were more gradual with time and were independent of storm events. Tissue concentrations were not significantly altered by storm events. The concentrations were approximately 10 mg/kg at both sites regardless of vegetation type, indicating a maximum absorbable tissue load of 10 mg/kg. Half-life of the isopropylamine salt of imazapyr in plant tissue ranged from 12 to 40 days; half-life in pine tissue was 40 days (Michael, 1986). A half-life for imazapyr in vegetation of 15 to 50 days is expected on ROW.

Imazapyr is confirmed as mobile in plants by a study with alligator weed. Tucker, Langeland and Corbin (1994) showed imazapyr was more readily absorbed by foliage than glyphosate. Additionally, more imazapyr was translocated to underground plant storage tissues than glyphosate. The authors conclude these factors together probably account for the resistance of alligator weed to glyphosate and its susceptibility to imazapyr.

Behavior in Soil and Groundwater

Aerobic and anaerobic metabolism of Arsenal® were evaluated in laboratory experiments in a sandy loam soil. Aerobic microbes in the soil slowly degrade Arsenal®. The aerobic half-life of Arsenal® under laboratory conditions was calculated to be 17 months. Carbon dioxide was the only major metabolite detected. Under anaerobic conditions, Arsenal® was not metabolized (American Cyanamid Company, 1986b).

A soil photolysis study for Arsenal® showed that at 1,680 g/ha (1.5 lb/ac) on a sandy loam soil, there was 11 percent degradation of imazapyr over the 28 days of continuous irrigation. There were at least five degradation products formed, none of which accounted for more than 10 percent of the applied dose. The half-life under the conditions of these tests was calculated to be 149 days (American Cyanamid Company, 1986a).

Field studies showed that loss of radioactivity was primarily from the top 3 inches of the soil profile, indicating that imazapyr has low potential for leaching. Imazapyr has low potential for bioaccumulation. The hydrolytic half-life of imazapyr at pH 7.0 was calculated to be 325 days,

indicating that hydrolysis is not a major route for environmental degradation. Michael (1986) found that the half-life of the isopropylamine salt of imazapyr in soil at a forest study site was 10 to 34 days.

Field dissipation of Arsenal® was studied in a coarse, low organic matter (<1.0%) soil in New Jersey. From day 0 to day 169, approximately 97 percent of the recovered carbon-14 residues were found in the top 6 inches of the soil profile, and 81 percent of that which was recovered from soil samples collected at day 357 was in the top 6 inches. Loss of radioactivity was primarily from the top 3 inches of the soil profile. During this study the site received a total of 43.3 inches of precipitation. The results indicate that Arsenal® has low leaching potential under field conditions receiving approximately 40 inches of precipitation (American Cyanamid Company, 1986a).

Additional field dissipation studies of Arsenal® were conducted in New York, Missouri, New Jersey, and Michigan. Arsenal® residues in the 0 to 4-inch, 4- to 8-inch, and 8- and 12-inch soil depth had declined to below the validated method sensitivity of 0.05 ppm at 92 and 96 days after treatment (DAT) in the Missouri and New York studies, respectively. In the 8- and 12- inch soil profile, apparent Arsenal® residues never exceeded 0.07 ppm during either study (American Cyanamid Company, 1986a).

In the New Jersey study, apparent Arsenal® residues in the 0 to 3-inch soil profile decreased to 0.15 ppm 33 days after treatment and less than 0.05 ppm 206 days after treatment. Apparent Arsenal® residues were less than 0.05 ppm in the 3- to 6-inch and 6- to 10-inch soil profiles. In the Michigan study, apparent Arsenal® residues in 0 to 4-, 4- to 8-, and 8- to 12-inch soil profiles were less than 0.05 ppm at the only sampling time, 173 days after treatment. The results of these four field studies demonstrated that Arsenal® declined to undetectable levels in the soil 3 to 7 months after application and did not leach to any significant depth in the soil profile (American Cyanamid Company, 1986a).

A bioassay was used in a study in Israel to evaluate the persistence patterns of imazapyr in two soils and under different soil moisture levels and temperatures (Ismail and Ahmad, 1994). These studies showed the decomposition was faster under higher moisture levels and at higher temperatures (Tables 2-21 and 2-22).

Table 2-21
Half-lives (days) of imazapyr in clay (C) and clay loam (CL) soil at different moisture levels. The standard bioassay was performed at ambient temperature (32°C).

Moisture levels (%)	Imazapyr	
	C	CL
20	29	11
50	16	10
80	12	10
LSD 0.05	2	2

Adapted from Ismail and Ahmad, 1994

Table 2-22
Half-lives (days) of imazapyr in clay (C) and clay loam (CL) soil at different temperatures

Temperature (°C)	Imazapyr	
	C	CL
25	22	19
30	16	10
35	7	5
LSD 0.05	3	5

Adapted from Ismail and Ahmad, 1994.

The fate of Arsenal® in forest watersheds after aerial application at a rate of 2,240 g/ha (2.0 lb/ac) showed the appearance of Arsenal® in only 1 percent of the samples taken below 30 cm (12 inches) (Michael, 1986). The 7 to 29 day half lives found by Ismail and Ahmad (1994) and the results of the study by Michael (1986) in the southeastern US, show imazapyr is rapidly dissipated from soil under the conditions expected on a forest-associated ROW. The investigator expects a soil half-life of imazapyr to be less than 30 days, with no detectable leaching, and therefore no contamination of groundwater.

Behavior in Surface Waters

Laboratory studies conducted with imazapyr (1 mg/L in deionized water) found that the herbicide is rapidly photodegraded in aqueous media and depending on pH; four photoproducts of imazapyr were obtained (Santoro et al., 1999). The half life at pH 3 was 3.5 days.

The fate of Arsenal® in water was evaluated in laboratory studies of hydrolysis and photolysis in aqueous media. The hydrolytic half-life of Arsenal® at pH 7.0 was calculated to be 325 days, indicating that hydrolysis is not a major route for the environmental degradation of Arsenal®. Under simulated sunlight in an aqueous media, Arsenal® photo degraded rapidly. The half-life of ^{14}C-labeled Arsenal® in distilled water, a pH 5.0 buffer solution and a pH 9.0 buffer solution was

calculated as 3.7 days, 5.3 days, and 2.5 days, respectively, with 12 hours of light exposure (American Cyanamid Company, 1986a; USEPA, 1985).

An anaerobic aquatic degradation study for Arsenal® showed that anaerobic sediment samples spiked with 0.34 mg (equivalent to 1,680 g/ha after treatment) of ^{14}C-labeled Arsenal® did not degrade at 0, 1, 2, and 3 months after treatment (American Cyanamid Company, 1986a).

After aerial application of Arsenal® for forest weed control, imazapyr was observed to move offsite in streams arising from the treated watershed. Observed movement occurred principally in storm runoff and dropped to near background levels within 40 days after treatment for the worst case studied. Water samples taken from the stream as a function of storm events showed little or no baseflow contamination, an indication that movement of imazapyr through the soil profile did not contribute to stream contamination by way of leaching (Michael, 1986).

The highest observed stream concentration occurred at a site where no attempt was made to observe a streamside management zone (SMZ), i.e. there was no buffer. Aerial application over the stream channel resulted in direct deposition of imazapyr in the stream. At the site where an attempt was made to observe an SMZ, imazapyr was not detected in the stream during application. One sample taken approximately 2 hours after completion of application contained 15 ppb of imazapyr. Subsequent samples did not contain quantifiable residues until the first post application precipitation. Most offsite movement at both sites occurred with the first two storm events. Differences in storm runoff concentrations between the two sites suggest maintenance of an SMZ can significantly reduce the amount of offsite movement of imazapyr in storm flow. There was no significant offsite movement by way of stream sediment from either site (Michael, 1986).

Imazapyr is subject to photolysis in aqueous media. Mallipudi et al. (1991) reported its dissipation in water when exposed to light followed first order kinetics, with half lives of 1.9 to 2.3 days in distilled water, 2.7 days in pH 5 buffer solution and 2.3 days at pH 9.

In a study of the operational application of imazapyr as Chopper® (and other herbicides) on electric utility rights-of-way in New York, Norris (1991) monitored the entry of herbicide to water on or near the right-of-way. Buffer strips of 30 to 100 feet were used depending on the method of application. No imazapyr was found in samples from two sites. At two other sites, two and five samples respectively were found to contain imazapyr at concentrations of less than 6 ppb. After the first month, seven and eight samples respectively contained detectable residues of imazapyr (maximum concentration detected at any time in this study was 0.006 ppm. These latter samples were collected in connection with periods of heavy rainfall, 4.2 inches in eight days, and 2.32 inches in 7 days. These results show it is possible for imazapyr to move to water courses from treated portions of the right-of-way. The concentrations found were all well below levels that are harmful to aquatic organisms, or other water users.

Residues in Animals

Imazapyr is rapidly excreted by animals. Tests on rats showed that rats excreted imazapyr in urine and feces with approximately 50% of the dose excreted in less than one day and 98% of the doses excreted within 6 days. No significant tissue residues were found.

Rats given technical imazapyr by stomach tube excreted approximately 87 percent of the dose in feces and urine within 24 hours, and 95.1% percent of the dose was excreted within 6 days. The half-life of technical imazapyr in the rat was less than 1 day. When administered intravenously more than 90% of the imazapyr was excreted in the urine and about 6% in feces and very small percentages of minor metabolites were also excreted. After 24 hours, very small concentrations of imazapyr were found in the liver and kidney. Small residual levels were found in muscle, fat and blood at both 1 and 8 days after treatment. USEPA (1985) considered that these residue amounts were not significant. Similar patterns have been observed in lactating goats and chickens.

Wildlife Hazard Analysis

Most research on the toxicology of imazapyr has been done by the registrant or its contractors to support registration[2] by USEPA (and/or by states and by agencies outside of the US) and results are not publicly available. The data have been reviewed by the registering agencies and their external consultants. The basic toxicology of imazapyr is reviewed in the human health risk analysis (EPRI, 2003).

Avian and Terrestrial Species

Imazapyr is slightly toxic to mammals based on acute oral LD_{50}'s ranging from greater than 2,000 mg/kg in mice to greater than 5,000 mg/kg in rats (USEPA, 1985; American Cyanamid Company, 1985a,b). Weeks, et al. (1988a) found no effect at 500 mg/kg in acute and chronic feeding studies. This was the highest rate tested.

Imazapyr is characterized by USEPA (1985) as practically nontoxic to avian species. Acute oral LD_{50}'s of technical imazapyr and the Arsenal® formulation are greater than 2,150 mg/kg (HDT) bobwhite quail and mallards (American Cyanamid Company, 1984; USEPA, 1985). Dietary LC_{50}'s for formulated and unformulated imazapyr are greater than 5,000 ppm (HDT) for mallards and bobwhites (American Cyanamid Company, 1984). No adverse effects were observed at any of these doses.

Imazapyr appears to be relatively nontoxic to insects. The LD_{50}'s for honey bees of technical imazapyr are greater than 100 ug/bee (HDT), and the Arsenal® formulation is greater than 25 ug/bee (HDT) (American Cyanamid Co., 1984). No effects were observed at either of these doses.

[2] The information required is specified by the regulatory agencies, as are the testing protocols. Good Laboratory Practices principles are specified and an auditing system is in place.

The wildlife risk analysis is in Table 2-23.

Table 2-23
Wildlife risk analysis for imazapyr

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	0.5	8	430	2150	bobwhite
Mammals	0.8	15	400	2000	rabbit
Amphibians	0.5	8	430	2150	bobwhite
Reptiles	0.6	10	430	2150	bobwhite

[1] The common laboratory test species used to represent the group of wildlife species.

The risk analysis shows there is a wide margin of safety for wildlife species.

Aquatic Species

Technical imazapyr, the isopropylamine salt of imazapyr, and the Arsenal® 2.0 AS formulation are practically nontoxic to rainbow trout, bluegill, and channel catfish. The water flea, the only aquatic invertebrate that has been tested, had an LC_{50} value greater than 350 mg/liter Arsenal®. 100 mg/liter to technical imazapyr and 750 mg/liter to the isopropylamine salt of imazapyr. The review by Norris (1991) reports a no effect level of 180 ppm for Daphina (American Cyanamid Company, 1985a,b). No studies have been reported with amphibians. Chronic or reproductive studies have not been reported in the literature.

The risks of adverse effects from exposure to imazapyr were estimated for representative aquatic species. In cases where no acute toxicity reference value was available for a representative species, a value was selected from the table of the most closely related species.

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by USEPA where the risks of adverse effects to fish or invertebrates are described in Table 2-24.

Table 2-24
Definition of "Q" value

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-25.

Table 2-25
Aquatic species risk analysis for imazapyr

		Off-Site Drift			Accidental Direct Spray	
	Critical LC_{50}	Typical	Maximum	Spill in Pond	Typical	Maximum
PPM ----------Q Value----------						
Imazapyr	100	<0.1	<0.1	<0.1	<0.1	<0.1

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to any of the representative aquatic species for typical and maximum exposures to imazapyr resulting from off-site drift, spill in a pond or the accidental direct application to surface water. All Q values are less than 0.1.

Other Aquatic Species

Phytotoxicity of imazapyr to an aquatic macrophyte, *Myriophyllum sibiricum* was studied under laboratory conditions to determine inhibitory concentration 25 and 50 [IC_{25} and IC_{50}, concentrations (mg a.i./L) that inhibit an endpoint parameter by 25 and 50%, respectively]. Roshon et al. (1999) used the maximum label rate of 2 kg/ha (expected environmental concentration of 1.33 mg a.i./L). *M. sibricum* was highly sensitive to imazapyr: IC_{25} and IC_{50} for shoot growth were 0.013 and 0.032, respectively; IC_{25} and IC_{50} for root numbers were 0.022 and 0.029, respectively; and IC_{25} and IC_{50} for root dry mass were 7.9^{-3} and 9.9^{-3}, respectively. All IC_{25} and IC_{50} values were well below the expected environmental concentration, indicating that imazapyr is phytotoxic to *M. sibricum*.

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

The volatility of imazapyr is virtually nil, hence spray droplets are the primary source of imazapyr in air. Once droplets settle out, there should be no measurable residues in air and therefore no environmental risk.

Soil and Groundwater

Imazapyr is rapidly dissipated from soil. The investigator expects a soil half-life of less than 30 days. Movement of imazapyr through the soil profile does occur but it is largely restricted to the upper 30 cm, indicating a very low probability of groundwater contamination. There is no indication that imazapyr adversely affects soil organisms at the concentrations likely to occur in soil; therefore, no impact on long-term soil productivity is expected and imazapyr in soil present little or no environmental risk.

PART 6. RISK ASSESSMENT FOR METSULFURON METHYL

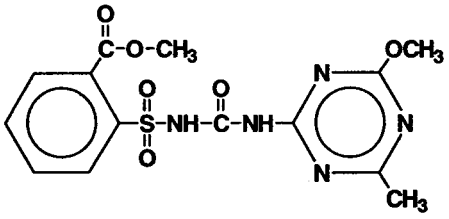
Introduction

Metsulfuron methyl is a broad-spectrum herbicide for selective and nonselective broadleaf weed and brush control. The most common formulations containing metsulfuron methyl are Escort® (labeled for control of annual and perennial weeds in non-crop areas) and Ally® (labeled for use in agriculture) and are produced by Du Pont. Application rates range from 0.9 to 1.8 oz. a.i./acre. The most common methods of ground application for Escort® are backpack (low volume foliar) and boom or hose and gun hydraulic spray (high volume foliar) operations. Occasionally, metsulfuron methyl is applied in combination with 2,4-D or 2,4-D and picloram.

The principal sources of information used for this report include USDA Forest Service (1989 and 1997), SERA (2000), and USEPA (1999b). These are not specifically quoted in the discussion. It is not the purpose of the present review and assessment to repeat the analyses of those reports but rather to provide a manageable discussion of environmental and wildlife risk associated with use of the herbicide. Recent works that may not have appeared in these reviews, and published work with pertinent details are referenced.

Physical-Chemical Properties and Environmental Behavior

Metsulfuron methyl is a sulfonylurea herbicide. Escort® contains 60% (w/w) metsulfuron methyl and 40% (w/w) inerts.



Metsulfuron Methyl

Important properties of metsulfuron methyl are in Table 2-26.

Table 2-26
Selected physical and chemical properties of metsulfuron methyl with selected additional properties for the commercial formulations Escort®

Property	Value
Vapor pressure	3.3×10-7 mm Hg
Water solubility	109 mg/L at 25C (pH 4.1) 9500 mg/L at 25C (pH 6.7)
K_{ow}	0.018
K_{oc}	42 (4 to 206) L/kg

From SERA (2000)

Behavior in Air

There are no published data on metsulfuron methyl in air. It has a relatively low vapor pressure, meaning the tendency to vaporize is limited.

Behavior in Plants

Metsulfuron methyl is readily absorbed by the foliage and roots of both sensitive and resistant herbaceous species and is translocated rapidly throughout the plant. The primary mode of action of metsulfuron methyl is inhibition of cell division in growing tips of roots and shoots of susceptible species, followed by numerous, less sensitive secondary mechanisms. Detoxication in resistant species is the basis of selectivity for metsulfuron methyl. Resistant plants metabolize the compound to nonherbicidal metabolites (Du Pont 1984).

Behavior in Soil and Groundwater

Soil

Metsulfuron methyl has a soil half-life of 1 to 2 months, depending on soil moisture, temperature, pH, and organic matter content, although there are instances where it has been reported to last longer, including carry-over phytotoxicity to very sensitive plants one year after application. The principal modes of metsulfuron methyl degradation in soils are acid hydrolysis and microbial degradation. Bastide et al. (1994) showed that roughly half of the decomposition of metsulfuron methyl is due to microbial activity and half due to hydrolysis, based on studies with steam sterilized and unsterilized soils in a laboratory setting.

In Malaysian soils, Ismail and Lee (1995) reported that half-life values for metsulfuron sandy loam and clay soils were 9.0 and 11.2 days, respectively at an incubation temperature of 35 ºC and a 20% field capacity; and decreased to 5.7 and 4.6 days, respectively at 35 ºC and a 80% field capacity. Ismail and Azlizan (2002) reported that metsulfuron methyl half-life values for three soil types (in the lab) at a temperature of 20 ºC (80% field capacity) were 4.79, 4.9, and 3.3 days. By increasing the temperature to 30 ºC, the half-life values decreased and to 2.78, 3.5, and 1.9 days, respectively. At 30 ºC, the half-life also decreased as soil moisture increased from 20% to 80% (Ismail and Azlizan, 2002).

Increasing soil temperature from 8 to 24°C in the loam and sand reduced the duration of metsulfuron methyl bioactivity from 267 to 136 days (49 percent) and from 235 to 187 days (20 percent), respectively. Increasing soil water level from 0.06 to 0.10 (kg/kg) increased metsulfuron methyl degradation in the sand at 24°C, but not at 16°C. In the loam, increasing soil water level from 0.15 to 0.25 kg/kg did not affect metsulfuron methyl degradation at either temperature Anderson (1985).

Smith (1986) determined the half-life of radioactive metsulfuron methyl in clay, clay loam and sandy loam soils at 85% of field capacity at 20 degrees C in the laboratory. He measured both the metsulfuron methyl and a primary metabolite - carboxymethylbenzenesulfonamide (CMBS) - at various periods. The results showed degradation followed first order kinetics, with half lives for metsulfuron methyl of 70, 102 and 178 days in clay loam, sandy loam and clay. The metabolite was found at less than 5% of the total except in the clay loam, where it was 32% of the radioactivity recovered (Table 2-27).

Table 2-27
Radioactivity recovered from soils treated with ^{14}C metsulfuron methyl following incubation at 20°C and 85 percent field capacity

	Percent of Applied ^{14}C Extracted as						
	Sandy Loam		**Clay Loam**			**Clay**	
Time	MM	CMBS	MM	CMBS	Time	MM	CMBS
1 h	100	<1	98	<1	1 h	100	<1
7 d	98	2	86	9	7 d	100	<1
14 d	95	<1	84	4	14 d	95	<1
28 d	82	1	70	18	28 d	90	2
56 d	66	3	56	24	49 d	83	1
84 d	58	2	45	32	70 d	76	2
					98 d	69	3

Notes: MM = Metsulfuron methyl; CMBS = 2-Carboxymethylbenzenesulfonamide. Adapted from Smith, 1986.

Metsulfuron methyl is not tightly adsorbed in soil, so is subject to leaching pressure as water moves through the soil profile. Studies generally show some leaching of the herbicide occurs, but the amount of herbicide is simultaneously decreasing due to a combination of microbial and chemical (hydrolysis) degradation. See early research and reviews by Anderson (1985), Walker and Welch (1989), Blair and Martin (1988), and Labat-Anderson (UD).

Vicari et al. (1994) studied the movement and persistence of metsulfuron methyl at four locations in Italy, after application at 8 grams per hectare, of active ingredient. They also sampled areas which had received three successive annual applications, reporting metsulfuron methyl was detected in soil only in the samples collected one month after application, and not thereafter. At locations where rainfall was about 45 mm in the first month, there was no metsulfuron methyl detected except in the surface soil sample (0 to 10 cm). At a site where 110 mm fell during this same period of time and the soil had a high proportion of coarse sand (27.8%) compared to the other sites (0.6 to 18.7 % coarse sand), metsulfuron methyl was detected in decreasing quantities at all depths sampled (to 50 cm).

In studies with eight different soils, Walker and Welch (1989) showed metsulfuron methyl persistence was negatively correlated with soil pH, and was generally lower in sub-soil than in the soil in the top 40 cm, although there was relatively little mobility to depths greater than this. Using data from field trials, they calculated the distribution of metsulfuron in soil as a function of time after application, showing only a slight tendency to leach in soil over a 148 day test period, with major reductions in the concentration in soil occurring during this time. However this was during a period of limited precipitation (220 mm). When the data are projected over 355 days with a total precipitation of 659 mm, the metsulfuron methyl was projected to move to a depth in excess of 120 cm, with the maximum concentration being at about 110 cm, although the total amount of the original dose remaining at that time was small. In testing surface soils for

residues using a very sensitive bioassay one year after field applications at various concentrations, initial applications rates greater than 8 g/ha resulted in a detectable (significant) reduction in the fresh weight of sugar beets (a plant species particularly sensitive to metsulfuron methyl). A field test of persistence and mobility in northern Greece by Kotoula-syka et al. (1993) produced similar results.

Metsulfuron methyl soil mobility was studied in detail by Bergström (1990) in lysimeter studies involving packed soil or undisturbed soil monoliths. Three soil types were represented, two sandy soils and one clay soil. Herbicide was applied at 4 and 8 g a.i. ha^{-1}, representing normal and double doses of the compounds for spring cereals. All lysimeters received supplementary watering in addition to natural rainfall. No metsulfuron methyl was detected in leachate from lysimeters receiving the normal dose of the herbicides, but at double the dose level 21 ng/L metsulfuron methyl was found in the leachate. Converted to fluxes over the 7-month period, considerably less than 1% of the applied metsulfuron methyl appeared in leachate.

In laboratory studies metsulfuron methyl will show a longer persistence due to the absence of some factors (such as leaching pressure, volatilization, photodecomposition) that may contribute to the dissipation of metsulfuron methyl in the field. In alkaline soils it will last longer, but in the more acid, organic matter rich soils of forest associated ROW it should not persist in significant amounts beyond 1 or 2 months.

Sarmah et al. (1998) reviewed the fate and behavior of metsulfuron methyl in Australian soils. They showed that the movement of the sulfonylurea herbicides is largely influenced by organic matter and soil pH, and that the herbicides may have substantial leaching potential in sandy Australian soils. The sorption of metsulfuron, is pH-dependent with a strong negative correlation with pH (at pH>8.0 sorption is low). In acidic soils, sorption is influenced by soil temperature, clay content, and, particularly, organic matter content.

Black et al. (1999) reported that biologically active residues of metsulfuron methyl decomposed quickly in southeastern Australian soils (sampled 80 cm soil profiles); 40-85% of the residues were lost in the first 5-7 months at 3 of 4 sites studied. The fourth site had soils with a high clay content, and residues occurred longer and at 12–24 months, about 0.3–1.0 ng/g (ppm) remained in the soil profile.

Wadd and Drennan (1989) studied the persistence and leaching of metsulfuron methyl in barley and fallow fields. It was not detected in the upper 20 cm of soil 120 days after application (Table 2-28). There was no evidence of substantial leaching, and degradation occurred in the warm, moist soil conditions.

Table 2-28
Metsulfuron methyl residues at three soil depths in a sandy loam soil with 2.9 % organic matter and pH 5.8 with and without a growing crop (table from Wadd and Drennan 1989). Metsulfuron was applied at a dose rate of 6 g a.i./ha.

Time (days)	System	Soil depth		
		0-5 cm	5-10 cm	10-20 cm
		estimated metsulfuron methyl concentration (ng/g)		
0	Cropped	3.55	-	-
	Fallow	3.55	-	-
15	Cropped	2.74	1.75	0.27
	Fallow	2.43	1.31	0.08
30	Cropped	2.63	0.16	0.20
	Fallow	2.92	0.36	0.16
45	Cropped	1.78	0.09	0.12
	Fallow	2.00	0.08	0.24
60	Cropped	0.09	0.01	0.04
	Fallow	0.08	0.02	0.04
120	Cropped	0.00	0.00	0.00
	Fallow	0.00	0.00	0.00

Groundwater

Due to the relatively short persistence and leaching potential as illustrated in field studies, there is little expectation that metsulfuron methyl will enter ground water except under unusual circumstances. Two studies have reported the fate of the herbicide in aquifers. Madsen et al. (2000) reported that metsulfuron had a low sorption to low organic carbon sediments (total organic carbon content below 1 g/kg) in 10 Danish aquifers. The mean sorption values (K_d) for metsulfuron was 0.06 L/kg and ranged from <0.01 to 0.19 L/kg. In general, the sorption of metsulfuron was insignificant and was only found at a pH below 5.2. Sorption was significant for only three sediments and was highest for a sediment with easily extractable aluminum and high iron oxides. Albrechtsen et al. (2001) reported that metsulfuron methyl did not degrade in shallow, sandy Danish aquifers under experimental trials.

Behavior in Surface Waters

Metsulfuron methyl is subject to degradation by acid hydrolysis and is much more stable in neutral to alkaline solutions. At pH 7 to 9 and at 25°C, metsulfuron methyl is stable for more than 1,000 hours; at pH 2 and 5 the half-life of metsulfuron methyl is 15 and 800 hours,

respectively (Du Pont, 1984). In the field, photodecomposition and volatilization of metsulfuron methyl is negligible (USEPA, 1988b).

Metsulfuron methyl is subject to photolysis in aqueous solutions. Thomas and Harrison (1990) found the photolysis followed first order reaction kinetics, with a half life of 16 days in solutions with no surfactant, and 2.9 or 1.5 days in solutions containing surfactant materials. Thompson, MacDonald and Staznik (1992) studied the persistence of metsulfuron methyl in a mixed-wood/boreal forest lake in Ontario, Canada. They found the dissipation to be concentration dependent, with slower dissipation occurring at concentrations of 1 ppm. At this concentration, which is unlikely to occur in operational applications of this herbicide, the half time for dissipation was 84 days. At 0.01 ppm, the half time dissipation was 29 days, as is expected from the DuPont (1984) report where pH 5 half-life in water was 33 days.

Metsulfuron methyl is subject to degradation in aquatic systems. Under anaerobic conditions, Swanson (1988) (in a summary of a longer report) reported the half life of metsulfuron methyl to be 36 days in pond sediment and water. 99% of the material had dissipated in 124 days. Methane and carbon dioxide were final products of decomposition. In sterilized sediment and water from the same system, the half life of the metsulfuron methyl was 139 days. In aerobic systems of sediment and water, Muttzal and Vonk (1991) reported a longer half life (19 to 42 weeks, depending on the sources of the sediment and water). In lab experiments, metsulfuron methyl photodegraded rapidly (50% degraded in 15 hrs) in aqueous solution (Samanta et al., 1999).

There are no published reports of studies that have looked for metsulfuron methyl in water in connection with its application in the field, but the investigator expects residue levels similar to those for 2,4-D with appropriate adjustments for rate of application.

Behavior in Animals

Metsulfuron methyl is soluble in water (9500 ppm) but is nearly insoluble in hexane, which means metsulfuron methyl is unlikely to accumulate in animals, and will clear the body quickly when exposure stops. Rats administered metsulfuron methyl excreted 78-96% of the herbicide in urine and 4.8-13.3% in feces. The biological half-lives were 9-16 hours for low-dose groups and 23-29 hours for high-dose groups.

Tissue data from rats also indicated little or no potential for retention or accumulation of metsulfuron methyl or its metabolites. Residues in organs and tissues were very low (<0.01 - 0.03 ppm) in low-dose groups and low, but proportionately greater, in high dose groups. Most (85-95%) of the radioactivity from labeled metsulfuron methyl was recovered unchanged. Metabolites were sequential hydrolysis products that eventually formed saccharin. In fish, Han and Anderson (1984) exposed bluegill sunfish to sulfometuron at 0.01 and 1.0 mg/L for 28 days and found no indication of bioconcentration. SERA (2000) assigned a bioconcentration factor of 1.

Wildlife Risk Analysis

Avian and Terrestrial Species

Based on an acute oral median lethal dose (LD_{50}) value of greater than 5,000 mg/kg in rats (USEPA, 1988b), metsulfuron methyl is classified as very slightly toxic (Maxwell 1982). Based on the chronic dietary feeding studies with metsulfuron methyl, USEPA (1988) established systemic NOEL values of 50 ppm (1.25 mg/kg/day), 500 ppm (25 mg/kg/day), and greater than 5,000 ppm (750 mg/kg/day), in dogs, rats, and mice. Metsulfuron methyl is of low toxicity to birds and mammals, based on acute oral LD_{50}'s of greater than 5,000 mg/kg in rats and greater than 2,510 mg/kg in mallard ducks (Du Pont, 1984). The 8-day dietary LC_{50} for mallards and bobwhite quail is greater than 5,620 mg/kg (Du Pont, 1984). Toxicity values for other terrestrial wildlife were not available.

The wildlife risk analysis is in Table 2-29.

Table 2-29
Wildlife risk analysis for metsulfuron methyl

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	0.5	8	502	2510	mallard
Mammals	0.8	15	1000	5000	rat
Amphibians	0.5	8	502	2510	mallard
Reptiles	0.6	10	502	2510	mallard

[1] The common laboratory test species used to represent the group of wildlife species.

The risk analysis shows there is a wide margin of safety for wildlife species.

Aquatic Species

Metsulfuron methyl has a low toxicity for fish and aquatic invertebrates. The 96-hour LC_{50} for trout and bluegill is greater than 150 mg/l (Du Pont, 1986). The 48-hour LC_{50} for *Daphnia magna* is also greater than 150 ppm (Du Pont, 1986). Toxicity values for amphibians and other aquatic invertebrate species were not found in the literature.

The risks of adverse effects from exposure to metsulfuron methyl that drifts offsite from foliar applications were estimated for representative aquatic species. In cases where no acute toxicity reference value was available for a representative species, a value was selected from the table of the most closely related species.

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations (I-5). The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by USEPA where the risks of adverse effects to fish or invertebrates are defined in Table 2-30.

Table 2-30
Definition of "Q" value

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-31.

Table 2-31
Aquatic species risk analysis to metsulfuron methyl

		Off-Site Drift			Accidental Direct Spray	
	Critical LC_{50}	Typical	Maximum	Spill in Pond	Typical	Maximum
	PPM	----------Q Value----------				
Metsulfuron methyl	150	<0.1	<0.1	<0.1	<0.1	<0.1

The results of the risk analysis indicate that there is no significant risk of acute adverse effects from metsulfuron methyl to any of the representative aquatic species for typical and maximum exposures resulting from off-site drift, spill in a pond or accidental direct spray of surface water. All Q values are less than 0.1.

Other Aquatic Species

Phytotoxicity of metsulfuron methyl to an aquatic macrophyte, *Myriophyllum sibiricum* was studied under laboratory conditions to determine the concentrations that cause 25% and 50% inhibition (IC_{25} and IC_{50}). Roshon et al. (1999) found that the IC_{25} and IC_{50} for shoot growth

were 1.5^{-4} and 3.9^{-4} ppm, respectively. Root numbers and root dry mass were of similar magnitude All of these values are well below the expected environmental concentration, indicating that metsulfuron methyl is phytotoxic to *M. sibricum*.

Wendt-Rasch et al. (2003) examined the short-term (2 weeks) effects of metsulfuron methyl using four levels of herbicide (0, 1, 5, 20 µ/L) on two macrophytes, *Elodea canadensis* and *M. spicatum*, in freshwater enclosures. Root growth decreased following exposure to the lowest concentration tested (1 µ/L). Additionally, pH in the aquatic community decreased, which was most likely an indication of decreased macrophyte primary production. Herbicide exposure did not alter the biomass or species of the phytoplankton community in the enclosures. Wendt-Rasch et al. (2003) concluded that the concentrations studied were within the range of expected environmental concentrations and that aquatic ecosystems, particularly those dominated by macrophytes, may be affected at concentrations present in water bodies adjacent to agricultural lands.

Peterson et al. (1994) also determined the phytotoxicity of metsulfuron methyl to aquatic organisms when applied at a the label rate of 0.0045 kg/ha (expected environmental concentration of 0.003 mg/L) and reported that metsulfuron methyl at the expected environmental concentration, had little or no toxicity to algal species and caused >60% inhibition of growth in duckweed (*Lemna minor*). For *L. minor*, Fairchild et al. (1997) reported a 96-hr EC_{50} value of 0.4 µg/L and an NOEL of <0.2. Nyström et al. (1999) reported that the species-dependent variation in algal sensitivity to the metsulfuron methyl was large for freshwater and marine species. Cyanobacteria and dinoflagellates were the most sensitive algae classes to metsulfuron methyl. EC_{50} for the two cyanobacteria species tested was 6 and <1 nmol. A dinoflagellate species was the most sensitive marine species (EC_{50} value < 1 nmol), while the most tolerant species had EC_{50} values > 1000 µmol.

Toxicity of metsulfuron methyl to single-celled green alga *Chlorella fusca* at pH 6.5 was reported to be: 14-hr EC_{50} value was 3.2 µmol/L for the inhibition of cell growth; and 24-hr EC_{50} 2.2 µmol/L for the inhibition of cell reproduction (Fahl et al., 1995). For *C. pyrenoidosa*, the 96-hr Log EC_{50} (inhibition rates were linearly regressed with logarithmic concentrations of compounds tested) were -0.21 mg/L for metsulfuron methyl, and 2.89 and 1.97 mg/L for two degradation compounds from metsulfuron methyl (Wei et al., 1998). Wei et al. (1998) concluded that the rapid degradation of metsulfuron would lead to a rapid decrease in acute toxicity to the green algae tested. For another green alga (*Selenastrum capricornutum*) Fairchild et al. (1997) reported a 96-hr EC_{50} value of 190 µg/L and an NOEL of <19.

Enclosure studies in boreal forest lakes in Canada showed, in general, no significant effects to phytoplankton (Thompson et al., 1993a) or zooplankton (Thompson et al., 1993b) communities. Metsulfuron methyl at concentrations as high as 1.0 mg/L (approximately 40× the worst case expected environmental concentration) induced only slight transient effects and only in the Cyanophyta (Thompson et al., 1993a).

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

There are no published data on metsulfuron methyl in air. It has a relatively low vapor pressure, meaning the tendency to vaporize is limited, which largely eliminates the environmental risk from residues in air.

Soil and Groundwater

Metsulfuron methyl has a soil half-life of 1 to 2 months, depending on soil moisture, temperature, pH, and organic matter content, although there are instances where it has been reported to last longer. It is not tightly adsorbed in soil, so is subject to leaching pressure as water moves through the soil profile. Studies generally show some leaching of the herbicide occurs, but the amount of herbicide is simultaneously decreasing due to a combination of microbial and chemical (hydrolysis) degradation. For instance in one study considerably less than 1% of the applied metsulfuron methyl appeared in leachate over a 7-mponth study period. In another field study there was no evidence of substantial leaching, and degradation occurred in the warm, moist soil conditions.

The investigator concludes that due to the relatively short persistence and leaching potential as illustrated in field studies, there is little expectation that metsulfuron methyl will enter ground water except under unusual circumstances. It would be prudent to avoid application in areas of very coarse soils, shallow water tables and abundant rainfall.

Effects on Soil Organisms

Soil biota can be affected short-term by metsulfuron applications. In a lab experiment, bacterial populations in a clay soil decreased with increasing herbicide concentration (0, 0.5, 2.0 ppm of a.i.) during the first 9 days of incubation, and after 27 days, the treated populations exceeded the control population (Ismail et al., 1996). In a sandy loam soil, bacterial populations were reduced during the first 3 days after application but increased to the levels of untreated controls after 9 days incubation (Ismail et al., 1996). Fungal populations in both soils increased with increasing herbicide concentration. Soil respiration was also stimulated in both soils but decreased during days 3 and 9 of the incubation period before increasing again later (Ismail et al., 1996). These findings indicate that effects on soil productivity are short lived, and there should be no long-term effects on soil productivity.

Metsulfuron methyl is persistent in soils and its bioactivity has been shown to injure susceptible agricultural crops. At high doses, metsulfuron may be phytotoxic to highly sensitive species for up to a year after application (Walker and Welch, 1989). Inhibition of corn roots by metsulfuron

bioactivity was affected by soil temperature in both loam and sandy soils, although the effect was less pronounced on the sandy soil (Anderson, 1985). These findings indicate the caution needed if portions of the ROW likely to be treated with metsulfuron methyl are also going to be used for gardens or growing of other sensitive plants.

PART 7. RISK ASSESSMENT FOR PICLORAM

Introduction

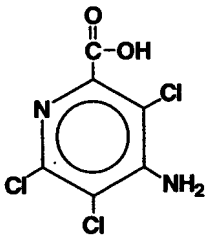
Picloram is a broad-spectrum herbicide used in the control of a number of broadleaf weeds and woody plants and is available in a variety of commercial products and formulations, including Tordon® K, Tordon® 22k and Tordon® 101. Application rates for range from 0.125 to 1 lb a.e./acre.

Most of the information on the toxicology of picloram is considered proprietary and is not publicly available. However, it has been reviewed by USEPA (1995b), California Department of Pesticide Regulation (1999), SERA (2003c). The fate of picloram in the environment is in the published literature and is included in reviews by USDA Forest Service (1989 and 1997). USDA Forest Service (1989) included picloram in a comprehensive risk assessment, as did Norris et al. (1991) with respect to aquatic organisms. These reviews are the primary resource for this risk assessment and are not specifically referenced. A useful and relatively non-technical review has been prepared by the Extension Service and is available at www.ace.orst.edu/info/extoxnet.

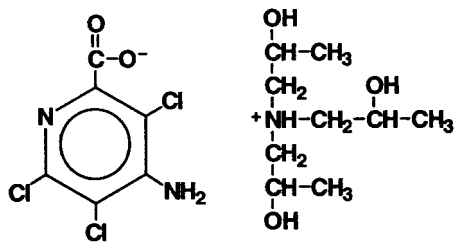
Physical-Chemical Properties and Environmental Behavior

Physical-Chemical Properties

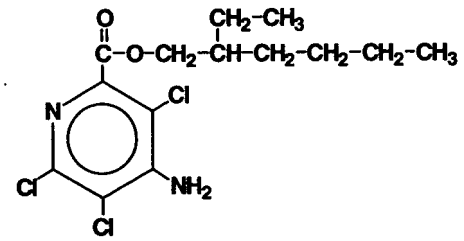
Both Tordon® K and Tordon® 22k contain the potassium salt of picloram (24.4% w/v, equivalent to a concentration of 2 lb a.e./gallon). The remaining 75.6% of the formulation consists of inerts, including a polyglycol. Tordon® 101 contains 0.54 lb a.e./gal picloram and 2 lb a.e./gal 2,4-D both as the triisopropanol amine salts. The USEPA has placed the polyglycol on List 3 of the inerts that may be used in the formulation of pesticides.



Picloram Acid



Picloram Triisopropanolamine Salt



Picloram Isooctyl Ester

Some of the important physical properties of picloram are as follows (WSSA, 1994):

Water Solubility: 430 mg/L @ 25C
Vapor Pressure: picloram acid 6.16 X 10^{-7} mm Hg @ 35C, no vapor pressure for potassium salt
K_{ow}: 1.4 @ pH 7.

Environmental Behavior of Picloram

The effectiveness of picloram as an herbicide has been attributed to its inherent potency, its ability to be translocated within the plant, and its relative resistance to degradation at ambient temperatures. Although studies indicate that picloram is not readily metabolized as a primary energy source by microorganisms, the compound is subject to biochemical and physical degradation in the environment in plants, soils, water, and microorganisms (Goring and Hamaker, 1971).

Behavior in Air

Picloram has no substantive vapor pressure so it is not volatile and residues in air would be restricted to spray droplets during application. While not explicitly studied, it is likely that picloram would be subject to photodecomposition in the air when exposed to light, just as it is in water.

Behavior in Plants

Picloram is readily absorbed by plant roots and less readily absorbed by foliage. Once absorbed, picloram is readily translocated throughout the plant with a tendency to accumulate in new growth. Although relatively little is known about the metabolism and fate of picloram in plants, most studies indicate that the compound is quite stable and remains largely intact within the plants (National Research Council of Canada, 1974; USDA Forest Service, 1973; and Witt and Baumgartner, 1979).

In residue studies reported by Frank et al. (1983), picloram residues were measured on ROW treated with 0.3–1.0 kg/ha to wild red raspberries. On one site, picloram was sprayed at a rate of 0.3 kg/ha (in a mixture with 2,4-D) to green fruit; and ripe berries collected 17 days after

treatment contained 0.14 mg/kg picloram residues, which declined to below 0.1 mg/kg 24 days after treatment and to 0.01 mg/kg 42 days after treatment. On a second 1.0 kg/ha picloram (mixed with 2,4-D) was applied to green berries; 21 days after treatment when the first fruits were ripe, picloram residues were <0.01mg/kg. On a third site picloram was applied at a rate of 0.9 kg/ha (with 2,4-D) to ripe berries. Initial residues were 5.7 mg/kg, which declined to 0.78 mg/kg 21 days after treatment, the end of the picking season.

Additionally, residues of picloram applied at 0.9 kg/ha (with 2,4-D) to ripe wild blueberries on a ROW site declined from initial residues of 3.20 to 0.45 mg/kg 21 days after treatment (Frank et al., 1983).

Behavior in Soil and Groundwater

The fate of picloram in soil is determined by several factors including volatilization, photodecomposition, adsorption and leaching, runoff, and chemical and microbial degradation. Volatilization is not considered to be a major determinant of environmental fate because of the low vapor pressure of picloram. The moisture, temperature, and organic content of the soil influence the rate of dissipation and movement of picloram in soil (National Research Council of Canada, 1974). Picloram persistence in soils is related to both treatment rate, climate and several soil properties. It is considered to be moderate to high, since it may exist at phytotoxic levels for a year or more following normal application (Mitchell, 1969 and National Research Council of Canada, 1974). The half-life of the compound has been reported to range from more than four years in arid regions to approximately one month under highly favorable conditions.

Since picloram degradation in soil does occur by microbial routes (Hance, 1967), degradation increases under conditions that favor microbial growth. Consequently, persistence is generally shorter in soil with high organic content (Gratkowski, 1980 and Witt and Baumgartner, 1979), high temperatures (Caro et al., 1974), and high soil moisture (Hunter and Stobbe, 1972). Studies also indicate that low pH (Youngson and Meikle, 1967), the presence of light-textured sandy soils, and the presence of plant roots also decrease the persistence of picloram (Herr et al., 1966; Hunter and Stobbe, 1972; Meikle, et al. 1966 and Merkle et al., 1967).

Both field and laboratory studies show the rate of dissipation and leaching of picloram in soil changes over time. The longer the herbicide is in the soil, the slower the rate of decomposition, and the lower the extent of leaching. McCall and Agin (1985) evaluated the desorption kinetics of picloram as a function of time in the soil, and found that the amount of picloram adsorbed to “internal sites” in the soil particles increased with time. Residues in these internal sites appear to be less available for decomposition or for leaching. This study provides an important theoretical basis for understanding the behavior of picloram (and probably other herbicides) in the soil.

Picloram is degraded by natural sunlight and ultraviolet light, although the extent of photodecomposition under field conditions has not been quantified. Under laboratory conditions, however, it has been found that 15 percent of picloram applied to soil surfaces was degraded by Texas sunlight after one week (Merkle et al., 1967).

Fryer et al. (1979) found picloram residues in a sandy loam soil 1 year after application at 0.04 and 0.22 LB /acre). At application rates of 1.5 LB /acre, 0.5 percent of the total amount of picloram applied remained in the soil as residue after 222 weeks. Laboratory soil tests of picloram degradation were conducted on two Philippine clay soils (Maahas and Luisiana) obtained from paddy fields (Yoshida and Castro, 1975). Half of the soil samples were maintained at submerged conditions, while the other half were maintained at 80% of field moisture capacity (upland conditions). Picloram was applied at a concentration of 1 ppm. Picloram residues were determined 0, 3, and 6 months after incubation. The amount of picloram in the Maahas soil after 3 months was less than half of the initial amount, but degradation did not occur in the Luisiana soil until 3 months after incubation began. After 6 months, less than 40% of picloram remained in both the submerged and upland conditions in the Maahas soils, 74% remained in the upland Luisiana condition samples, and 45% remained in the submerged samples.

Soil persistence and lateral movement of picloram (in a mixture with 2,4-D) was examined following brush control treatments applied at 1.2 kg/ha to power line ROWs (Meru et al., 1990). Soil samples were collected at intervals from 0.14 to 48 weeks after treatment. Only 1 of 85 downslope soil samples contained detectable residues of picloram No residues were found in soil samples 15 or 48 weeks after treatment.

Picloram is generally considered to be a mobile herbicide since its adsorption to soil particles is low. Factors that affect adsorption include soil type, pH, rainfall, formulation type, and application rate. The factors that generally result in relatively low adsorption (and, consequently, relatively high mobility) are: low organic matter content of the soil; low content of hydrated oxides of aluminum and iron in the soil; neutral or high pH (basic); and highly permeable, sandy, light-textured soils (Biggar and Cheung, 1973; Farmer and Aochi, 1974; Ghassemi et al., 1981; Grover, 1971; Hamaker et al., 1963; McCall et al., 1972; National Research Council of Canada, 1974; and Norris, 1970). The low pH silt loams with higher organic matter content, as tends to occur on forest related ROW are factors that reduce the mobility of picloram.

Preliminary studies with various soil types indicate that picloram is usually confined to the upper 1 foot (30 cm) when application rates are low (less than 1 lb/acre [l.12 kg/ha]), but that picloram can readily move to depths greater than 3 feet (approximately 1 meter), in sandy, glaciated soils even in relatively dry areas, when the application rate is high (3 to 9 lb/acre [3 to 10 kg/ha]) (personal communication cited in National Research Council of Canada, 1974).

Deubert and Corte-Real (1986) evaluated soil residues of picloram and triclopyr (as Garlon® 3A) after selective foliar application on a utility right of way in southern New Hampshire where soils were described as “very porous”. They concluded that for quantities of material applied in selective foliar application on utility rights-of-way, initial soil residues are small due to 30-90% interception (average about 50%) by non-target vegetation. Soil residues increase during the first 1-4 weeks after application due to washoff from vegetation. However, the rate of breakdown is faster than the rate of accumulation and soil residues of both chemicals decrease after 2-4 weeks. Leaching of picloram, the more mobile of the two chemicals studied, below 20-30 in. is improbable.

In an Appalachian forest in North Carolina, Neary et al. (1985) evaluated the movement and the persistence of picloram in the forest soil on the site. They found picloram residues were highest (11.58 mg/kg) in the upper 0.07 m. Picloram had a half-life of about 4 weeks, and declined to near detection limits 28 weeks after application. The soil solution contained the highest picloram levels (350 mg m-3). at a depth of 0.6 m in the soil. Picloram residues (<25 mg m-3) were detected in the soil solution at a depth of 1.2 m in the soil. Residues persisted for 60 weeks. Intensive sampling of two springs detected trace levels for a period of 18 days.

Neary et al. (1988) reported a soil half-life for picloram of 138 days in a sandy soil with low level of organic matter in Florida. Newton et al. (1990) reported the movement and persistence of picloram and triclopyr in a SW Oregon forest. This area has a hot dry summer, but during the periods of the study when there was precipitation, Newton et al. (1990) reported the half-life of the two herbicides varied between 11 and 25 days.

On a right-of-way near Rome, New York, Klippel et al. (1985 as cited in USDA Forest Service, 1997)) followed the leaching and persistence of picloram and triclopyr. They found that entry into wells or groundwater through leaching is highly unlikely. Downward leaching of herbicides on the sprayed sites through the soil to a depth of 10-15 inches was rare, occurring only at three locations. The leaching was believed to be caused by three types of circumstances:

1. Rainfall immediately after application.
2. A large amount of rainfall within a day after application.
3. The basal application of a high concentration of herbicides to a single spot on the site.

Herbicide concentrations in seepage from the top 6 inches of soil followed similar trends in mobility and persistence as soil samples. Picloram did not persist past ten weeks.

Based on the nature of the soils and environment on most forest related ROW, the most relevant field studies (Duebert and Corte-Real, 1986 and Neary et al., 1988) suggest a soil half-life of about 90 days, and leach to depths less than 30 cm, before residue levels have dropped below detectable levels (>0.01 ppm) in soil. These values suggest picloram will be less mobile and less persistent than reported in some areas. The investigator believes the texture, pH, moisture and organic matter content of the forest soils found on most ROW are consistent with this expectation.

Behavior in Surface Waters

Because of its mobility, picloram may be transported by surface runoff to nontarget areas including ponds and rivers. However, only a small amount of the total picloram applied is actually removed by runoff. It is generally accepted that there is a potential for high concentrations of picloram in runoff if heavy rainfall occurs soon after application (Gwinn, 1975).

Maximum concentrations of 400-800 ppb have been detected in surface runoff in instances where heavy rainfall occurred immediately after spraying a 1:1 mixture of triethylamine salts at 1 lb/acre (1.12 kg/ha) on grassland watersheds in Texas (Bovey et al., 1967). However, on the

Bovey et al. study site the conditions enhance the likelihood of such runoff compared to that on an electric utility ROW where there is a significant litter and forest floor layer.

Runoff accounts for less than 3 percent of the total quantity of picloram applied to soil and the concentration of picloram in runoff generally decreases with time as well as with the time lapse between application and the first rainfall (Trichell et al., 1968 in National Research Council of Canada, 1974). Other factors which decrease the concentration of picloram in runoff include flatter terrain, lower amounts of rainfall, the use of slow release granular formulations rather than liquids, higher infiltration capacity of the soil, and the greater distances over which the runoff flows. The lateral movement of picloram (in a mixture with 2,4-D) was examined following brush control treatments applied at 1.2 kg/ha to power line ROWs (Meru et al., 1990). Runoff water was collected at time increments from 0.14 to 48 weeks after treatment. Only 1 of 56 water runoff samples (sample was collected 10 m away from treatment site) contained detectable residues. No residues were found in water samples at either 15 or 48 weeks after treatment.

Only sporadic, low-level picloram residues were detected in streamflow from nested 10-ha and 28-ha watersheds during a 70-week period. Use of the herbicide picloram did not affect the quality of streamflow from Watershed 19 for domestic or agricultural purposes.

In studies of picloram injection into a stream system, Johnsen and Warskow (1980) found that the highest recovery of 38 percent of injected picloram was located 0.25 miles (0.4 km) from the point of injection. At a point 1 mile (1.6 km) downstream of the injection site, less than 5 percent of the injection concentration was found in water samples.

Aerial application of a mixture of picloram at 2.5 lb active equivalent (a.e.) per acre (2.8 kg/ha) and 2,4-D at 5 lb a.e. per acre (5.6 kg/ha) onto 279 acres (113 ha) of a pinyon-juniper watershed resulted in detectable levels of picloram in runoff for 30.5 months (Johnsen and Warskow, 1980). The highest concentration of picloram detected was 320 ppb in the initial runoff event following treatment. Of the total picloram applied, 1.1 percent ultimately left the area in runoff water.

Neary et al. (1979) monitored streamflow from a 10-acre (4-hectare) watershed for picloram residues following application of Tordon 10K® pellets at a rate of 4.5 lb a.i./acre (5 kg/ha) for site preparation. Picloram was applied in mid-May and two pulses of residues were detected in watershed outflow; a 3 ppb pulse in early June following precipitation and a maximum 8 ppb pulse in early July (no precipitation). No further residues were detected (reported through October) at detection limits of 1 ppb although there were relatively heavy rains in early August.

Picloram behavior in runoff water was evaluated in two studies. Mayeux et al. (1984) followed the concentration of picloram in runoff from grassy watershed, where no apparent buffer was provided. At another site, picloram was intentionally injected into a stream to achieve a concentration 1000 grams of picloram per cubic meter of water.

Rapid dilution of picloram was observed after direct introduction of the herbicide near the upper end of a larger stream during rising stage of flow. Maximum concentrations at 90, 1170, and 5,400m downstream from the point of introduction of 1270 g of picloram were 13,720, 470, and

5 mg/m3, respectively. The calculated quantity of picloram that was detected at the 5400-m sampling point in streamflow was only 0.13% of that introduced. Mixing of water during streamflow and additions of uncontaminated water along the channel apparently reduced the picloram concentration below that detectable by gas chromatography.

In Texas, Bovey and Richardson (1991) followed the fate of picloram after application to an area overlying a shallow perched water table. Approximately 90 days after application, about 92% of the picloram had disappeared. Most of the remaining herbicide was in the upper 30 cm of the soil. The area was treated in two successive years. No herbicide was detected in subsurface flows after the first year, but 1 to 4 ppb picloram were found in seep flow 11 days after the second treatment.

In a study of the operational application of picloram as Access® herbicide (and other herbicides) on electric utility rights-of-way in New York, Norris (1991) monitored the entry of herbicide to water on or near the right-of-way. Buffer strips of 30 to 50 feet were used depending on the method of application. He found picloram at a concentration of 0.001 ppm in one sample, and a few other samples with trace amounts (less than 0.001 ppm). The concentrations found were all well below levels that are harmful to aquatic organisms, or other water users, and showed that picloram persistence and mobility on ROW are sufficiently limited such that the likelihood of movement to streams is extremely limited.

In a study of the movement and persistence of herbicides on electric utility rights-of-way near Rome, New York, Klippel et al. (1985, as noted in USDA Forst Service, 1997) sampled soil and water at various periods after application of picloram, triclopyr and 2,4-D by basal, foliar and boom spray techniques. They reported that overland flow of herbicides in runoff did not occur under normal conditions, but two soil samples collected off right-of-way did contain detectable levels of picloram. In both of these instances the herbicide application was immediately followed by rainfall. The linear extent of overland flow was minimal and, when it occurred, the herbicide degraded rapidly. Furthermore, after the initial application, there was no indication in the data that overland migration of herbicide off the site was occurring. Rather, the trend was towards degradation of herbicides to undetectable levels.

The data indicate that entry into streams from overland flow is highly unlikely when appropriate non-treatment buffer zones are established adjacent to water resources.

Picloram is subject to photodecomposition in water. Using results from an elegant experiment to determine the quantum yield of light-mediated picloram decomposition in water, Skurlatov, Zepp and Baughman (1983) calculated picloram would have a half-life of 2.2 days in water at 40 degrees north latitude in late summer. Woodburn et al. (1989) measured the photolysis of picloram in natural water from a forest. They found a half-life of 2.6 days at 25 degrees C., at 40 degrees north latitude in mid summer. From this the investigator concludes that the effects of dilution and photodecomposition will prevent any runoff of picloram from having an important effect on downstream water quality.

It is improbable that aquatic microorganisms will degrade picloram in surface waters to an appreciable extent since microbial degradation rate in soils are fairly low (Youngson et al.,

1967). Photo degradation of picloram, which has been demonstrated by some researchers, may be an important mechanism in the dissipation of the compound (National Research Council of Canada, 1974; Gear et al., 1982; and Johnsen and Warskow, 1980). The major photo degradation pathway in the presence of sunlight or ultraviolet light is dechlorination (Gear et al., 1982). Johnsen and Warskow (1980) found that sunlight decomposed 57 percent of picloram in containers after less than 9 hours of exposure, although in natural soil environments picloram would not be directly exposed to sunlight and thus would not be photo degraded so rapidly. However, due to the more open nature of rights-of-way, the potential for photo decomposition is believed to be greater than would occur in typical forest settings.

Levels in farm ponds adjacent to plots treated with picloram at 1 lb/acre (1.1 kg/ha) have reached 1 ppm (Haas et al., 1971 in National Research Council of Canada, 1974). However, these levels decreased to less than 10 ppb within 100 days primarily due to dilution and photodegradation of picloram. The results of the study by Norris (1991) and Klippel, et al. (1985 as noted in USDA Forest Service, 1997), both on ROW in the northeastern US, establish the very limited entry of picloram to water from operations on such sites.

Residues in Animals

In addition, picloram does not appear to accumulate to any significant extent in animal tissues (USDA Forest Service, 1984). It is rapidly absorbed from the digestive tract. Of a 1400 mg/kg dose, 94% appeared as parent compound in urine of rats within 48 hours. Only traces remain in tissues for a few days after administration (USEPA, 1988a).

Table 2-32 indicates bioconcentration factors (BCF) observed for several organisms exposed to picloram (National Research Council of Canada, 1974). Several studies are summarized in National Research Council of Canada (1974). They show that 90 percent of the picloram fed in the diet to dogs was excreted within 48 hours in the urine, with small amounts appearing in the feces. In rats, picloram appeared unchanged in the urine reaching a peak in two hours. Tissue levels reached a maximum at 2-3 hours, falling rapidly to undetectable levels 12 hours. In cows, 97 percent was found unchanged in the urine. No measurable residues were found in milk samples from dairy cows fed 10-100 ppm picloram in the feed. Milk samples from cows fed 150-1,000 ppm picloram in the feed contained low levels (0.05-0.29 ppm) of residue, which declined rapidly and were undetectable 58 hours after withdrawal from the feed.

Table 2-32
BioConcentration factors (BCF) observed for picloram in aquatic and terrestrial species

Organism	Medium	Compound	Tissue	BCF Based on Acid Equivalent of Picloram[a]	Time Scale
Daphnia	Water	K-salt[b]	Whole body	1.0	7 weeks
Mosquito Fish	Water	Acid	Whole body	0.02	18 days
Dairy Cattle	Diet	K-salt	Milk	0.0003	2 weeks
Steers[c]	Diet	Acid	Blood muscle and fat kidney	0.001 0.0005 0.01	2 weeks
Sheep	Diet	Acid	Blood	0.001	1 week

[a]Concentration in tissue (acid equivalent) divided by concentration in water or diet (acid equivalent). A factor of one or less means no accumulation greater than that found in the medium of exposure, i.e., diet or water, over the time scale of the exposure. These numbers are not considered absolutes but only indicators since time-dose-dependent studies per se have not generally been carried out.
[b]k-salt = potassium salt.
[c]Upon continuous exposure, residues reached a plateau within three days. The concentration factor, ppm-blood divided by ppm-diet, was independent of time for the remainder of the study and the concentrations in the blood are nearly directly proportional to the concentrations in the feed.
Source: National Research Council of Canada 1974.

Wildlife Risk Analysis

Avian and Terrestrial Species

Picloram is slightly toxic to mammals, based on acute oral LD_{50}'s ranging from greater than 540 mg/kg in calves to 8,200 mg/kg in rats (Lynn, 1965; Jackson, 1965). The Tordon® 101 formulation caused no ill effects in sheep at single doses of 1,900 mg/kg, but it caused death at levels of 2,200 mg/kg and above (Lynn, 1965). Temporary weight loss was the only adverse effect seen in calves given Tordon® 101 in single doses of 1,900 to 3,163 mg/kg (Lynn, 1965). No toxic signs or adverse effects on growth were observed in sheep given 18 mg/kg/day of technical picloram in the diet for 33 days (Jackson, 1965). Stimulated growth and improved feed efficiency were observed in swine given 22 mg/kg of feed for an unspecified time (McCollister and Leng, 1969). Metabolic and residue studies in mammalian species indicate that picloram is rapidly eliminated unchanged in the urine following ingestion (USDA Forest Service, 1984). No metabolites have been detected (USDA Forest Service, 1984). Picloram is slightly toxic to birds based on LD_{50}'s that range from greater than 2,000 mg/kg in mallards and pheasants to approximately 6,000 mg/kg in chickens (Lynn, 1965; Hudson et al., 1984). Regurgitation occurred shortly after mallards were treated, and pheasants exhibited tremors and mild decline of muscle coordination after treatment (Hudson et al., 1984). Subacute dietary LC_{50}'s for bobwhite

and Japanese quail, ring-necked pheasants, and mallard ducks were all greater than 5,000 ppm. The 8-day dietary LC_{50} of the Tordon® 101 formulation is greater than 10,000 ppm for bobwhite quail and mallard ducks (USEPA, 1984b).

Japanese quail given 100 ppm in a 2-week dietary study showed no effects on feathering, reproduction, mortality, and weight (Kenaga, 1969). In a similar test at 1,000 ppm, egg fertility and hatchability were reduced the first week but not the second (Kenaga, 1969). A three-generation study with Japanese quail showed no effects on food consumption, reproduction, survival, and body weight when given 100, 500, or 1,000 ppm in the diet (Kenaga, 1969). In a 1-year study in which Japanese quail were given 100 ppm to 10,000 ppm in their diet, no effects on reproduction, feeding, or body weights were observed. Mortality rates of treated quail were lower than those of controls (Kenega, 1969).

The LC_{50} of mallard eggs immersed in an aqueous emulsion of picloram was equivalent to a field application rate of 112 kg/ha (100 lb/acre), which is more than 10 times the recommended field application level (Hoffman and Albers, 1984). Spray treatment of fertile chicken eggs or ring-necked pheasant eggs with a dose equivalent to 2.8 kg/ha (2.5 lb/acre) of Tordon® 101 did not affect embryonic development or subsequent growth of hatched chicks (USEPA, 1984b).

Picloram is relatively nontoxic to insects based on an acute contact LD_{50} of greater than 14 ug/bee in honey bees (Kenaga, 1979). Honey bees given 1,000 ppm picloram in a 60-percent sucrose syrup showed no toxic effects after 14 days and no increase in mortality compared to the control group after 60 days (USDA Forest Service, 1984).

The wildlife risk analysis is in Table 2-33.

Table 2-33
Wildlife risk analysis for picloram

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	1	24	400	2000	pheasant
Mammals	3	49	144	720	sheep
Amphibians	0.03	1.3	400	2000	pheasant
Reptiles	0.04	1.3	400	2000	pheasant

[1] The common laboratory test species used to represent the group of wildlife species.

The risk analysis shows there is a wide margin of safety for wildlife species.

Aquatic Species

Tordon® 101 (a mixture of picloram and 2,4-D) is slightly toxic, and picloram is generally moderately to slightly toxic to aquatic organisms. All reported LC_{50}'s for Tordon® 101 are greater than 10 ppm. Tordon® K is a potassium salt of picloram. Tordon® 22 K is similar and provides the relevant aquatic toxicology data. The most sensitive species of fish is bluegill sunfish, with a 96-hour LC_{50} of 5.4 ppm (USDA Forest Service, 1984).

Woodward (1979) reported increased fry mortality in cutthroat trout at concentrations of picloram (technical grade) greater than 1.3 ppm and reduced fry growth above 0.61 ppm (flow-through tests). No adverse effects to cutthroat fry occurred below 0.29 ppm. The reported concentrations are initial peak concentrations, which are intended to simulate concentration resulting from runoff from a rainstorm. Mean concentrations for the exposure period were not reported. Similar findings have been reported by Scott et al. (1977, as cited in Mullison, 1985). Woodward (1976) has also reported chronic studies on lake trout, where 0.035 ppm of picloram adversely affected the rate of yolk sac absorption and growth of fry.

Mayes et al. (1987) conducted chronic toxicity studies with embryo-larval rainbow trout exposed to technical picloram. They reported an MATC of between 0.55 ppm and 0.88 ppm and estimated as 0.70 ppm based on the geometric mean. Larval survival was significantly reduced at 2.02 ppm, and growth was significantly reduced at 0.88 ppm.

Studies with picloram (Tordon® 50-D) have reported 96-hour LC_{50}'s for 1-week-old tadpoles of 95 ppm for Adelotus brevis and 105 ppm for Limnodynastes peroni (Johnson, 1976). Access® (isooctyl ester of picloram and 1:2 triclopyr formulation) has a 96-hour LC_{50} of 4.0 ppm to fish, and 1.4 ppm to crustaceans.

When tested across a range of aquatic species, Mayes and Dill (1984) found rainbow trout was most sensitive to three different forms of picloram. The toxicity of picloram was tested with various life stages of rainbow trout by Mayes et al. (1987). They found no statistically significant difference between controls and treated fish for % embryo hatch or the % of larvae that are normal at hatching at any concentration tested (up to 2). They found a 19% decrease in larval survival at 2 ppm, but no difference at 1.34 ppm, and a reduction in both body weight and body length at concentrations of 0.88 ppm and higher, with both of these responses showing a dose response at the higher concentrations. The no effect level for the most sensitive indicator was 0.55 ppm.

Mayes and Oliver (1985) summarized the literature in doing an aquatic hazard assessment for picloram. They concluded that because of its relatively low toxicity to aquatic species and its relatively low potential for entry to aquatic systems that it is not expected to present a hazard to the aquatic environment.

The risks of adverse effects from exposure to picloram from foliar applications were estimated for representative aquatic species. In cases where no acute toxicity reference value was available for a representative species, a value was selected from the table of the most closely related species.

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by USEPA where the risks of adverse effects to fish or invertebrates are defined in Table 2-34.

Table 2-34
Definition of "Q" values

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-35.

Table 2-35
Aquatic species risk analysis for picloram

		Off-Site Drift			Accidental Direct Spray	
	Critical LC_{50}	Typical	Maximum	Spill in Pond	Typical	Maximum
PPM	----------Q Value----------					
Picloram	5.4	<0.1	<0.1	<0.1	<0.1	<0.1

All Q values are less than 0.1. The results of the risk analysis indicate that there is no significant risk of acute adverse effects from picloram to aquatic species for typical and maximum exposures resulting from off-site drift, a spill in a pond or accidental direct spray to surface water.

Other Aquatic Species

Aquatic insects and crustaceans have 24- to 96-hour LC_{50}'s of greater than 25 ppm for technical picloram. A 48-hour LC_{50} of 50.7 ppm has been reported for Daphnia magna exposed to technical picloram (Mayes and Dill, 1984). Daphnia sp. showed no effect during a 24-hour exposure to 380 ppm of Tordon® 101 (USDA Forest Service, 1984). However (Johnson, 1976) using Access® (isooctyl ester of picloram and 1:2 triclopyr formulation) reported a 96-hour LC_{50} of 1.4 ppm to crustaceans..

No adverse effects on growth were reported for algae or Daphnia sp. exposed to 1 ppm picloram for 10 weeks (Lynn, 1965, as cited in Ghassemi et al., 1981). Chronic studies with Daphnia

magna by Gersich et al. (1985) indicated an MATC of between 11.8 and 18.1 ppm with a geometric mean of 14.6 ppm. The MATC endpoint was based on mean total young/adult.

Peterson et al. (1994) found that picloram applied at 2.64 k/ha (expected environmental concentration 1.760 mg/L) had a low toxicity to 10 algal species tested and at the expected environmental concentration rate caused <50% inhibition of growth in a vascular water plant, duckweed.

From this information the investigator concludes that the use of picloram on electric utility ROW offers no substantive risk to other aquatic species

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

Picloram has no substantive vapor pressure so it is not volatile and residues in air would be restricted to spray droplets during application, which largely eliminates the environmental risk from residues in air.

Soil and Groundwater

The primary environmental risk from picloram is its longer persistence and relatively high mobility in soil. Laboratory studies reinforce these points, but interestingly field studies show much less leaching than might be expected based on the lab studies. However, there are clearly circumstances where persistence and leaching can combine to result in potential ground water contamination by this herbicide. For this reason, the investigator recommends caution in the use of picloram. Fortunately, the long but narrow configuration of rights-of-way on the landscape should reduce this risk compared to larger geographic scale applications as might be made for forestry or rangeland management purposes.

PART 8. RISK ASSESSMENT FOR SULFOMETURON METHYL

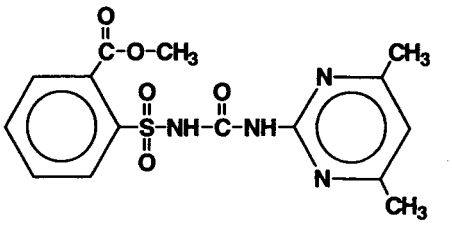
Introduction

Sulfometuron methyl is a non-selective, sulfonylurea herbicide used to control broadleaf weeds and grasses. The primary commercial formulation is Oust® manufactured by Dupont. Sulfometuron methyl can be applied in combination with other herbicides such as diuron, glyphosate, or hexazinone. The most common methods of application of Oust® involve backpack (low volume foliar) and boom and hydraulic hose and gun spray (high volume foliar) operations. Application rates range from 0.5 to 6 oz. a.i./acre.

Very little of the research on the toxicology of sulfometuron methyl is published in the open scientific literature and is considered to be proprietary and confidential. This information has been provided for technical review. The primary sources used for this assessment are USDA Forest Service (1989 and 1997), SERA (1998a), and California Department of Pesticide Regulation (CDPR, 2002b) and are not specifically referenced. Where appropriate, recent research not included in these reviews are specifically cited. A brief summary of sulfometuron methyl behavior and toxicology is also available from the Extension Service (Extoxnet, 1996). It is not the purpose of the present review and assessment to repeat the analyses of those reports, but to provide a manageable discussion of environmental and wildlife risk associated with ROW vegetation management.

Physical-Chemical Properties and Environmental Behavior

Sulfometuron methyl is the common name for 2-[[[[(4,6-dimethyl-2-pyrimidinyl)- amino] carbonyl] amino] sulfonyl] benzoic acid methyl ester. Oust®, the principal commercial formulation of sulfometuron methyl, is a non-selective sulfonylurea herbicide formulated as a water dispersible granule containing 75% sulfometuron methyl and 25% inert ingredients.



Sulfometuron Methyl

The important physical properties of sulfometuron methyl are in Table 2-36.

Table 2-36
Selected physical and chemical properties of sulfometuron methyl

Property	Value
Water solubility	10 mg/L @ 25C, pH 5; 300 mg/L @ 25C, pH 7
Vapor pressure	5.5×10^{-16} mm Hg
K_{ow}	11 @ pH 5: 0.346 @ pH 7; 0.0136 @ pH 9
K_d (Soil adsorption)	Highly variable: 0.04 to –3

From SERA 1998a

Environmental Behavior

Behavior in Air

Sulfometuron has a vapor pressure that is extremely low. There should be no significant vaporization from droplets or environmental surfaces, meaning that residues in air will be restricted to the distribution of droplets during the application.

Behavior in Plants

Sulfometuron methyl is readily absorbed into foliage and (from treated soil) roots. SERA (1998a) reports plant uptake as 10% in 72 hours. It translocates in both xylem and phloem, accumulating in the meristematic plant tissues. Growth of treated plants is inhibited shortly (a few hours) after application although injury symptoms typically do not become apparent for a few weeks. Symptoms typically include chlorosis of meristematic areas, foliar chlorosis and necrosis. It acts by inhibiting a key enzyme in the biosynthesis of branch-chained amino acids, and plant death results from the events related to this inhibition, although the precise mechanisms is not known.

Little is known of its persistence in plant tissue, except for a study done by Michael (2003) in connection with reforestation in Mississippi. Sulfometuron residue levels in loblolly pine foliage and foliage from competing understory plants were measured for 27 days after foliar application of Oust® at 0.42 kg/ha (6 oz. a.i./acre). The data show a rapid dissipation of sulfometuron in foliage, with a half-life of less than 7 days in each case (Table 2-37).

Table 2-37
Sulfometuron methyl residues (mg/kg) in forest vegetation in Mississippi after application of 0.42 kg/ha sulfometuron methyl as Oust®

	Days After Treatment					
	0	1	3	7	14	27
Vegetation	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
Loblolly Pine	6.14	4.07	1.49	0.42	0.19	0.19
Grass	6.89	5.83	5.38	1.46	1.63	ND
Pokeweed	33.60	24.16	14.06	10.55	4.55	0.82
Blackberry	40.77	24.86	11.33	2.23	0.39	ND

From Table 6, Michael (2003), ND is not detectable

Michael and Neary (1988) reported that residues of sulfometuron methyl dissipated from vegetation, litter, and soil rapidly applied aerially (April 1985) to a 445 ha Mississippi forested watershed and by ground application to a 4 ha watershed area (June 1985) in Florida at a rate of 0.42 kg a.i./ha. Residues in vegetation were not detectable after 90 days. SERA (1998a) cite a foliar half-life of 10 days based on a report by Knisel et al (1992 as cited in SERA 1998a).

Behavior in Soil and Groundwater

According to WSSA (1994), typical field half-life is 20-28 days at pH 6-7, but persistence is increased by cool temperatures, low soil moisture and higher pH. Microbial degradation does occur, but slowly. Non-microbial hydrolysis appears to be an important mechanism in sulfometuron methyl dissipation. It is moderately rapid at pH 6, but quite slow at pH 8. Sulfometuron methyl degradation in soil occurs most rapidly at lower pH values where it is dominated by hydrolysis causing cleavage of the sulfonylurea bridge. Mobility in soil is greater at higher pH values and lower levels of organic matter. Volatilization from soil is insignificant. Neary and Michael (1989) reported half-lives for sulfometuron methyl of 5 and 33 days on forested sites in Mississippi and Florida.

Cambon et al. (1992) reported that the mechanism for soil degradation of sulfometuron methyl is most likely chemical, based on finding the Arrhenius relationship [rate of chemical reactions in relation to temperature] was followed for temperatures up to 70 ºC in lab tests.

Soil half-lives determined for sulfometuron methyl are shown in Table 2-38.

Table 2-38
Sulfometuron methyl field dissipation study half-life (t ½ in weeks) determinations (modified from Table 1 in Trubey et al. 1998)

Location	Field type	Treatment date	$t_{½}$ (weeks)	Reference
Newark, DE	bare ground	July 3, 1980	2-3	Anderson and Dulka (1985)
Raleigh, NC	bare ground	July 18, 1980	2-3	Anderson and Dulka (1985)
Rosetown, SK	bare ground	July 29, 1980	5	Anderson and Dulka (1985)
Pendleton, OR	bare ground	Sept. 8, 1980	13	Anderson and Dulka (1985)
Ft Collins, CO*	bare ground	Nov. 14, 1980	21	Anderson and Dulka (1985)
Gainesville, FL	pine forest	June 13, 1985	0.7	Michael and Neary (1993)
Wahalak, MS	pine forest	April 9, 1985	4.7	Michael and Neary (1993)
Greenville, MS	bare ground	May 13, 1991	14	Trubey et al. (1998)
Rochelle, IL	bare ground	May 15, 1991	12	Trubey et al. (1998)
Uvalde, TX	bare ground	May 15, 1991	15	Trubey et al. (1998)
Madera, CA	bare ground	April 3, 1991	25	Trubey et al. (1998)

A series of field studies reviewed by SERA (1998a) reported on the persistence of sulfometuron at several sites (Table 2-39).

Table 2-39
Soil persistence of sulfometuron methyl at field sites

Location	Rate of Application	Time after application	Proportion remaining
Delaware	1.1 kg/ha	I year	1%
North Carolina	0.91 kg/ha	1 year	1%
Oregon	0.44 kg/ha	2 years	3%
Colorado	0.15 kg/ha	78 weeks	9%
Saskatchewan	0.11 kg/ha	2 years	5%

From SERA 1998a.

In a laboratory study (as reported by SERA, 1998a) sulfometuron applied at 120 g/ha showed a half-life of about 1 month. There was no degradation in a sterile soil, suggesting the decomposition was microbially mediated. In a separate study, SERA (1998a) reported complete decomposition of sulfometuron after 1 year in both a sterile and non-sterile soil systems. In these systems, the half-life was 17 and 96 days in the non-sterile system, and 53 days in the sterile system.

Michael and Neary (1988) reported that residues of sulfometuron methyl dissipated from litter and soil rapidly after aerially application to a 445 ha Mississippi forested watershed and by ground application to a 4 ha watershed in Florida (application rate was 0.42 kg a.i./ha). Residues in litter were still detectable (>0.050 ppm). In the soil profiles sulfometuron methyl was not detected below 30 cm and was not detected after 60 days (detection limit for soil was 0.020 ppm).

Harvey et al. (1985) reported that C_{14}-labeled sulfometuron methyl was mobile on soils when tested using thin-layer plates and soil columns. Hubbard et al. (1989) reported that during simulated rainfall tests, sulfometuron methyl was primarily lost due to percolation on two sandy Coastal Plain soils (Red Bay loamy sand and Bonifay sand) whereas the primary route of lost herbicides from a clayey soil, Greenville sandy clay loam, was from surface runoff (Table 2-40)

Table 2-40
Percent of sulfometuron methyl lost during 2-hour simulated rainfall trials on three Coastal Plain soils (modified from Hubbard et al. (1989)). Sulfometuron methyl was applied at a rate of 0.6 kg/ha.

	Low intensity (43 mm/hr)		Medium intensity (75 mm/hr)		High intensity (125 mm/hr)	
Soil	Surface runoff (%)	Percolation (%)	Surface runoff (%)	Percolation (%)	Surface runoff (%)	Percolation (%)
Greenville sandy clay loam	4.2	0.2	24.5	>0.1	34.7	1.0
Red Bay loamy sand	0.0	71.3	1.0	82.2	15.9	81.3
Bonifay sand	0.2	56.1	2.5	53.9	24.3	11.8

According to a study by Stone et al. (1993) reviewed by SERA (1998a) sulfometuron methyl and its metabolites from the phenyl portion of molecule are mobile in most soils. More so in sandy vs. loamy soils and less so in high organic matter soils and less so in soils at pH 6 and below. In field lysimeters with intact soil columns of bare sandy soils from the northern lake state forests sulfometuron methyl applied at 42.5 g a.i./ha, the mean concentration in soil water was 0.5 μg/L at 10 cm and 0.4 μg/L at 20 cm and none detected in soil water below 20 cm. By 80 days post-treatment most of the compound had been degraded or irreversibly sorbed into the upper soil layers. Mean concentrations detected from 80 to 130 days post-treatment were 0.5 μg/L at 10 cm, 0.4 μg/L, and 0.0 μg/L at 40 and 150 cm.

Based on this information, the investigator concludes that sulfometuron methyl will be neither very persistent nor mobile in the types of soils found on most forest associated ROW. The reasons are higher level of organic matter, the lower pH and the degree to which infiltration is promoted by a layer of vegetation and litter. Applications on bare and/or compacted soils may results in runoff of this herbicide and possible damage to adjacent vegetation.

Behavior in Surface Waters

There is little data on sulfometuron residues on surface water. However, the investigator believes residue levels will be similar to that for other herbicides (adjusted for rates of application), since the dominant route of entry is drift or direct application. These processes are largely independent of the herbicide.

The most relevant data are from Neary and Michael (1989) who followed an application of 0.36 lbs metsulfuron methyl a.i./acre to forests in the southeastern US. Monitored levels of sulfometuron methyl in ambient water ranged from 0.005 mg/L to 0.044 mg/L at a 445 ha treated site in Mississippi and at a 4 ha watershed in Florida. Herbicide was detected only briefly after application, with detectable residues in only 10 out of 85 samples collected. Herbicide residues did not persist beyond 7 days at the Florida site or 63 days in Mississippi.

In a simulated runoff study, Wauchope et al. (1990) report that 1 to 2% of sulfometuron methyl applied at a 0.4 kg/ha rate was lost in runoff (simulated rainfall event equaled 69 mm/h until 2 mm of runoff occurred) from a loamy sand soil one day after application on plots with bare ground or with grass cover. Losses were similar for both suspension and emulsifiable concentrate formulations. Authors reported that total losses were sensitive to the length of time between rainfall initiation and runoff initiation.

Harvey et al. (1985) reported that C_{14}-labeled sulfometuron methyl was stable in water at pH 7 and 9, but hydrolyzed readily at pH 5.0 with a reported half-life of 2 weeks. Products of hydrolysis were [^{14}C]methyl 2-(aminosulfonyl) benzoate and [^{14}C] 1,2-benzisothialzol-3-one, 2,3-dihydro 1,1-dioxide (saccharin). C_{14}-labeled sulfometuron methyl was completely photolyzed to $^{14}CO_2$ under aquatic photolysis conditions.

SERA (1998a) estimated an ambient sulfometuron methyl concentration from forestry activities over prolonged periods of 0.0002 (0.00005-0.0005) mg/L. In an accidental spill, they estimate ambient levels are likely to be about 0.33 mg/L with a range of 0.053-2.29 mg/L. They note that sulfometuron is stable in water at pH 7 or 10, but it decomposes appreciably in water at pH 5. It is important to put the SERA (1998a) analysis in perspective. They are projecting the larger areal scale of usage that might occur in traditional forest management operations. The investigator concludes that the patterns of use on a ROW would result in much lower ambient levels on a watershed scale, although the projected SERA values might hold for very small areas in close proximity to a ROW.

Residues in Animals

Sulfometuron methyl does not accumulate in game animals or fish, which is consistent with its low octanol-water partition coefficient (K_{ow}). One study in bluegill sunfish found no bioconcentration after exposure to sulfometuron. In addition, no bioconcentration occurred in channel catfish exposed to aged sediments containing sulfometuron methyl. Sulfometuron methyl does not accumulate in game animals or fish. Harvey et al. (1985) reported that C_{14}-labeled sulfometuron methyl has a low partition ration (0.31) and did not accumulate in fish tissue when bluegill sunfish were exposed to 0.01 ppm or 1.0 ppm C_{14}-labeled sulfometuron

methyl. This lack of bioconcentration is consistent with its low octanol-water partition coefficient (K_{ow}).

Wildlife Risk Analysis

Avian and Terrestrial Species

The mammalian toxicity of sulfometuron methyl is relatively well characterized in experimental mammals; however, there is relatively little information regarding nontarget wildlife species. It seems reasonable to assume the most sensitive effects in wildlife mammalian species will be the same as those in standard experimental mammals such as the rat, mouse, rabbit and dog (SERA, 1998a). In these species, the lowest NOEL value reported in WSSA (1994) is 50 ppm in the diet as part of a 24-month dietary exposure to rats. A daily dose of 2.5 mg/kg/day for 2 years had no effect.

Data on toxicity to wildlife species are in WSSA (1994), which is presumably included in the registration materials provided by DuPont to the USEPA. Some specific wildlife toxicity data are in Table 2-41.

Table 2-41
Toxicity of sulfometuron methyl to wildlife

Species	Type of Exposure	Response Measured	Value Reported
Rat	Oral	LD_{50}	> 5000 mg/kg
Rabbit	Dermal	LD_{50}	> 2000 mg/kg
Rat	24-month Diet	NOAEL	50 ppm
Bobwhite Quail	8-day Dietary	LC_{50}	> 5620 ppm
Mallard Duck	Oral	LD_{50}	> 5000 mg/kg
Mallard Duck	8-day Dietary	LC_{50}	> 5000 ppm

From WSSA 1994 or USDA 1989

All of the data suggest a very low order of toxicity to wildlife species. Birds appear to be somewhat less sensitive than experimental mammals to the toxic effects of sulfometuron methyl. Dose levels of 312 mg/kg or less were not associated with signs of toxicity or changes in body weight (SERA, 1998a).

Bees are less sensitive than either mammals or birds to sulfometuron methyl (SERA, 1998a). Dosages up to 100 μg/bee (1075 mg/kg) resulted in no mortality.

The wildlife risk assessment is in Table 2-42.

Table 2-42
Wildlife risk analysis for sulfometuron methyl

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	0.8	9.6	1000	5000	mallard
Mammals	1.6	21.2	1000	5000	rat
Amphibians	0.9	10	1000	5000	mallard
Reptiles	0.47	6	1000	5000	mallard

[1] The common laboratory test species used to represent the group of wildlife species.

None of the dose estimates approach 1/5 LD_{50}, leading us to conclude that there is no substantive risk to avian or terrestrial wildlife species from the use of sulfometuron methyl on electric utility ROW.

Aquatic Species

Sulfometuron methyl toxicity is also low to aquatic species. Some toxicity values are in Table 2-43.

Table 2-43
Toxicity of sulfometuron methyl to aquatic species

Species	Type of Exposure	Response Measured	Value Reported
Daphnia	In Water	48-hour LC_{50}	>12.5 mg/L
Crayfish	In Water	96-hour LC_{50}	> 5, 000 m g/L
Bluegill Sunfish	In Water	96-hour LC_{50}	> 12.5 mg/L
Rainbow Trout	In Water	96-hour LC_{50}	> 12.5 mg/L

From WSSA 1994 or USDA 1989

Acutely toxic effects in fish are not likely to be observed at concentrations less than or equal to 150 mg/L. Based on assays of fathead minnow embryo hatch, larval survival, or larval growth over 30-day exposure periods, no adverse effects would be expected at concentrations up to 1.17 mg sulfometuron methyl/L, which is the NOEL value used in this analysis for aquatic species (SERA, 1998a).

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria

proposed by USEPA where the risks of adverse effects to fish or invertebrates are defined in Table 2-44.

Table 2-44
Definition of "Q" values

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-45.

Table 2-45
Aquatic species risk analysis for sulfometuron methyl

		Off-Site Drift			Accidental Direct Spray	
	Critical LC_{50}	Typical	Maximum	Spill in Pond	Typical	Maximum
	PPM	----------Q Value----------				
Sulfometuron methyl	>12.5	<0.1	<0.1	<0.1	<0.1	<0.1

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to any of the representative aquatic species for typical and maximum exposures to sulfometuron methyl resulting from off-site drift, spill in a pond or accidental direct application to surface water. All Q values are less than 0.1.

Other Aquatic Species

Phytotoxicity of sulfometuron methyl to an aquatic macrophyte, *Myriophyllum sibiricum* was studied under laboratory conditions to determine the concentration causing 25% and 50% inhibition (IC_{25} and IC_{50}). Roshon et al. (1999) found *M. sibricum* was highly sensitive to sulfometuron methyl with IC_{25} and IC_{50} for shoot growth of $1.6^{-4\ ppm}$ and $3.7^{-4\ ppm}$, respectively. A similar range of values were reported for root numbers and root dry mass, indicating that sulfometuron methyl is phytotoxic to *M. sibricum*.

Sulfometuron methyl also appears to be relatively non-toxic to aquatic invertebrates, based on acute bioassays in daphnia, crayfish, and field-collected species of other aquatic invertebrates. Acute toxicity studies using Oust with daphnia yielded an NOEL of 2400 mg/L. One daphnia reproduction study with sulfometuron methyl (not formulated) noted a decrease in the number of neonates at 24 mg/L but not at 97 mg/L or any of the lower concentrations tested (SERA, 1998a). The investigator considers the effect observed at 24 mg/L to have been a random

variation, and considers it as the NOEL for purposes of this risk assessment to aquatic invertebrates.

Other than for possible effects on aquatic vegetation sulfometuron methyl should not affect other aquatic species. Some caution to avoid entry to surface waters appears warranted to protect aquatic macrophytes.

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

Sulfometuron has a vapor pressure that is extremely low. There should be no significant vaporization from droplets or environmental surfaces, meaning that residues in air will be restricted to the distribution of droplets during the application. This will largely eliminates the environmental risk from residues in air.

Soil and Groundwater

In a lysimeter leaching study using sandy soil Stone et al. (1993) reported no detectable sulfometuron methyl below 20 cm. When combined with limited soil persistence reported by others the investigator concludes that there is little likelihood of sulfometuron movement to ground water in most ROW settings.

PART 9. RISK ASSESSMENT FOR TRICLOPYR

Introduction

Triclopyr is effective in controlling of wide variety of woody plants by mimicking a natural plant growth hormone, thereby disrupting normal growth. Triclopyr is used in one of two forms in herbicides and is available in Garlon® 3A, Garlon® 4, Forestry Garlon® 4, Pathfinder® II, and Remedy® RTU and other commercial products. Application methods used in ROW management include backpack (selective) foliar, streamline and basal bark applications, and boom or hydraulic spraying with application rates ranging from 0.5 lb a.e./acre to 6 lbs a.e./acre, with 1 lb a.e./acre being common.

Several principal sources were used for this report and are not specifically quoted including: USEPA (1998a), ExToxNet (1996), SERA (1996a and 2003b), Norris et al. (1991), and various reviews submitted to USEPA to support the registration of triclopyr. It is not the purpose of this current review and assessment to repeat the analyses presented in earlier reports, but to provide a

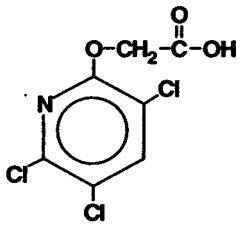
manageable discussion of environmental and wildlife risk associated with use of the herbicide, including recent research that may not have been available or included in the other reviews. The environmental fate of triclopyr and its metabolites are also discussed in detail by Cessna et al. (2002).

Physical-Chemical Properties and Environmental Behavior

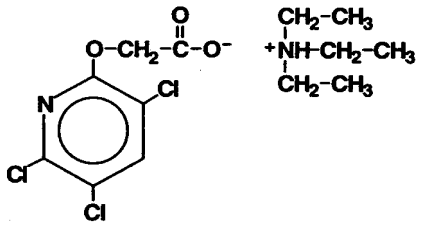
Physical-Chemical Properties

Triclopyr is the common name for [(3,5,6-trichloro-2-pyridinly)oxy]acetic acid. Two forms of triclopyr are used commercially as herbicides: the triisopropylamine salt and the butoxyethyl ester. Garlon® 3A contains the triethylamine salt of the triclopyr and Garlon® 4 and Pathfinder® II contain the butoxyethyl ester of triclopyr, kerosene and proprietary surfactants. Both the amine and ester forms of triclopyr hydrolyze to the acid or salt form, depending on the pH of the environment.

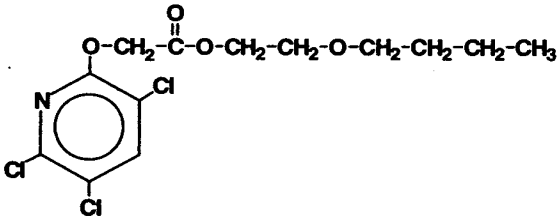
Deodorized kerosene, a constituent of the formulation, is classified by USEPA as a List 3 Inert, lacking in toxicological data. However, it has been reviewed by ATSDR (2002). At sufficiently high doses, kerosene can cause gastrointestinal, central nervous system (CNS), and renal effects.



Triclopyr Acid



Triclopyr Triisopropylamine Salt



Triclopyr Butoxyethyl Ester

Important physical properties of triclopyr acid and ester are in Table 2-46. Triclopyr acid and amine are ionized in the environment to the appropriate salt and can be considered as the acid.

Table 2-46
Physical, chemical, and biochemical properties of triclopyr

Property	Triclopyr Acid Value	Triclopyr Ester Value
Water Solubility	435 mg/L @ 25C, amine salt water solubility is 2,100 g/L	2.1 mg/L at 25°C
Vapor Pressure	1.50 x 10^{-6} mm Hg @ 25C	Not available
Log K_{ow}	0.42 (pH 5), -0.45 (pH 7), -0.96 (pH 9)	4.01
K_{oc}	20 to 59	560 to 780

From SERA 2003b

Environmental Behavior

SERA (1996a) provides a summary of the environmental behavior of triclopyr relative to the risk assessment process. It degrades rapidly in soils under environmental conditions that favor microbial growth (reported half-lives up to about 46 days); is rapidly degraded, primarily by photolysis, in water; and is readily eliminated by animals when ingested.

Behavior in Air

Although the ester form (Garlon® 4) is more volatile than the amine (Garlon® 3A) the vapor pressure of both forms is low, meaning the potential for vaporization to the atmosphere is low. Droplet sizes are for the most part too large to form an aerosol that can be inhaled, and the volume of distribution in atmosphere is enormous compared to the volume of air inhaled over time. Bentson and Norris (1991) found dissipation of the ester in both light and dark on glass slides and leaves, indicating hydrolysis is occurring. These factors indicate a low potential for entry to air other than in the form of spray droplets during the application. Specific studies of triclopyr residues in air are not in the published literature.

Behavior in Plants

Triclopyr is taken up by plants through both roots and foliage and is then readily translocated throughout the plant. It induces an auxin-type response in growing plants in that it appears to interfere with the normal growth process. Thus, maximal plant response to foliar application occurs if application is made soon after full leaf development and when there is sufficient soil moisture for plant growth (Dow Chemical Company, 1983a and 1983b).

Triclopyr ester is subject to loss from leaves in both light and dark, although on some species of vegetation the loss is greater in the light, indicating photo-degradation as well as other mechanisms are operating (Bentson and Norris, 1991).

Residue levels have been measured following application of triclopyr in field and greenhouse studies, however, Siltanen et al. (1981) reported on residue levels of triclopyr in cowberries,

bilberries, and lichen following application of Garlon® (specific formulation not given) on small experimental field plots in Finland. For application rates of 0.22 to 2.01 lbs active ingredient per acre (0.25 to 2.25 kg/ha), residues were 0.3 to 1.9 ppm (by weight) in cowberries at 30 days post-treatment, 0.2 to 4.0 ppm in bilberries at 8 days post-treatment, and 0.15 to 1.7 ppm in lichens at 13 months post-treatment. In all cases, increasing application rates resulted in increased levels of residue.

Triclopyr was applied once to cowberries at a rate of 0.67 lb active ingredient per acre (0.75 kg/ha) in mid-June, July, August, or September, and the berries were harvested in late September, such that intervals of 6 to 98 days existed between treatments and harvesting. Residues in the cowberries declined with residence time. These residues were 2.4 ppm 6 days post-treatment (September treatment), 0.7 to 1.1 ppm 30 to 36 days post-treatment (August treatment), 0.2 ppm 64 to 70 days post-treatment (July treatment), and 0.2 to 0.3 ppm 92 to 98 days post-treatment (June treatment). In bilberries similarly treated with 0.67 lb/acre (0.75 kg/ha) in mid-June, July, or August and harvested in late August, residues were 0.9 ppm (7 days post-treatment) to 0.7 ppm (69 days post-treatment) with no relationship between residence time and residue levels. It is unclear whether the differences in residue levels in cowberries is due to the time interval between treatment and harvest, or due to the environmental conditions, plant metabolism, and growth stage at the time of treatment with triclopyr (Siltanen et al., 1981).

Residues of triclopyr in greenhouse-grown honey mesquite were measured 3, 10, and 30 days following application of the ethylene glycol butyl ether ester or triethylamine salt of triclopyr to foliage, to soil, or to both foliage and soil (Bovey and Mayeux, 1980). The triclopyr was applied at a rate equivalent to 1 lb/acre (1.1 kg/ha). Maximum residues of 28 ppm were found in leaves 10 days after foliar application of the amine salt. Maximum stem residues of 3.6 ppm triclopyr were found 10 days after foliar application of both the ester and the amine salt. In general, foliar applications of triclopyr resulted in higher leaf and stem residues, while soil application resulted in higher root residues.

Bovey et al. (1979) measured triclopyr residues in greenhouse-grown huisache (*Acacia farnesiana*) following application of the triethylamine salt at rates equivalent to 1 and 2 lb/acre (l.12 and 2.24 kg/ha) as soil, foliar, and soil-plus-foliar treatments. Maximum residues of 37 ppm were found in the stems immediately after foliar application of 1 lb/acre and 3 days after 2 lb/acre application. Maximum residues of 37 ppm were also found in stems immediately after soil-plus-foliar application at 1 lb/acre. In general, highest residue levels were in the stem and intermediate levels were in foliage for both foliar and soil-plus-foliar applications. Highest levels occurred in roots for soil applications.

In an Oregon forest treated with triclopyr for forest management purposes, Newton et al. (1990) found the initial concentration of herbicide in foliage of over-story plants exposed to spray was about 44 ppm per kg/ha applied. Under-story foliage was about 17 ppm per kg/ha applied. Residues persisted in the dead foliage, but showed little tendency to leach with rainfall towards the soil surface. Decomposition in the litter was more rapid, with estimated half-life for both Garlon® 3A and Garlon® 4 reported to be 31 days.

In northern Idaho, triclopyr ester was applied to forest vegetation and the residue levels measured at various times after application. The results are in Table 2-47.

Combining the data for all species, Whisenant and McArthur (1989) found 42% decline in residue levels after 6 days, 72% in 28 days and more than 98% in 365 days. The herbicide concentration on the foliage immediately after application varied among species, ranging from 79 to 362 ppm after application at 2.3 kg/ha. These data are particularly helpful in seeing the pattern of exposure organisms which consume treated vegetation would experience, and when combined with the data of Siltanen et al. (1981) indicate a half-life of triclopyr in vegetation of 30 days in berries and less than 7 days in foliage.

Table 2-47
Triclopyr levels in terminal branch and leaf segments following application of 2.3 kg of triclopyr ha^{-1} to a southwest slope near Boville, Idaho

	Days after Applications					
Species	1	6	28	76	277	365
	(PPM) Southwest slope					
Shinyleaf ceanothus	362.6	253.7	96.2	46.1	24.5	4.7
Douglas fir	198.9	81.5	9.4	7.6	3.3	1.2
	Northeast slope					
Shinyleaf ceonothus	354.7	249.9	187.0	74.1	25.7	6.7
Douglas fir	151.4	8.0	20.3	6.7	2.4	1.5
Mountain clover	176.3	7.0	32.3	15.3	7.6	2.6
Woods rose	79.1	9.7	33.2	-----	-----	-----
Sticky currant	100.4	68.6	35.7	-----	-----	-----
Snowberry	108.6	81.5	49.9	-----	-----	-----
Grasses	156.7	53.6	11.4	4.7	1.2	0.2

Adapted from Whisenant and McArthur 1989

Behavior in Soil and Groundwater

Triclopyr has a short persistence in soil due to microbial degradation. It is not strongly adsorbed to soil particles and therefore is potentially mobile in soil (Dow Chemical Company, 1983b). Although there is no direct evidence of photodegradation of triclopyr in soils, its rapid photolysis in water indicates that, when on the surface of soils, it is likely to photodegrade. Because of its low vapor pressure, volatilization of triclopyr should not be a major means by which triclopyr is lost from the soil (Ghassemi et al., 1981).

Although triclopyr is generally considered to have a short persistence in soils, the actual half-life is strongly dependent on specific soil type and climatic conditions. McKellar et al. (1982) estimated the half-life of triclopyr residues in soil in a small watershed in West Virginia to be between 14 and 16 days. However, much of the aerially applied triclopyr (Garlon®3A applied at a rate of 10 lb/acre [11.2 kg/ha]) was intercepted by foliage. Average triclopyr residues in soil from the treated area in this study, measured on the day of treatment, were nondetectable in densely wooded areas, 4.4 ppm in lightly wooded areas, and 18 ppm in open areas. Residues in soil from the latter two areas declined to <1 ppm within 4 weeks and were nondetectable at 28 weeks and at subsequent sampling periods. The Dow Chemical Company (1983a) reports a half-life of 10 days in a silty clay loam and 46 days in a loam. The loam used in this latter study was maintained in the laboratory at 95°F (35°C) with moisture at field capacity. The Dow Chemical Company (1983b) reports the average half-life of triclopyr in soil to be 30 days. Laskowski et al. (1982) report a soil half-life of 40 days for triclopyr. An average half-life of 46 days is reported by Weed Science Society of America (WSSA, 1983) and in Ghassemi et al. (1981).

In a field study in Sweden, Torstensson and Stark (1982) applied 2.0 lb Garlon®3A active ingredient/acre (2.2 kg/ha) and 1.7 lb Garlon® 4 active ingredient/acre (1.9 kg/ha) to 8 different forest soils. Residues of triclopyr persisted for 1 to 2 years, and in some cases in excess of 2 years, at levels approximately 10 percent or less of residues sampled immediately after application. It should be noted, however, that measured summer soil temperatures were usually only 55.4°F (13°C) or less and never more than 57.2°F (14°C). These temperatures are not particularly favorable to microbial degradation, and warmer soil temperatures, abundant soil moisture and the organic content of forest related soils provide a more favorable environment for triclopyr degradation.

The soil half-lives for triclopyr reported in studies examined by Cessna et al. (2002) were variable and ranged from 8 to 96 days; however, the half-life of the ester was short and usually less than a few days. When triclopyr ester was applied to a sandy loam soil, the half-life of triclopyr acid was calculated to be 10.6 days, following the very rapid hydrolysis of the ester (half-life 1.1 days). (Petty and Gardner, 1993 as cited by Cessna et al., 2002).

Triclopyr applied to rice paddies at rate of 0.43 kg a.i./ha after emergence dissipated rapidly with initial dissipation of 50% of the initial concentration (DT_{50S}) in paddy water, dryland-rice soil, and bareground soil being 10 days or less (Johnson et al., 1995). Triclopyr was not detected in water 28 days after application and was slightly more persistent in soil than water. Poletika and Phillips (1996 as cited by Cessna et al., 2002) reported first-order half-lives for triclopyr applied to rice fields before and after flooding to vary from 2.2 to 7.6 days and from 1.8 to 3.4 days, respectively.

Newton et al. (1990) reported on the movement and persistence of picloram and triclopyr in a SW Oregon forest (warm dry summer climate). The relevance of the Newton et al. (1990) study is during the periods of the study when there was precipitation. During this period they reported the half life of the herbicides varied between 11 and 25 days.

Triclopyr rapidly adsorbs onto soil particles. The degree of adsorption is dependent on soil type, with soil organic matter the primary parameter involved. Increased soil organic matter results in

increased adsorption of triclopyr. A major degradation product of triclopyr, trichloropyridinol, is less mobile than the parent triclopyr (Dow Chemical Company data reported in Ghassemi et al., 1981). Wolt (1998 as cited by Cessna et al., 2002) found that triclopyr adsorption in soil is strongly related to soil surface cover, soil organic matter content and pH; however, the physical properties of the herbicide and field studies suggest that triclopyr has a limited potential for leaching.

In a soil composed of loam and sand with low amounts of organic matter (0.62% organic carbon), 75 to 80 percent of applied triclopyr leached through a 12-inch column between days 11 and 15 when 0.5 inch of water was applied daily. The trichloropyridinol metabolite required twice as much water to leach through the soil than did the parent triclopyr. Leaching of triclopyr and its two metabolites, trichloropyridinol and trichloromethoxy pyridine, was studied in six soils in various parts of the United States following field application of Garlon®3A at a rate of 3 gal/acre under conditions of normal rainfall. Small amounts of triclopyr and the metabolites were observed in the 6 to 12 inch and in the 12 to 18 inch soil layers 4 to 8 weeks after application. The trichloropyridinol degradate leached less than the parent triclopyr, with maximum residues in the 0 to 6 inch layer at 4 to 8 weeks. Residue levels for this compound subsequently declined. Trichloromethoxy pyridine maintained a concentration of 0.1 ppm or less in all soil layers sampled (Dow Chemical Company data reported in Ghassemi et al., 1981).

In a recent laboratory study of triclopyr adsorption in soil, Pusino et al. (1994) found that organic matter content and pH were both important factors in determining the soil behavior of triclopyr. Generally, a more acid pH, such as that found in forest soils of ROW, and increasing organic matter increased the degree and strength of adsorption to soil materials. This work is important because it provides a strong theoretical basis for understanding triclopyr behavior observed in field studies, and shows that the lower pH and higher organic matter levels found in forest soils will reduce triclopyr persistence and mobility.

A recent report by Johnson and Lavy (1994) provides data helpful in assessing the persistence of triclopyr in soils subjected to periodic flooding. While their study was to simulate rice production strategies, it is a reasonable approximation of wetland conditions or riparian zones where flooding is common, and triclopyr use might occur. At 2 cm, 20 cm and 60 cm depths, they found no detectable herbicide remaining at 280 days, 184 days and 736 days after application, respectively. The half lives at these depths were 10, 10 and 39 days, according to first order kinetics - with 11 % of the original amount remaining after 34 days at 2 cm, 13% remaining at 20 cm and 54% remaining at 60 cm. These results show triclopyr continues to dissipate under conditions where soils are periodically inundated.

A highly relevant study of triclopyr behavior in a Canadian forest soil was conducted by Lee et al. (1986). In their study triclopyr and its ethylene glycol butyl ether ester (EGBE) were applied, separately, to the top layers of columns packed with a loam soil or with quartz sand. Water, equivalent to 2.5 cm of precipitation, was leached through each column every second day. After 54 days, residues were found only in the top 10-cm layers of the soils and sand, from 65% to 100% of the triclopyr applied was found in the eluate after 54 days of leaching.

These results show triclopyr mobility is limited, and that decomposition proceeds in a forest soil where organic matter content is 34%, even under significant leaching pressure (2.5 cm precipitation on alternate days over a 54 day period). These results parallel those of Duebert and Corte-Real (1986), who studied soil residues of triclopyr after selective foliar application on a right-of-way in New Hampshire. They found all of the triclopyr in the surface 10 inches of soil throughout the 2.5 month study, with residues becoming either non detectable or showing a 90% level of dissipation level in this time. On a New York right-of-way, triclopyr did not persist at measurable levels past 10 weeks following foliar application (water carrier). Triclopyr in oil applied as a basal treatment persisted up to 18 weeks.

In another Canadian study by Stephenson, et al. (1990), movement and persistence of triclopyr were evaluated in both a clay soil and a sandy soil in northern Ontario, Canada. They evaluated both vertical movement (leaching), and lateral movement (runoff) down a slope. They reported 50% and 90% disappearance in 2 weeks and 4 weeks, respectively, regardless of soil type. Evidence of triclopyr leaching in response to heavy rainfall was observed 7 days after application in both soils, but residues never exceeded 6 ug/kg at a depth of 25-30 cm. when present. In a study of lateral movement of triclopyr with runoff water, residues (in the range 0.01-0.96 ug/L) were recovered in a collection ditch 12-13 m downslope; however, there was no evidence of mass movement of triclopyr at quantifiable levels (0.54 ug/kg) downslope in the soil.

Field studies of persistence and mobility confirmed earlier laboratory results and indicate that environmental problems are unlikely to occur as a result of excessive triclopyr persistence and/or mobility in soils typical of Northern Ontario forestry areas. Even less is expected in the warmer, acid, relatively high organic soils on forest related ROW in the U.S.

Behavior in Surface Waters

McKellar et al. (1982) monitored residues of triclopyr and trichloropyridinol in streamflow draining a small West Virginia watershed that had received application of 10 lb/acre (11.2 kg/ha) Garlon®3A by helicopter. Water samples were taken periodically up to 510 days after treatment from two streams running through the treated area. Sample sites were 100 feet downslope from the treated area for each stream, with a third sample site at a weir where the two streams joined. Residues in these samples ranged from nondetectable to 0.02 ppm at the weir (about 200 feet from the treatment area) in one stream, and from nondetectable to 0.08 ppm in the second stream. No residues of trichloropyridinol were detected in any water samples during the course of this study.

Rainfall relatively soon after application of triclopyr may result in runoff loss into watersheds due to material being desorbed from soil particles and carried in solution, or carried adsorbed onto eroded sediment (Ghassemi et al., 1981). Monitoring studies of runoff in a small Oregon watershed found residues of 6 ppb in the runoff water 5 months after treatment with 3 lb/acre (3.36 kg/ha) of triethylamine salt of triclopyr and approximately 59 inches of natural rainfall (Norris et al., 1976 summarized in USEPA registration data reported by Ghassemi et al., 1981). In a study of lateral movement of triclopyr with runoff water, Stephenson et al. (1990) reported triclopyr residues (in the range 0.01-0.96 ug/L) were recovered in a collection ditch 12-13 m

downslope; however, there was no evidence of mass movement of triclopyr at quantifiable levels (0.54 ug/kg) downslope in the soil.

Thompson et al. (1991) reported on the fate of triclopyr butoxyethyl ester following direct aerial application to a boreal forest stream. The average deposit monitored at the stream surface was 3.67 kg acid equivalent (a.e.)/ha (range = 3.35-3.99 kg a.e./ha). Residues of triclopyr up to 0.35 mg/L were found due to direct over spray of the water course. Average concentrations of triclopyr ester in stream water ranged from 0.05 to 0.11 mg/L during the first 12 to 14 hours post-application and declined to levels below the limits of quantification (0.001 mg/L) within 72 hours post-application. Transient residues of triclopyr acid were observed in stream water, with a maximum concentration (0.14 mg/L) 6 hours post-application. Results indicate that natural dissipation mechanisms reduce both the period and the concentrations to which aquatic organisms would be exposed. Thompson et al. (1995) reported the fate of triclopyr ester in a first-order stream in Ontario, Canada. Maximum concentrations of the ester were 0.848 and 0.949 μg/ml at sampling stations nearest two injection points. Average triclopyr ester concentrations ranged from 0.23 μg/ml at stations nearest to injection points to 0.02 μg/ml approximately 225 m downstream. Periods of exposure to aqueous triclopyr ester in excess of 0.001 μg/ml ranged from 55 minutes in fast-flowing upstream locations to 120 minutes at slower downstream locations. Natural mechanisms rapidly converted the ester to the acid.

In a study of the operational application of triclopyr as Garlon® 4 (and other herbicides) on electric utility rights-of-way in New York, Norris (1991) monitored the entry of herbicide to water on or near the right-of-way. Buffer strips of 30 to 50 feet were used depending on the method of application. He found triclopyr residues in five samples, at concentrations or 0.001 or 0.002 ppm. A few other samples contained trace amounts of triclopyr (less than 0.001 ppm). None of the samples with triclopyr were collected the day of application, indicating the buffer strips prevented stream contamination during application. The samples with detectable residues of triclopyr were collected during the first storms after application, indicating some movement of this herbicide from treated areas into the water. The concentrations found were all well below levels that are harmful to aquatic organisms, or other water users.

Triclopyr dissipation in natural waters (static and flowing), and in sediments is attributed to photodecomposition and microbial activity (Cessna et al., 2002). Photodegradation in water is rapid, with a half-life as short as 10 hours at 77°F (25°C) (WSSA, 1983). The ester form of triclopyr is rapidly converted in water by hydrolysis to the acid form (Dow Chemical Company, 1983b).

McCall and Gavit (1986) measured the aquatic photodecomposition of triclopyr acid and ester. They found a half life of 26.8 hours for the ester. The ester form also disappeared in the dark, although with a longer half life (140 hours), indicating both photolysis and hydrolysis of the ester occurs. The ester form is converted to the acid form, which is subject to photodecomposition as well, with a half life of 5.4 hours.

The persistence of triclopyr (applied at 0.3 or 3.0 kg/ha triclopyr as the ester) was studied in field enclosures in aquatic systems in northern Ontario, Canada by Solomon et al. (1988). Residue levels in both water and the sediment in the system were measured. They found approximately

99% of the triclopyr remained in the water and 1 % was adsorbed to sediment. Dissipation from both systems was rapid, with a 50% dissipation reported to be about 4 days, and 95% dissipation to be 16 days.

McCall, Lazkowski and Bidlack (1988) used a simulation model to estimate the aquatic fate of triclopyr ester based on studies of its persistence in various studies. The results showed the ester concentration in pond water would decline with a half life of 6 to 24 hours. Woodburn (1993) evaluated the environmental dissipation of triclopyr (as the triethylamine salt) under aquatic-use conditions using an application rate of 10 gal/acre.

In Lake Seminole (Georgia) the half-life for aqueous phase triclopyr ranged from 0.5 to 3.6 days. Triclopyr did not accumulate in the sediment. The half-life of triclopyr metabolized by aquatic plants averaged 4 days (Woodburn, 1993).

Residues in Animals and Fish

Triclopyr is easily absorbed from the digestive tract of animals and poorly absorbed through the skin. It is rapidly excreted by the kidney and is not stored in any tissue. Residues in game animals are not toxicologically significant and will not be detectable a few days after exposure ends. In the Lake Seminole, Georgia study Woodburn (1993) found that fish did not exhibit any bioconcentration of triclopyr or its decay product, TCP. Only trace amounts of either compound were found in fish tissue. However, clams and crayfish contained detectable residues of triclopyr with an observed half-life of 1.5 days in clams and 12 days for crayfish.

Lickly and Murphy (1987) also showed the limited degree to which triclopyr and its metabolites accumulate in fish. At 2.5 ppm exposure level in water, they report a bioconcentration factor of 0.03 for the flesh (edible portion for humans) and 0.5 for the whole fish (potentially edible by other animals). The predominant residue was triclopyr, with the remainder as triclopyr-pyridinol or -pyridine metabolites. Barron, Hansen and Hall (1991) studied the pharmacokinetics and metabolism of triclopyr in crayfish and also found a low potential for accumulation in this species.

Johansen and Geen (1990) measured the concentration of triclopyr and triclopyr ester in fish killed due to their exposure to triclopyr ester. They found bioconcentration factors of 1.7 for the ester, 9.63 for the triclopyr and 0.6 for the pyridinol metabolite. The higher factor for triclopyr probably is the result of the hydrolysis of the ester in the fish rather than the accumulation of the triclopyr from the water, given its low fat solubility and high water solubility. The residue levels were generally low, being a maximum of 1.72 ppm triclopyr ester, 8.19 ppm triclopyr and 0.74 ppm pyridinol, all in fish exposed to 1 ppm triclopyr ester.

Barron et al. (1990) measured the pharmacokinetics and metabolism of triclopyr ester in coho salmon yolk-fry. As part of their study, they also compared the toxicity of triclopyr ester to yolk-sac fry and juvenile coho. In yolk-sac fry, which were killed by exposure to 0.56 ppm triclopyr ester, they found a total body burden residue level of 65 ppm with more than 99% of this as triclopyr, showing rapid hydrolysis of the ester occurs in the fish. Figure 3-3 shows the pattern of both the acid and ester forms of triclopyr in water and fish, showing the relatively rapid

hydrolysis of the ester in both mediums, and the clearance of both compounds with time from the fish - even during the period of exposure.

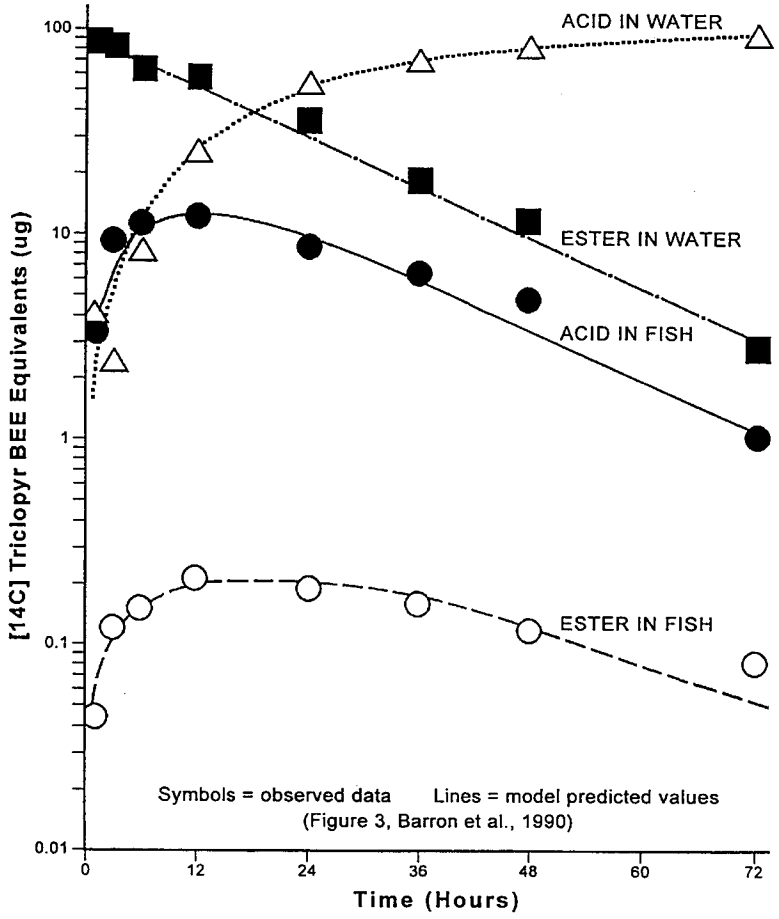


Figure 2-1
Kinetics of Triclopyr BEE (Ester) and Triclopyr Acid (Acid) in Fish and Water during Static Exposure to Triclopyr BEE (Barron et. al. 1990)

Wildlife Hazard Analysis

Most of the information on the toxicology of triclopyr is unpublished data from studies conducted by the registrant for the purpose of reregistration of the herbicide with USEPA. The principle sources for this section are USDA Forest Service (1989 and 1997), USEPA (1998 and 2002b), and SERA (2003b) and are not specifically cited.

Avian and Terrestrial Species

Triclopyr is moderately toxic to mammals based on LD_{50} values that range from 310 mg/kg in guinea pigs to 729 mg/kg in male rats (USEPA, 1985b; SERA, 1996a). The Garlon® 3A and Garlon® 4 formulations are slightly toxic, with oral LD_{50}'s of 2,830 and 2,140 mg/kg in rats (males and females, respectively) (Dow Chemical Company, undated). Ponies exposed to four

daily doses of 60 mg/kg of triclopyr exhibited no adverse effects; however, exposure to four daily doses of 300 mg/kg caused depression, recumbency, decreased gastrointestinal activity, and respiratory and muscular distress (Osweiler, 1983).

Based on acute oral and dietary studies, triclopyr, Garlon® 3A and Garlon® 4 are slightly toxic to birds. The acute oral LD_{50} of technical triclopyr is 1,698 mg/kg for mallard ducks, and the dietary LC_{50} ranges from 2,935 to greater than 5,000 ppm (Dow Chemical Company, undated; Kenaga, 1979). The dietary LC_{50}'s of Garlon® 3A and Garlon® 4 are all greater than 9,000 ppm (Dow Chemical Company, undated). A one-generation reproduction study showed no reproductive effects, symptoms of toxicity, or abnormal behavior when mallards were given up to 500 ppm in their diet for a 20-week period, including 10 weeks prior to egg laying and 10 weeks during egg laying (Dow Chemical company, 1987). A similar study reported no reproductive or toxic effects in bobwhite quail exposed to dietary levels of up to 800 ppm for a 20-week period, including 11 weeks prior to egg laying and 8 weeks during egg laying (Dow Chemical Company, 1987).

A frog embryo teratogenesis assay was done to determine the effects triclopyr may have on the embryonic development of the frog species *Xenopus laevis* (Perkins et al., 2000). The authors determined that the triethylamine salt had a LC5 and a LC_{50} of 119 and 162.5 mg a.e./L, respectively. For the butoxyethyl ester the LC 5 and LC_{50} were 6.7 and 9.3 mg a.e./L, respectively.

The effect of triclopyr on frog embryos and tadpoles was reported by Berrill, et al. (1994). Following exposure for 48 hours to triclopyr ester at concentrations from 0.6 ppm to 4.8 ppm. They found no effect on hatching rate or success, or on avoidance response, and no tadpoles died during a 9-day post treatment recovery period and there was no effect on body length when organisms were exposed in the embryo stage. When exposure in the tadpole stage occurred there was no effect at 0.6 ppm, but about half the organisms did not respond normally to a prodding response at 1.2 ppm, although all recovered within 3 days following exposure. At the higher concentrations all green frog and bullfrog tadpoles died, but few leopard tadpoles died although all were initially unresponsive to prodding. These data indicate concentrations of less than 0.6 ppm triclopyr as the ester will likely be safe for frogs in their various young life stages.

The acute contact LD_{50} of triclopyr in honey bees is greater than 60 ug/bee, indicating that it is relatively nontoxic to insects (Kenaga, 1969). The contact LD_{50} for honey bees is greater than 100 ug/bee based on a 1985 study (Dow Chemical Company, 1987).

The wildlife risk analysis is in Table 2-48.

Table 2-48
Wildlife risk analysis for triclopyr

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species1
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	5	75	340	1698	mallard
Mammals	8	150	62	310	guinea pig
Amphibians	5	75	340	1698	mallard
Reptiles	6	96	340	1698	mallard

[1] The common laboratory test species used to represent the group of wildlife species.

This analysis shows that there is an adequate margin of safety for wildlife species for typical rates of application, but that mammals are at some risk when maximum rates of application are used. Mitigation strategies are suggested in Section 4 of this report.

Aquatic Species

USDA Forest Service (1984), Norris (1991) and SERA (1996a) review the aquatic toxicity of triclopyr. The lowest toxicity values (LC_{50}) reported for Garlon® 3A are more than 100 ppm for fish and 895 ppm for crustaceans, and for Garlon® 4 the values are 0.74 ppm for fish and 2.2 ppm for crustaceans.

The butoxyethyl ester is highly toxic to fish, whereas the triethylamine (TEA) salt is practically nontoxic. The 96-hour LC_{50} for bluegill exposed to the butoxyethyl ester is 0.87 ppm and is 891 ppm for exposure to the triethylamine salt. Unformulated triclopyr also is practically nontoxic to aquatic organisms.

Wan, Moul and Watts (1987) conducted an intensive study of the toxicity of triclopyr in its various forms and its metabolites in juvenile Pacific salmon. While there was some minor variation among species, the average 96-hour LC_{50} for triclopyr as Garlon® 3A was 347 ppm and as Garlon®4, was 2 ppm. Pink salmon were most sensitive to Garlon 4 (LC_{50} 1.2 ppm) and Chum salmon were most sensitive to Garlon® 3A (LC_{50} 267 ppm). The pyridinol and pyridine metabolites are about the same toxicity as the triclopyr ester in Garlon® 4, but the authors conclude that these metabolites are not expected to occur at toxic levels in water. No-effect levels were not reported in this study, but they commonly are 20 to 30% of the 96-hour LC_{50}.

Servizi, Gordon and Martens (1987) reported 96-hour LC_{50} values for Garlon® 4 of 1.2 ppm for *Daphnia*, and sockeye salmon fry, 1.4 ppm for sockeye fingerlings and 2.2 ppm for rainbow trout and coho salmon fry. In testing these same species with Roundup® formulation of

glyphosate, the authors report 96-hour LC_{50} values ranging from 25.3 ppm for *Daphnia* to 42 ppm for coho salmon fry.

Barron et al. (1990) measured the toxicity of triclopyr ester to yolk-sac fry and juvenile coho. They found the yolk-sac fry had a 96-hour LC_{50}, which was 0.47 ppm, compared to 1.7 ppm in juveniles. Toxicity of triclopyr butoxyethyl ester to three salmonid species in three water types was determined by Wan et al. (1991). The 96-h LC_{50} for coho salmon in soft, intermediate and hard water was 1.8, 2.7, and 2.4 mg/L, respectively; for pink salmon was 1.1, 1.3, and 0.8 mg/L, respectively; and for rainbow trout was 2.4, 1.8, and 1.5 mg/L, respectively.

Janz et al. (1991) looked for physiological stress responses in juvenile coho salmon exposed to sub-lethal concentrations of Garlon® 4, Garlon® 3A and Vision (glyphosate) herbicides. They reported none at concentrations ranging from 5% to 80% of the 96-hour LC_{50}. Also looking for sub-lethal effects, Morgan et al. (1991) measured avoidance reactions and behavior responses of juvenile rainbow trout to these same four herbicide formulations. They reported the threshold exposure level for response was 37.5 ppm for Vision with 10% surfactant, 13.5 ppm for Vision with 15% surfactant, 0.6 ppm for Garlon® 4 and 200 ppm for Garlon® 3A. Johansen and Geen (1990) reported the LC_{50} of triclopyr as Garlon® 4 to juvenile coho salmon is 0.84 ppm. At concentrations greater than 0.56 ppm fish were distressed, at 0.32 ppm they were lethargic and showed reduced oxygen consumption. There was no mortality in a two-month post-exposure period in fish exposed to concentrations less than 0.56 ppm, although there were some behavior responses at lower levels. For instance, there appeared to be increased photoperiod sensitivity in fish exposed to 0.03 and 0.1 ppm triclopyr ester, although the response returned to normal in a 24-hour post-exposure period, indicating a transitory response.

For the salt of triclopyr, the adjusted LC_{50} (values adjusted for density and % a.i.) for channel catfish was determined to be 122.6 mg/L at 48-hours and 109.5 mg/L at 96-hours; the adjusted LC_{50} for bluegill sunfish was determined to be 313.0 and 226.3 mg/L, respectively; and the adjusted LC_{50} for crawfish was 1436.0 and 750.1 mg/L, respectively (Abdelghani et al., 1997).

Kruetzweiser et al. (1994) reported that the toxicity of triclopyr ester to rainbow trout and chinook salmon increased with exposure time, but the rate of increase in toxicity declined with increasing exposure duration. Median lethal concentrations for rainbow trout exposed to 1, 6, or 24 hours were 22.5, 1.95, and 0.79 mg a.e./L triclopyr ester, respectively. Median lethal concentrations for chinook salmon for the same durations were 34.6, 4.7, and 1.76 mg. a.e./L.

The risks of adverse effects from exposure to were estimated for representative aquatic species. In cases where no acute toxicity reference value was available for a representative species, a value was selected from the table of the most closely related species.

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations. The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by USEPA where the risks of adverse effects to fish or invertebrates as defined in Table 2-49.

Table 2-49
Definition of "Q" values

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

The aquatic species risk analysis is in Table 2-50.

Table 2-50
Aquatic species risk analysis for triclopyr

	Critical LC_{50}	Off-Site Drift		Spill in Pond	Accidental Direct Spray	
		Typical	Maximum		Typical	Maximum
	PPM	----------Q Value----------				
Triclopyr						
amine	100	<0.1	<0.1	<0.1	<0.1	<0.1
ester	0.7	<0.1	<0.1	2.57	0.23	0.69

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to any of the representative aquatic species for typical and maximum exposures to triclopyr resulting from off-site drift, or for typical or maximum exposure to amine formulations of triclopyr due to accidental direct spray to surface water. There is significant risk of adverse effects if triclopyr ester (Garlon®4) is spilled into a pond. Garlon®4 presents a risk to aquatic species if there is direct application to water such as might occur in an accidental spray. See Section 5 on Strategies for Risk Mitigation for approaches to eliminating or reducing this risk.

Other Aquatic Organisms

Kreutzweiser, Holmes and Behmer (1992) studied the effect of triclopyr ester on aquatic insects. They found no significant mortality in 10 of 12 species exposed to 80 ppm triclopyr ester. Using probit analysis, they calculated LC_{50} values that ranged from 0.6 ppm to 303 ppm for a 1-hour exposure period and a 48-hour observation period. When they applied triclopyr ester to stream channels they found no effect on insect drift at 32 ppm in three species, 3.2 ppm in one species and 0.32 ppm in the most sensitive species.

Phytotoxicity of technical triclopyr to an aquatic macrophyte, *Myriophyllum sibiricum,* was studied under laboratory conditions to determine the concentration at which 25% and 50% inhibition occurred (IC_{25} and IC_{50}) concentrations (mg a.i./L) that inhibit an endpoint parameter by 25 and 50%, respectively]. Roshon (1997 as cited by Roshon et al., 1999) used the maximum label rate for technical triclopyr of 3.84 kg/ha (expected environmental concentration of 2.56 mg a.i./L). found IC_{25} and IC_{50} values of 1.46 ppm and 4.57 ppm for shoot growth, respectively. Similar values were found for the number of roots and for root length. All those these IC values are greater than Roshon (1997) considers "expected environmental concentration", the investigator believes the literature reviewed for this report establishes that, except for spills and accidental direct applications to surfaces water, environmental concentrations will be much lower, and no adverse effects on this species should result.

Peterson et al. (1994) also determined the phytotoxicity of triclopyr to aquatic organisms when applied at the label rate of 3.84 kg/ha (expected environmental concentration of 2.560 mg/L) and reported that triclopyr at the expected environmental concentration was had low toxicity to algal species and caused <50% inhibition of growth in duckweed (*Lemna minor*). Additionally, triclopyr caused significant stimulation of growth in two algal species and reduction in one algal species.

Kruetzweiser et al. (1994) reported that the toxicity to aquatic insects also increased with increasing exposure time. The author found no significant insect mortality of insects following 3 hour exposures to the maximum test concentrations used (approximately 110 a.e. mg/L). Median lethal concentrations following 9- and 24-hour exposures were 14.9 and 4.0 mg a.e./L for *Hydropsyche* spp., and 37.0 and 9.8 mg a.e./L for *Isonychia* spp., respectively.

Gardner and Grue (1996) reported that triclopyr applied to wetlands in Washington state was not associated with significant decreases in survival or growth of bioassay organisms. The acute lethality, 96-hr EC_{50}, for *Daphnia pulex* (zooplankton) determined by Servizi et al. (1987) exposed to the butoxyethyl ester of triclopyr was 1.2 mg/L.

Peterson et al. (2001) used laboratory bioassays to determine the susceptibility of six macroinvertebrates in the Pacific Northwest. Acute toxicity was expressed as the lethal concentration to 50% (LC_{50}) and 1% (LC_1) of the test population based on a 96-hour exposure. The LC_1 values for triclopyr (1.8, 3.9, 4.0, 4.2, 29.0, and 16.1 mg/L) were used to calculate the hazardous concentration to 5% of stream macroinvertebrate community (HC_5) based on the lower 95% confidence limit ($HC_{5/95}$). The hazardous concentration for triclopyr was 0.11 mg/L.

Based on this information the investigator concludes that other than for accidents, the normal use of triclopyr on electric utility ROW should present no risk to other aquatic species.

Environmental Risk Assessment

See the wildlife and aquatic risk assessments above, and the human health risk assessment in EPRI (2003) for environmental risk associated with residues in air, plants and surface water. Following are the results of the other environmental risk assessments.

Air

Sulfometuron has a vapor pressure that is extremely low. There should be no significant vaporization from droplets or environmental surfaces, meaning that residues in air will be restricted to the distribution of droplets during the application. This will largely eliminates the environmental risk from residues in air.

Soil and Groundwater

Field and laboratory studies have established that triclopyr has relatively short persistence and limited mobility in soil, under normal circumstances. This means that the potential environmental for ground water contamination due to the use of triclopyr on electric utility ROW is limited.

Ectomycorrhizal fungi often benefit host plants through symbiotic associations with the roots of vascular plants, and are particularly important for the growth of conifer trees. Estok et al. (1989) found in lab studies using an agar media, that triclopyr significantly reduced three species of ectomycorrhizal fungus at concentrations ≥ 1000 ppm and growth was completely inhibited at concentrations ≥ 5000 ppm, but these exposure levels are more than 100 times greater than would be expected to occur in soil as a result of normal use of triclopyr. Consequently, the investigator concludes there is no environmental risk of log-term loss of soil productivity from the use of triclopyr.

PART 10. RISK ASSESSMENT FOR HERBICIDE CARRIERS, BAR OIL AND HYDRAULIC FLUID

There is a large number of petroleum based and some non-petroleum based chemicals that are involved in the management of vegetation on electric utility rights-of-way. These include several carriers for herbicides such as diesel oil, mineral oil, kerosene, and some non-petroleum based carriers based on fatty acids. They also include bar oil used in chain saws and hydraulic fluid in various types of machinery. The mammalian toxicology and human risk of these materials, including their behavior in the environment are discussed in detail in the human health risk assessment (EPRI, 2003). Most of that information is not repeated here, so EPRI (2003) should be used as a primary reference with respect to these materials.

This seemingly diverse group of chemicals shares a number of similar characteristics but they are not specific chemicals or formulations, as are pesticides, nor are they registered on the strength of a broad base of required testing. Much less attention has been given in research to their behavior in the environment and the toxicology of these materials. There is little or no published information on wildlife exposure and toxicity, even generally, much less specifically in the context of ROW management. While the environmental fate properties of a large number of individual chemicals have been studied, environmental fate information does not appear to be available on individual large hydrocarbons. Defining the fate properties of a poorly defined mixture of these larger hydrocarbons would be difficult even if the requisite information existed for the individual constituents.

The California Department of Pesticide Regulation (CPDR) has reviewed much of the unpublished reports and published literature (CDPR, 2001). A useful general reference is "Applied Toxicology of Petroleum Hydrocarbons" (McFarland et al., 1984) which discusses not only toxicology but also identity and characteristics of petroleum fractions (see King, 1984 and Lewis et al., 1984).

It is possible, with these and other sources, to arrive at a qualitative judgment of the environmental safety of carriers, bar oil and hydraulic fluid as they may be used in ROW vegetation management. The lack of specific information is not surprising because these materials are a very small part of the overall petroleum market; neither the American Petroleum Institute nor the American Chemistry Council make any reference to these products, and "bar oil" or "chain oil" are not useful terms in searching the medical literature.

While the basic approach to wildlife and environmental risk assessments remains the same, the format used in this section is somewhat different than the one used with the eight herbicides in Section 2, Parts 2-9 of this report. Given the limited information and the uneven coverage of information for the individual materials, Part 10 of Section 2

(a) briefly describes what these materials are from a chemical standpoint. A detailed explanation of them is in the human health risk assessment (EPRI, 2003).
(b) Reviews their behavior in the environment and
(c) Provides a wildlife risk assessment.

Chemical Characterization of Herbicide Carriers, Bar Oil and Hydraulic Fluid

Herbicide Carriers

While herbicides are most commonly applied in water, in some instances they are applied in other types of carrier materials, including diesel oil, kerosene and light fuel oil, mineral oil, and non-petroleum based "oils".

Diesel Oil, Kerosene and Light Fuel Oil

Kerosene, diesel oil and light fuel oils are in the middle distillate class of petroleum fraction, with carbon chain lengths from 6 to 16 or 17, and boiling range of 150 to 290 degrees C. Kerosene is hydrodesulfurized (HDS kerosene) by reaction with hydrogen at high pressures, removing impurities and much of the odor. Diesel oil falls in the same boiling range and carbon chain length as kerosene, but may be considered as not quite so "clean". Diesel oil and kerosene, like mineral oils, are mixtures of hydrocarbon compounds. Octane and the hydrocarbons found in diesel oils and kerosene are shorter in length and lighter than those in mineral oils.

Mineral Oil

The term "mineral oil" is an all-encompassing term that includes almost all hydrocarbons of geologic origin. Customarily, it means refined products, without color, odor, and with little taste,

some of which are used in foods and pharmaceuticals. These refined oils are not pure substances, however, but are mixtures of a variety of long-chain hydrocarbons (15 to 50 carbons) with properties that fall within some defined range of viscosities and boiling points, and are free of certain impurities. The mineral oil used as a herbicide carrier is usually purified by reaction with hydrogen gas at 800 psi (hydrotreated). The Food and Drug Administration allows "intermediate contact with foodstuffs", which means that mineral oil can be used in lubrication of food processing machinery.

Hamilton (1993) provides a good description of the structure and chemical and physical properties of mineral oils used as spray adjuvants.

Non-Petroleum Based Carriers

The non-petroleum carriers, or oils, are common fatty acid esters of plant or animal origin and of chain length from 16 to 22 carbons. Many are common components of most human diets. Both saturated and unsaturated fatty acids are used. Oleic acid esters are cleared for use in the food industry. Limonene, an essential oil derived from citrus, is also used as a carrier for some herbicides. Hamilton (1993) provides a good description of the structure and chemical and physical properties of vegetable oils used as spray adjuvants.

Bar Oil

Bar oil is an important material for the effective operation of chain saws. It is not "one oil", but is a diverse group of products with a wide variety of properties, designed for different service environments. There are many manufacturers of bar oils, and their various products contain a variety of additives necessary to achieve good service under the various working conditions. The precise composition is usually proprietary and unidentified, making precise environmental and wildlife risk assessment impossible.

Hydraulic Fluid

Hydraulic fluids are used in various types of mechanical equipment, such as mowing machines. Inevitably, some hydraulic fluid will be lost to the environment.

Chemically, hydraulic fluids fall into seven classes: phosphate esters, oil-in-water and water-in-oil fluids, polyalphaolefin oligomers, polyhalohydrocarbons, polyglycols, silicate esters, and silicones. In 1980, it was estimated that mineral-oil based hydraulic fluids (including fire-resistant mixtures of mineral oil and water) represented 98% of the world demand for hydraulic fluids. The base of these fluids is dewaxed crude oils, with one or more additives. The number of hydraulic fluid products marketed at present approaches 150, each with its own characteristics, preventing a precise environmental or wildlife risk assessment

The Agency for Toxic Substance and Disease Registration (ATSDR) has published a Toxicology Profile for hydraulic fluids (ATSDR, 1997). It is the product of a joint effort by USEPA, the National Toxicology Program (NTP) and ATSDR, drawing together the known information on

potential health impacts of this group of chemicals, and is the source used in preparing this brief review.

Environmental Behavior of Herbicide Carriers, Bar Oil and Hydraulic Fluid

There is very little information in the literature on the environmental behavior of bar oil and hydraulic fluid. The investigator suggests that the herbicide carriers (diesel oil, mineral oil, kerosene and light fuel oil) be used as a surrogate for these materials.

Diesel Oil, Kerosene, Mineral Oil and Light Fuel Oil

Given the chemical similarities of mineral oil, diesel oil, kerosene, and light fuel oils, the environmental behavior of these materials are combined in this section. Literature information on the environmental behavior of octane and of diesel oil and kerosene is available to a limited degree.

Diesel oil and kerosene, like mineral oils, are mixtures of hydrocarbon compounds. Octane and the hydrocarbons found in diesel oils and kerosene are shorter in length and lighter than those in mineral oils. The inferences drawn about the environmental fate properties of mineral oils relative to octane, diesel oils, and kerosene are described in Appendix A of USDA Forest Service (1997). Literature information on the environmental fate properties of octane and of diesel oil and kerosene is available to a limited degree.

Behavior in Air

Diesel oil exhibits a vapor pressure of 2.07 mm Hg at 40 C. Kerosene and light fuel oil is likely to exhibit the same degree of volatility. Specific data on residues of these materials in air in connection with right-of-way management is lacking, Owing to the lighter, more volatile components of diesel oil, mineral oil should have a lower vapor pressure and therefore be less volatile. However, all of these materials have a significantly greater vapor pressure than any of the herbicides covered in this review. Thus it should be expected that they will tend to vaporize into the air to a greater degree. Members of the public commonly note the odor of spray materials and equate this with herbicide. In fact, unless they are inhaling spray droplets, what they are more likely smelling are the carriers.

The degree to which this is en environmental risk is unknown, but given the low to moderate toxicity of this material it is likely it is more of a nuisance than a risk, perhaps except for individuals, plants or other organisms with unique (allergic) sensitivities.

Behavior in Plants

The investigator found no reports of the persistence on plants. Neither the review by Weeks et al. (1988b) on diesel oil and kerosene, or the review by Chin et al. (1989) on mineral oil cover the persistence of these petroleum products on or in plant tissues. USDA Forest Service (1989)

estimated the exposure of humans to these materials based on expected residue on plants. While they do not cite the specific residues they used in their calculations, based on the exposure data they provide it is possible to evaluate the residue levels they started with. In each case, the highest exposure values are for "leafy" vegetable and berry. USDA Forest Service (1989) assumes human consumption of 400 grams of such vegetation for a 50 kg person.

To illustrate the calculation the investigator made using the data from USDA Forest Service (1989), the dose of diesel oil is 0.00039 mg diesel oil/kg of body weight/day. For a 50 kg person, this means the amount of diesel is 0.0195 mg in 400 grams of vegetation, which is the same as 0.04875 mg diesel per kg of vegetation or about 0.05 ppm. Using data from USDA Forest Service (1989) and the same basis for calculation, the investigator estimates the concentration of kerosene and limonene in vegetation at 0.055 and 0.021 ppm respectively. These seem like reasonable numbers, and can be used to represent vegetative residues for mineral oil and light fuel oil as well.

Kroening et al. (2001) reported that clover (*Trifolium repense*) and ryegrass (*Lolium perenne*) seeds were able to germinate in the presence of volatile diesel components and after being immersed in diesel oil for 24 weeks, suggesting that properties of their seed coats prevented damage to the seeds. Adam and Duncan (2002) found that the volatile fraction of diesel fuel delayed seed emergence and reduced the percentage of germination in select plants.

Siddiqui and Adams (2002) found that inhibition of *L. perenne* was linked to the presence of low molecular mass hydrocarbon in diesel (nC$_{10}$ and nC$_{11}$). These hydrocarbons degraded rapidly in soils contaminated with up to 50 mg/g of diesel and germination inhibition was removed after a maximum of 30 days. At a very high level of diesel fuel addition (136 mg/g), germination was inhibited for more than 24 weeks (Siddiqui and Adams, 2002).

In a controlled chamber study, Pichtel and Liskanen (2001) reported that there was no detectable uptake of diesel range organic compounds from soil by grasses (*Poa*, *Phleum*, *Agrostis*) or legumes (*Pisum staivum*, *Trifolium pratense*).

Trapp et al. (2001) reported that fresh diesel applied to soils at about 1000 mg/kg showed no effect on willows, *Salix alba*, although poplar trees, *Populus nigra,* showed some sensitivity, but this rate of contamination is far greater than would normally occur in soil as a result of the use of these materials on electric utility ROW.

Phytotoxicity of No. 2 diesel fuel to *Tradescantia* plants was determined in laboratory experiments. *Tradescantia* plants were transplanted into soils contaminated with 0, 0.1, 1.0, 10,0, or 100.0 mg/kg of diesel. Results showed a high degree of correlation between the percentage of dead and withered plants 1 week following transplantation and increasing fuel concentrations. Plants in the highest diesel concentration yielded no buds or flowers until the third week, and by the fourth week surviving plants in all treatment groups appeared to begin to recover. This coincided with the decreases in diesel fuel concentrations and increased bacterial activity over time.

A diesel spill from storage tank in subalpine meadows near Mt Baker, Washington had decreased plant cover from 100% (pre-spill) to 1% within two growing seasons (Belsky, 1982). All species except *Phyllodoce empertiformis*, *Carx lenticularis*, and *Rhacomitrium sudeticum* had died. Seedlings of *C. lenticularis* began to appear on bare soil after one year, followed by seedlings of other common subalpine species two to four years later. Nine years after the spill, 5 to 10% of the ground was covered with vegetation. The concentration of diesel in the soil was not reported; so it is not possible to compare it to the levels that might result from use of diesel oil as a herbicide carrier.

Behavior in Soil and Ground Water

In an agricultural field study, Haigh (1995) found that two highly refined mineral oils used for motorcycle and automotive engines underwent limited degradation in soil: residues decreased from 90 to 70 mg/g soil residues concentration within a 12 month period.

Angehrn et al. (1999) reported that residual contaminants, designated as total solvent extractable material, remain in soils contaminated with mineral oil and that 1 year after bioremediation, the major portion (93%) of the residual contaminants can be recovered from the top soil and only 7% was lost. The majority of the total loss (>98%) was due to transformation processes (biodegradation and aging effects), while small amounts escaped to the atmosphere (0.08%), were taken up by plants (<0.001%), or were leached (1.7%).

Octane adsorbs readily to soils, with a distribution coefficient (K_d) of 110 L/g (calculated by Lyman et al., 1982). Because of the larger carbon structures and lower water solubilities, the components of mineral oil will be more likely to adsorb to soils than octane. Similar behavior is expected for diesel oil, kerosene and light fuel oil.

The biodegradation constant of octane is approximately 0.11 day_{-1} (means 11% of what is present biodegrades each day) (Ladd, 1956). Because of the lower relative water solubility of mineral oils (chemicals must be dissolved in water to be consumed by microorganisms), the greater relative soil adsorption of mineral oils (chemicals adsorbed to soil are unavailable for biodegradation and are less likely to partition to water), and the larger molecular size and weight, mineral oil components would be expected to have a biodegradation rate much lower than octane. Weeks et al. (1988b) evaluated both kerosene and diesel oil for USDA Forest Service. They concluded that the aromatics (such as benzene) that are components of diesel undergo degradation in soil at a slower rate than the aliphatics (such as octane). They estimate the half-life of benzene to be less than 1 month. Except in the case of a spill of a large volume of diesel, Weeks et al. (1988b) felt there was little concern with persistence or leaching in forest soils. USDA Forest Service (1989) summarized the leaching characteristics of a series of herbicides and also light fuel oil. They reported a soil half-life of 6 days and a non-significant rate of leaching. USDA Forest Service (1989) concludes that even under treated sites, there will be no measurable residues of light fuel oil in ground water. Given these limited data, the investigator concludes that the likelihood of leaching to ground water is extremely low.

Diesel oil undergoes bacterial degradation and volatilization in soils. Eriksson et al. (1998) reported that in a laboratory test, degradation of diesel fuel was complete after 3 weeks, when the

only remaining substances were decahydronaphthalenes. In a bioremediation study, Margesin and Schinner (1997) reported that 60-65% of residual diesel oil in contaminated alpine soils were degraded by microbial activity while the about 30% was eliminated by abiotic processes. Petroleum hydrocarbons in extractable fraction decrease rapidly (half-lives ranged from 11 to 26 days depending on extraction method) in a heavy clay soil contaminated with diesel fuel at 5000 mg/kg demonstrating natural attenuation by soil bacteria.

Kroening et al. (2001) reported that as much as 58% of diesel oil in soils were volatilized over a 360 day study period. Loeser et al. (1998) reported that under aerobic-anaerobic conditions in a laboratory experiment, less than 50% of diesel was degraded by microbes after 650 hours because the hydrocarbons were adsorbed to the soil. Degradation under aerobic conditions was similar (Loeser et al., 1998). In soil leachate analyses using two different soils, high-molecular-weight polycyclic aromatic carbons from diesel fuel were not detected. Low concentrations of 1-methylnaphthalene, 2-methylnaphthalene, fluorene, and phenanthrene detected in the soil leachate were similar to levels detected in groundwater taken from wells in a diesel contaminated area in a rail yard. The data indicated that there would be limited potential for future migration of low-molecular-weight polycyclic aromatic carbons from soils to groundwater at the contaminated area.

Environmental conditions and soil characteristics can affect the retention and volatilization of kerosene in soils. The volatility of kerosene is related to the vapor pressure of each of its components as well as environmental factors at the soil-air-water interface (Jarsjö et al., 1994). In a laboratory study, temperature had little effect on kerosene retention (Jarsjö et al., 1994). The cumulative volatilization of kerosene was 2–3 times higher at 27ºC than at 5ºC. Organic matter affected selective volatilization of the kerosene components but clay content was less influential.

Jarsjö et al. (1994) reported that in laboratory experiments, a simple linear relationship was found between water retention capacity and kerosene retention capacity. The kerosene retention capacity was lower than the water retention capacity. Additionally, the combined effects of soil porosity and soil moisture content on the kerosene retention capacity were significant and could be quantified with a linear relationship. Low kerosene retention values were found for relatively high water contents, which may be a result of water blocking the pore entrances and preventing the kerosene from filling up available pore space. Yaron et al. (1989) reported that soil moisture content was the most influential parameter in the adsorption-desorption process of vapor hydrocarbons from a synthetic “kerosene” source on different soils. The vapor adsorption and desorption of petroleum product components on soils and from soils occurs differentially as a function of each component’s properties, the soil composition, and soil moisture status (Yaron et al., 1989).

Galin et al. (1990) studied the stability of kerosene in soils as affected by volatilization in a laboratory column experiment. The authors determine losses in total concentration, and the change in composition of kerosene residuals in dune sand, a loamy sand, and a silty loam over a 50-day period. The differential volatilization of seven major kerosene components (according to their carbon number) resulted in an altered composition of kerosene residues.

Galin et al. (1990) felt that, as a general pattern, the relative concentration of hydrocarbons with higher carbon numbers increased as the concentration of hydrocarbons with smaller carbon numbers decreased. The rate of volatilization of the compounds was also affected by soil properties. The silty loam sand, characterized by small soil pore size, had the greatest amount of kerosene remaining at the end of the study, whereas the dune sand with large pores had least amount of kerosene remaining as a result of greater volatilization of low-molecular-weight compounds. Dror et al. (2001) found that the main processes controlling attenuation of kerosene in soils were volatilization and redistribution with soil depth. The effect of transport of kerosene was minor in relation to its volatilization since the more soluble and, therefore more mobile, fractions were volatilized shortly after application (Dror et al., 2001).

Based on this review the investigator concludes that with a summarized the leaching characteristics of a series of herbicides and also light fuel oil. They reported a soil half-life of 6 days and a non-significant rate of leaching (USDA Forest Service, 1989) that even under treated sites, there will be no measurable residues of light fuel oil in ground water. Others have corroborated that there would be limited potential for migration of low-molecular-weight polycyclic aromatic carbons from soils to groundwater, even at the contaminated site that was included in that particular study.

Behavior in Water

The mechanisms for entry to surface water are quite variable given this diversity of materials and how they are used. The herbicide carriers, including diesel oil, light fuel oil, kerosene, and non-petroleum based carriers can enter due to drift or accidental direct application to the water surface. This seems highly unlikely, though, because these carriers are not used in broadcast foliar applications but for penetration of bark on stems and stumps. These applications should be well controlled by the applicator.

Runoff is the other primary source of possible entry to surface water, but given the high degree of cover (duff) on the soil and the permeability of most areas on rights-of-way (except for areas of compacted soil), this mechanism of entry is likely to introduce only very small amounts of material in surface water. USDA Forest Service (1989) calculated the potential for runoff of benzene, the aromatic and more water soluble component of kerosene. The models they used included rainfalls of up to 7 inches and slopes up to 45%. Even with no buffer between the operational area and the surface water, they calculate no detectable residues will occur. The investigator did not find any specific reports of carriers or related materials in surface water in connection with right-of-way vegetation management programs.

Ward and Brock (1976) reported that a small percentage of heterotrophic bacterial populations in Wisconsin lakes could metabolize hydrocarbons including mineral oil. A 20 hour lag phase was the rate of oxidation varied with Temperature, which is the major variable affecting oxidation rates.

Cooney, et al. (1985) reported that microbial degradation of weathered kerosene contaminating three freshwater lakes in Ohio was limited by seasonally low water temperatures and phosphorous and nitrogen availability.

In Pennsylvania, the effects of a pipeline spill of aviation kerosene into a stream were monitored for two years (Guiney et al., 1985a; 1985b). Guiney et al. (1985b) reported that the concentration of kerosene-range hydrocarbons and total organic carbons in a contaminated stream decreased during the initial month after the spill and remained at or below background levels after 3 months. Elevated concentrations of hydrocarbons were detected in surface sediment samples and tissues of fish collected at two locations (primary boom recovery sites during spill cleanup) up to 14 months after the spill occurred. No kerosene-range hydrocarbons were detected in the surface sediments after 21.

USDA Forest Service (1989) calculated possible levels of water contamination in connection with forestry operations, including rights-of-way management, in the southeastern US. They report exposure values for 50 kg humans based on consumption of one liter of water. Using the same approach as outlined above for plants, the investigator calculated the typical and maximum residue levels in surface water based on the human exposure data in USDA Forest Service (1989) as follows: diesel oil - 0.0025 ppm typical, 0.021 ppm maximum; kerosene – 0.003 ppm typical, 0.008 ppm maximum; limonene – 0.001 ppm typical, 0.0065 ppm maximum. The investigator concludes that these calculated values provide an index to the concentration of all these materials in surface waters, and uses them in the wildlife and aquatic species risk assessments.

Wildlife Risk Analysis

The primary concern of this analysis is with kerosene and diesel oil. Mineral oil is of interest but given its relative purity and approval as a lubricant in food processing machinery, the investigator concludes that it would not be of any greater environmental or wildlife risk than diesel oil or kerosene, and is most likely of much less risk. The non-petroleum carriers, or oils, are plant- or animal-origin fatty acid esters. Many are common components of most human diets, and therefore offer risk no greater than that offered by mineral oil, in the opinion of the investigator. As is the case for the environmental behavior section, the wildlife risk analysis is dominated by data for herbicide carriers, which can provide a reasonable surrogate for bar oil and hydraulic fluids. This risk assessment relies heavily on the hazard analysis for light fuel oils (diesel and kerosene) in USDA Forest Service (1989 and1997) and Weeks et al. (1988b).

Avian and Terrestrial Species

Toxic effects of mineral oils in animals are reviewed by Bingham et al. (1980), Chircova (1982), and World Health Organization (1982) as cited in IARC (1984). Based on acute oral LD_{50}'s of greater than 25 ml/kg in rats mineral oils can be classified as very slightly toxic (Maxwell, 1982, as cited in Walstad and Dost, 1984). No toxic effects were observed in rats receiving mineral oil-class 5 at an oral (gavage) dose of 2 ml/kg three times weekly for three months (Kimborough et al., 1980 as cited in IARC, 1984). Beck et al. (1983, as cited in Kane et al., 1984) showed that solvent-refined naphthenic oil-class 3 has a low order of acute toxicity. This oil has an oral LD_{50} value of greater than 25 ml/kg.

Weeks et al. (1988b) summarized the toxicology literature on diesel oil and kerosene in the context of their use as herbicide carriers. They conclude that diesel oil is slightly toxic to orally exposed birds, with LD_{50} values of more than 16,400 mg/kg (roughly equivalent to 20 ml of diesel oil per kg body weight). However, small quantities of diesel oil do have adverse effects on bird-egg hatching success rate. For instance, even small quantities (such as 1 to 5 microliters of diesel oil adversely affected hatchability of duck and pheasant eggs. Kerosene was not lethal to bird embryos when applied at rates of up to 50 microliters to duck eggs. This lack of effect is attributed to the low aromatic content of kerosene compared to diesel oil.

According to Weeks et al. (1998b), diesel and kerosene are very slightly toxic to mammals. The LD_{50} in rats is 7380 mg/kg for diesel, and the lowest oral lethal dose is 28,000 mg/kg for kerosene.

USDA Forest Service (1989) summarized the toxicity characteristics of limonene, which is used as a non-petroleum based carrier in some cases. They reported LD_{50} values of more than 5000 mg/kg in rats and more than 2000 mg/kg in rabbits. USEPA has approved limonene for control of ticks and fleas on dogs and cats, indicating a low order of toxicity to these mammals.

Toxic effects of mineral oils in animals were reviewed by Bingham et al. (1980), Chircova (1982), and World Health Organization (1982) as cited in IARC (1984). Beck et al. (1983, as cited in Kane et al., 1984) showed that solvent-refined naphthenic oil-class 3 has a low order of acute toxicity. This oil has an oral LD_{50} value of greater than 25 ml/kg and dermal LD_{50} value of greater than 5 g/kg.

Mineral oil has been used to decrease pest bird population by preventing embryos in sprayed eggs to develop (e.g., Bedard et al., 1992). White mineral oil has been shown to kill embryos in incubated chicken eggs from developing and hatching. Pochop et al. (1998) reported that oiling incubated chicken eggs (either 5 or 16 days after incubation began) with white mineral oil killed all embryos.

The wildlife risk analysis for mineral oil is in Table 2-51

Table 2-51
Wildlife risk analysis for mineral oil

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds	43	1050	4200	21,000	rabbit
Mammals	76	2250	4200	21,000	rabbit
Amphibians	43	1054	4200	21,000	rabbit
Reptiles	58	1347	4200	21,000	rabbit

[1] The common laboratory test species used to represent the group of wildlife species.

The wildlife risk analysis for diesel oil is in Table 2-52

Table 2-52
Wildlife risk analysis for diesel oil (adapted from USDA, 1989)

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds (eastern bluebird)	18	163	3280	16,400	mallard
Mammals (red bat)	32	292	1476	7,380	rat
Amphibians	27	243	3280	16,400	mallard
Reptiles (hognose snake)	36	314	3280	16,400	mallard

[1] The common laboratory test species used to represent the group of wildlife species.

The wildlife risk analysis for kerosene is in Table 2-53

Table 2-53
Wildlife risk analysis for kerosene (adapted from USDA, 1989)

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds (eastern bluebird)	21	211	3280	16,400	mallard
Mammals (red bat)	36	378	5,600	28,000	rat
Amphibians	31	316	3280	16,400	mallard
Reptiles (hognose snake)	40	408	3280	16,400	mallard

[1] The common laboratory test species used to represent the group of wildlife species.

The wildlife risk analysis for limonene is in Table 2-54

Table 2-54
Wildlife risk analysis for limonene (adapted from USDA, 1989)

Species Group	Dose Estimate		Critical Toxicity Values		Laboratory Represent. Species[1]
	Typical	Maximum	1/5 LD_{50}	LD_{50}	
	mg/kg	mg/kg	mg/kg	mg/kg	
Birds (eastern bluebird)	8	159	---	---	Not available
Mammals (red bat)	14	288	1000	5000	rat
Amphibians	5	102	___	___	Not available
Reptiles (hognose snake)	6	131	___	___	Not available

[1] The common laboratory test species used to represent the group of wildlife species.

There is adequate toxicity information for diesel oil and mineral oil. The risk analysis shows that none of the wildlife species in the risk analysis tables experience a level of exposure (maximum exposure) that is greater than the no effect level (NOEL). The margin of safety is greater when typical exposure values are considered.

Data on the toxicity of limonene is limited to the rat. In this case there is a large margin of safety between maximum exposure and the no effect level. Assuming the other species groups have a

toxicity proportional to their toxicity to diesel oil, mineral oil and kerosene, limonene should present no toxic risk to these species either.

Aquatic Species

There is no specific data available on the toxicity of mineral oil to aquatic species, representing a gap in knowledge. Chin et al. (1989) conducted a review of the toxic properties and environmental fate characteristics of mineral oils for the US Department of Agriculture. Their report included toxicity information for mammalian species and birds, but they did not include data for aquatics. The inherent toxicity of mineral oil to mammals is lower than that of diesel oil and kerosene. It is a reasonable assumption that this would be the case for aquatics as well, hence the data on aquatic toxicity for diesel and fuel oil is used as a surrogate. The investigator believes that this is a conservative approach and that it will provide a greater margin of safety than is apparent from this analysis.

Weeks et al. (1988a) prepared a background statement on kerosene and diesel oil for the U. S. Dept. Agriculture. Following is the information they included in their report.

> Fish
>
> Diesel fuel, jet fuels, and fuel oils are moderately to highly toxic to fish (based on the toxicity categories of USEPA (1985). Jenkins et al. (1977, as cited in Burks, 1982) studied the acute and chronic toxicity of jet fuels to several fish species. They reported 96-hour LC_{50}'s (static tests) for the golden shiner (*Notemigonus crysoleucas*) of 0.68 and 0.94 mg/l for the jet fuels RJ-4 (a 12-carbon molecule) and RJ-5 (a 14-carbon molecule), respectively. They also reported a 97-day nonlethal concentration for rainbow trout (*Salmo gairdneri*) of less than 0.03 mg/l for RJ-4 and 0.04 mg/l for RJ-5; and a no-effect level for eggs of the flagfish (*Jordanella floridae*) exposed by continuous-flow to RJ-4 of 0.2 mg/l. Reduced hatchability was observed in flagfish eggs from exposure to RJ-5 at concentrations above 0.05 mg/l.
>
> Acute toxicity tests with freshwater fish showing 96-hour LC_{50}'s of greater than 0.19 mg/l for diesel fuel and greater than 1.2 mg/l for No. 2 fuel oil have been reported by USEPA (1976, as cited in DOE, 1983). Tagatz (1961, as cited in Burks, 1982) reported much lower toxicity, with a 48-hour LC_{50} for No. 2 fuel oil of 125 to 251 mg/l with juvenile American shad. His reported LC_{50} is based upon the amount of oil applied to the waters' surface (nominal concentration) and not the water soluble fraction; this may account for the apparent lower sensitivity of the shad.
>
> Irwin (1964, as cited in Burks, 1982) calculated a "ration of resistance" to allow the ranking of the sensitivities of fifty-seven fish species to oil refinery wastewater. The guppy (*Lebistes reticulatus*) was least sensitive and was assigned a ratio of resistance of 100. The ratios of resistance for some of the common freshwater fish were as follows: rainbow trout (*Salmo gairdneri*), 34.68; smallmouth bass (*Micropterus dolomieui*), 35.60; northern pike (Esox lucius), 37.31; fathead minnow (*Pimephales promelas*), 49.19; largemouth bass (*Micropterus salmoides*), 53.27; bluegill (*Lepomis macrochirus*), 54.10; and channel catfish (*Ictalurus punctatus*), 60.15. This study may be useful in predicting

the relative order of sensitivities of these species to diesel fuels and other petroleum products.

Aquatic Invertebrates

The 96-hour LC_{50} for adult blue crabs (*Callinectes sapidus*) exposed to No. 2 fuel oil was 14.l mg/l. No histopathological changes were observed in the gills, hepatopancreas, or muscles of the blue crab after 2 weeks of exposure to No. 2 fuel oil at 0 to 1.0 ppm (Melzian, 1983).

A spill of No. 2 fuel oil into a small stream in Virginia was acutely toxic to some fish, crayfish, and caddis flies. At 2 weeks after the spill the density of benthic macro invertebrates downstream was 25 percent less than the density upstream from the spill, but species diversity was not affected. The density of the macro invertebrates had returned to normal levels by 18 weeks after the spill (Hoehn et al., 1974 as cited in Burks, 1982).

Bass et al. (1987) assessed the impacts a pipeline spill of kerosene had on aquatic communities. After approximately 120 m^3 of kerosene were spilled into Bull Run Creek in Virginia, the authors did not find any clear evidence of long-term adverse effects to the periphyton and macrobenthic communities.

Breteler et al. (1988) reported that acute toxicity tests using saltwater mysid shrimp (*Mysidopsis bahia*) exposed to drilling muds with mineral oil had LC_{50} values of 3,090 mg/L. Mineral oil was not found to be lethal to other saltwater organisms when exposed to drilling mud solids in a ten-day solid phase toxicity test containing 5% mineral oil: sand worm (*Nereis viren*) survival 97 %; grass shrimp (*Palaemonetes pugio*) survival 98%; and soft-shell clam (*Mya arenaria*) survival 95%. Saha (1983) reported 96-hr LC_{50} for the freshwater plankton species, *Diaptomus forbesi* as 175 ppm. Kerosene had low toxicity to the freshwater worm species, *Branchiura sowerbyi* (96-hr LC_{95} = 2,000 ppm). Panigrahi and Konar (1989) reported 96-hr LC 50 for the zooplankton species, *Cyclops viridis* to be 1525 mg/L; for the aquatic mollusk species *Thiara tuberculata* the value was 1340 mg/L; and for chironomid larvae the value was 1370 mg/L.

Saha (1983) reported that the fish *Tilapia mossombica* tolerated 10,000 ppm of kerosene for 96 hours. Saha (1983) reported 96-hr LC_{50} for the freshwater plankton species, *Diaptomus forbesi* as <1.0 ppm (the 96-hr LC_{95} = 86 ppm). Diesel oil had low toxicity to the freshwater worm species, *Branchiura sowerbyi* (96-hr LC_{95} = 80,000 ppm).

Bass et al. (1987) assessed the impacts a pipeline spill of No. 2 fuel oil had on aquatic communities. Approximately 340 m^3 of fuel oil was spilled into Mine Run Creek in Virginia. The fuel oil reduced the standing crop, density, and taxa number of the periphyton and macrobenthic organisms. Among the macrobenthos, the collector-filter group was reduced while the collector-gathers and scrapers were more abundant at the affected sites.

Aquatic Risk Analysis

To estimate the risk of adverse effects occurring, the selected toxicity values were compared to the typical and maximum estimated environmental concentrations (I-5). The ratio of the EEC to the LC_{50} is called the quotient value (Q value). The Q values were compared to the risk criteria proposed by USEPA where the risks of adverse effects to fish or invertebrates are estimated as follows:

Table 2-55
Definition of "Q" values

	Q VALUE	RISK
EEC/LC_{50}	0.1	No acute risk
EEC/LC_{50}	>0.1 - <0.5	Presumption of risk that may be mitigated
EEC/LC_{50}	>0.5	Presumption of significant risk of acute effects
EEC < NOEL or MATC		No chronic risk

Table 2-56
Aquatic species risk analysis for herbicide carriers in surface water Adapted from USDA (1989)

Herbicide	Critical LC_{50}	Off-Site Drift		Spill in Pond	Accidental Direct Spray	
		Typical	Maximum		Typical	Maximum
	PPM ----------Q Value----------					
Diesel oil	> 0.19	<0.1	<0.1	<0.1	<0.1	<0.1
Kerosene	> 0.19	<0.1	<0.1	<0.1	<0.1	<0.1
Limonene	5.2	<0.1	<0.1	<0.1	<0.1	<0.1
Mineral Oil	0.21	<0.1	0.16	15.2	7.0	32.6

The results of the risk analysis indicate that there is no significant risk of acute adverse effects to any of the representative aquatic species for typical and maximum exposures resulting from off-site drift, except for mineral oil at maximum drift. All Q values are less than 0.1, except mineral oil with a Q of 0.16. Mineral oil is used only in the low volume basal applications, where drift is not a factor, thus no mitigation is required.

There is significant risk of adverse effects if mineral oil or triclopyr ester (Garlon®4) is spilled into a pond, with Q values of 17.4 and 2.57 respectively. However, there are no ponds on these rights-of-way so this risk is not present and no mitigation is required.

Mineral oil presents a risk to aquatic species if there is a spill or direct application to water such as might occur in an accidental spray. Strategies for mitigation are in Section 4.

3
EXTRAPOLATION ACROSS THE US

The movement, persistence and ultimate fate of chemicals in air, soil, plants, and water is influenced by many environmental as well as chemical factors. While the properties of the individual chemicals are fixed and do not change, the properties of the environment within which they are applied can be quit different. For instance, the summer temperature of the soil in the coastal plain of Georgia is likely to be higher than it is in the Cascade Mountains of Washington. Similarly, the soil moisture content, and the physical and chemical properties of the soils in these two areas are likely to be different. As a result of these differences the behavior of a given chemical in these two settings are also likely to be different.

The problem is how to extend the findings in this report to a particular geographic location or set of environmental variables. Some complex computer models for this have been developed for herbicide-soil behavior. They include the PRZM (Pesticide Root Zone Model) and LEACH (Leaching Evaluation of Agricultural Chemicals) both used by USEPA as part of the pesticide registration process. These models were used in USDA Forest Service (1989), for example, to make site specific estimates of the behavior of several herbicides and carriers in the southeastern US. Similar models for behavior in plants, air and water are lacking.

The PRZM and LEACH models could be used for some major soil types and ecoregions of the US, but it would be such a large undertaking as to be well beyond the scope of this risk assessment. It is the opinion of the investigator that the results of such modeling exercises can be misleading because they may give an impression of a greater degree of precision and accuracy than can be attained. In addition, the lack of models for herbicide behavior in plants, air and water would prevent a comprehensive approach to this problem.

The general effects of many environmental parameters are known based on field and laboratory studies, as follows:

- The rate of microbial decomposition of a given pesticide is proportional to the size and metabolic rate of the microbial community at a given location. In general, this means that in moist, warm environments where there is adequate organic matter, the rate of decomposition of a specific herbicide will be faster. Conversely, in a dry cold environment, the rate of decomposition will be slower and the herbicide will persist for a longer period of time.
- For a given amount of precipitation, herbicide leaching is greater in soils of high porosity and low organic matter content. In general, this means that in areas where there is abundant rainfall, and the soils are coarse and contain little organic matter, herbicide leaching will be greater. If, on the other hand, leaching pressure is low, as it is in more arid areas, the difference in leaching between a sandy soil and a loam may be small.

- Decomposition in warmer water will be faster than in colder water, but is also dependent on the rate of microbial activity, since this is the most common mechanism by which herbicide decomposition occurs. This means that for a given level of microbial activity, herbicide persistence in surface water in the northeast will likely be longer than in the southern states. Although as a practical matter, the rate of decomposition in water is probably much less important than other mechanisms of dissipation (such as dilution due to mixing in the water column or streamflow). This makes differences in persistence relatively unimportant, especially given the very limited amount of herbicide that enters surface waters from ROW vegetation management operations.

In this report, an effort was made to include as much specific field data as possible on the chemical movement and persistence of each chemical and, where appropriate, individual readers can take such site specific information and apply it to their own circumstances. In addition, all of the data from field and laboratory studies are included in the pesticide registration evaluations done by USEPA, hence the registration already incorporates the best knowledge that exists about such topics, and is reflected in the directions and limitations imposed by the specific pesticide label .

Given these perspectives, it is the opinion of the investigator that herbicide use consistent with the label and best management practices will make regional variations in herbicide behavior unimportant with respect to environmental and wildlife life risk. This means that the risk assessments in this report are applicable across the US, without consideration for regional environmental variables, except in the most extreme cases (e.g., the Arizona desert, during the summer).

4
SUMMARY OF ENVIRONMENTAL AND WILDLIFE RISK AND RISK MANAGEMENT

In this chapter, the environmental risk in air, plants, soil and ground water and surface water is summarized, with specific attention given to identification of instances where risk is greater and the application of risk management may be needed. The same is done for wildlife and aquatic species. The emphasis in this summary is on the herbicides covered by this report, because of the limited information available on the other chemicals.

Air

Chemicals can be in the air as droplets or due to volatilization. Droplets move with the air mass (drift), but larger (heavier) droplets will settle out faster. Differences in droplet size are a function of application parameters, such as nozzle size, pressure and height above the ground. None of these factors are unique to any given herbicide, but relate to chemical application in general. Adherence to label instructions and well established best management practices will minimize the production of overly fine spray particles and reduce the potential for damage due to herbicide spray drift.

Volatilization (either from droplets as they move in the air, or from surfaces (such as plants or soil) is largely a function of vapor pressure and environmental temperature. Following is a list of the herbicides covered by this report in decreasing order of their volatility, based on their vapor pressure. The difference in vapor pressure between some chemicals is small, making the precise position of any given chemical in the list less important than its general location:

2,4-D ester
Triclopyr ester
Fosamine Ammonium
Picloram
Metsulfuron Methyl
Imazapyr
Glyphosate
Sulfometuron Methyl

None are of such high volatility that entry to air via volatilization poses significant environmental risk.

The investigator concludes that behavior of herbicides (and by extension - the other chemicals covered by this report) will not present an environmental risk, if herbicide label instructions and best management practices are followed.

Plants

Residue level and persistence in plant tissue is the key element of environmental risk because this is the route through which the chemicals covered by this report may be ingested by animals, including humans. USDA Forest Service (1989) used *k* values from a variety of sources, as an index to herbicide persistence in plant tissue. The degradation rate *k* is defined as the fraction of remaining herbicide residue that is degraded per day. Following, is a listing of some of the herbicides covered by this report. They are listed in increasing order of persistence,b with their associated *k* values:

Herbicide	*k*
Sulfometuron Methyl	>0.347
Picloram	0.0693
Glyphosate	0.0495
2,4-D ester	0.0431
Imazapyr	0.0266
Triclopyr ester	0.004

Note that USDA Forest Service (1989) does not provide values for fosamine ammonium or metsulfuron methyl. Based on the expected half-life in plants of 10 days for fosamine ammonium, it would be about the same persistence as picloram. Data on persistence in plants is lacking for metsulfuron methyl.

The risks posed by residues in plants are assessed for humans in the human health risk assessment (EPRI, 2003) and in the individual sections of this report for wildlife, and are summarized below.

Soil and Ground Water

The most important element of behavior in soil, is the potential for movement to ground water. This element is a function of both persistence and mobility in the soil and their inactions. For example, a chemical such as fosamine ammonium, which is highly water soluble but very short in persistence, will show little tendency to leach to ground water. On the other hand, metsulfuron methyl, which is fairly short in persistence, has sufficient water solubility and limited tendency to bind to organic matter making it more mobile in soil. Based on consideration of these various factors and their interactions, following is a subjective ranking by the investigator of potential for ground water contamination. The compounds are listed in increasing order of leaching potential,

but the difference between some chemicals is small, meaning the general location in the list is more important.

Glyphosate
Imazapyr
Fosamine Ammonium
2,4-D
Triclopyr
Metsulfuron Methyl
Picloram

The investigator emphasizes that, with the possible exception of picloram, in some instances, that ground water contamination will be rare, largely eliminating the environmental risk from these chemicals in soil.

Surface Water

The entry to surface water is primarily a function of the location of the treated area relative to the water and the type of application involved. These are independent of the specific chemical, although data is included in this report on residue levels that have been observed in surface waters in some cases. Some important advances have been made in predicting off-site deposition of spray materials, as they are used in forestry (Teske, Thistle and Eav, 1998; Teske, Ice and Thistle 2002; Teske and Ice 2002; Teske, Thistle and Ice, 2003). While these emphasize aerial application, some work has also been done on ground applications, but in the opinion of the investigator, additional develop of such models is needed for them to be of significant use in ROW settings. The investigator believes this technique (i.e., modeling) can be very useful in planning vegetation management on ROW and should be developed for this purpose. The environmental risks from residues in water are reflected in the human health risk analysis (EPRI, 2003) and in the wildlife and aquatic species risk analysis in this report.

Wildlife Risk Analysis

The human health risk analysis used the margin of safety as an index to risk (EPRI, 2003). A similar approach can be used for wildlife risk. This is done by calculating the ratio of 1/5 LD_{50} to the typical and maximum exposure level to yield the margin of safety. The data to do this are from the wildlife risk analyses in this report. For aquatic species, the "Q" values in the aquatic species risk analyses are a similar ratio, i.e., the critical toxicity value compared to the exposure level.

The results of the risk analyses for both wildlife and aquatic species are summarized in Table 4-1. For each herbicide, it includes the smallest margin of safety found in this analysis for any wildlife species group and any "Q" values that are greater than 0.1, which is the no-acute-risk level. A margin of safety (MOS) of more than 1 is believed to provide an adequate margin of safety at the population level for wildlife species.

Table 4-1
Margins of safety (MOS) for wildlife species and "Q" values for aquatic species

Herbicide	MOS for Wildlife for Typical Application Rate	MOS for Wildlife for Maximum Application Rate	"Q" Value for Aquatics
2,4-D	2	0.2	1 for spill in pond, all others are < 0.1
Fosamine Ammonium	59	2.9	All < 0.1
Glyphosate	24	1.7	All < 0.1
Imazapyr	500	27	All < 0.1
Metsulfuron Methyl	836	50	All < 0.1
Picloram	48	3	All < 0.1
Sulfometuron Methyl	625	47	All < 0.1
Triclopyr, amine	7.75	0.4	All < 0.1
Triclopyr, ester	7.75	0.4	2.57 spill in pond, 0.23 and 0.69 typical and maximum accidental direct spray to surface water

The results of this summary show that the risk to wildlife is within the acceptable range in all cases, under typical application. Under maximum application rates, the summary shows that fosamine ammonium, glyphosate, imazapyr metsulfuron methyl picloram and sulfometuron methyl have an adequate margin of safety for wildlife species at either typical or maximum rates of application. 2,4-D and triclopyr amine and ester also have an adequate margin of safety at typical rates of application, but they do not at maximum rates of application.

For aquatic species, all herbicides provide an adequate margin of safety for under typical or maximum rates of application. 2,4-D also has an adequate margin of safety involving accidental direct application to surface water, but it does not in the case of a spill into a pond. Triclopyr ester does not have an adequate margin of safety for a spill in a pond or for a typical or maximum rate of direct application to surface water.

Risk Management

Risk management is the strategy and suite of tactics used to mitigate those instances where the margin of safety is smaller or the environmental risk is greater than is desired. The analyses of most of the chemicals in this report show levels of risk that should be considered acceptable. In a

few instances, this is not the case. These are identified below along with suggestions for mitigation.

Environmental Risk

The environmental risk of greatest importance is the potential for leaching to ground water. It is the opinion of this investigator that only picloram deserves special attention in this regard. A two-step strategy seems most prudent, as follows:

1. Select another chemical, if the vegetation control objective can still be attained.
2. If an efficacious alternative chemical is not available, then avoid application of picloram in areas with porous, low organic matter soils and shallow water tables.

The label provides some direction on this point, but the investigator suggests adoption of this two step strategy.

Wildlife and Aquatic Species Risk

Only the use of 2,4-D and triclopyr result in wildlife and aquatic risk that do not meet normally accepted guidelines.

For 2,4-D and triclopyr, the tactic for mitigation of wildlife risk is to avoid broadcast application of high rates of this chemical. More directed and limited application (selective foliar, basal, cut surface, etc.), especially at lower rates of application, should provide a level of risk to wildlife that is acceptable.

The unacceptable aquatic risk for 2,4-D and triclopyr ester results from the accidental spill of herbicide into a pond. Careful transportation, storage and handling of concentrates at the work site, and the selection of areas away from ponds for mixing and loading of chemical are obvious tactics for mitigation.

In addition, accidental direct application of triclopyr ester to surface water, even at typical rates of application, produces a level of aquatic risk that is not acceptable and must be avoided. Prespray surveys to determine the location of water bodies, including small streams, combined with marking them as necessary, and effective supervision of field crews are a set of tactics that can mitigate this risk.

The human health risk assessment (EPRI, 2003) provides important suggestions for personnel training and supervision and for project planning that are also important in risk management and mitigation. These are not repeated here, but the reader is encouraged to review them in EPRI (2003).

5
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A
GLOSSARY OF TERMS

The following is a glossary focused on this report. It has some overlap with the glossary in the human health risk assessment (EPRI 2003), which should be consulted if terms are found in the text of this report that are not included in this glossary.

Acute toxicity (Short term toxicity)-Acute toxicity is the quality or potential of a substance to cause injury or illness from a single dose or short period of exposure. (See subacute, subchronic and chronic).

Adjuvant-Any additive to a pesticide formulation that is not active itself, but is intended make the active ingredient work better.

Adverse-effect level (AEL)-Signs of toxicity that are not accompanied by grossly observable signs. Such symptoms must be detected by invasive methods, external monitoring devices or prolonged systematic observations. (See NOAEL).

Biodegradable-Capable of being metabolized by a biological process or organism.

Chronic toxicity-(Long-term toxicity)-Chronic toxicity is the quality or potential of a substance to cause injury or illness after repeated exposure for a long period of time. Chronic toxicity tests run for a year or more; for rodents the period may extend through the entire life span. A chronic effect persists for months or years and may arise from acute or long-term exposure. (See acute, subacute, and subchronic).

Contaminant-In a formulation, usually residues or impurities from the manufacturing process present in small quantities. Contaminants must be identified to the regulatory agency, which judges whether they are of concern.

Degradation-Breakdown of a compound by physical, chemical or biochemical processes into basic components with properties different from those of the original compound.

Detection limit-The lowest concentration of a chemical that can be identified in a substance (e.g., soil, foliage or body fluids). Analytical sensitivity varies among chemicals, and in different media. The detection limit is usually lower than the level that can be reliably measured. For example, it may be possible to find a substance present at 0.01 parts per billion, but only at levels above 0.03 ppb can the amount be stated.

Dose-The amount of a chemical that actually enters the body to be distributed to all of the organs and cells. Distribution to tissues and cells is selective, and depends on the nature of the chemical and characteristics of each kind of cell.

Dose-response relationship-The central idea in toxicology and in pharmacology (which is the science dealing with beneficial effects of therapeutic drugs). As the dose (or concentration) of a chemical increases, the effect increases, and as the dose is lowered, the effect becomes less. This response pattern applies to every interaction between a chemical and a biological system, whether human, fish, bacteria or any other kind of organism or tissue. The dose-response relationship is absolutely essential to judgment of the effect of any chemical.

EC_{50}-Acronym for median effective concentration.

Environmental chemistry-The study of the physical, chemical and biological processes that govern behavior and fate of a chemical such as a pesticide after it is used.

Exposure-Amount of a chemical that reaches a surface from which it might be absorbed. The dose is some fraction of the exposure. Exposure does not include material that is on nearby foliage or other surfaces. It is only the material that reaches the skin (by contact), respiratory tract (by inhalation) or digestive tract (by ingestion).

Formulation-A complete pesticide preparation as sold by a manufacturer for practical use. It includes the active ingredient and any necessary adjuvants and solvents. For use, it may or may not require further dilution or mixing with other substances. Formulation can also be defined as the process used by manufacturers in preparing a pesticide for practical use.

Half-life-The length of time required for disappearance of half of the material present in an organism or in environmental media. It is a more useful idea than “persistence” because it allows prediction of the time required to reach low target levels without making measurements over exceedingly long periods. A better term is “Half-time”, because the information only relates to a given location, and says nothing about the processes that deplete the chemical. If it evaporates or is carried away intact by water it may still exist in its original form. The term “half-life” originated with description of radioactive decay, in which elements become a totally different substance. The English language sometimes loses precision as it evolves.

Hazard-The kind of effect that a chemical can cause. Cancer, liver disease, skin irritation, reproductive problems, or some other more or less specific response that can be defined and measured. The term is also used non-specifically to signify any dangerous situation.

Herbicide-A chemical substance or cultured biological organism, used to kill or suppress the growth of plants.

Inert ingredient-Any component of a formulation that is purposely added and does not have pesticidal activity. Includes solvents and adjuvants, not manufacturing impurities.

Lethal-Causing death.

LD_{50}-Acronym for Median lethal dose.

Lethal concentration (LC_{50})-Rate at which 50 percent of test animals will be killed.

LOAEL-Acronym for lowest-observed-adverse-effect level.

Lowest-observed-adverse-effect level (LOAEL)-The lowest measured amount of a chemical that produces significant increases in frequency or severity of adverse effects in exposed subjects. In the general sense it includes all biochemical, pathological, behavioral, reproductive, genetic, and other measurable changes. The term may also be applied to any specific parameter under observation.

Margin of Safety (MOS)-In human risk assessments this is the difference between the estimated dose of a pesticide and the NOAEL. A MOS of 100 (estimated dose 100 fold less than the NOAEL) is usually considered to assure that no adverse effects will occur in human. In this report the same term is used in connection with wildlife risk assessment. The concept is the same as for humans except that rather than the NOAEL, 1/5 LD_{50} . In assessing risk for wildlife the risk is usually assessed at the population level rather than at the level of any given individual. For this reason a MOS of 1 is considered adequate.

Median effective dose (ED_{50})-The dose or dose rate that causes 50% of subjects to respond. The nature of response must be specified, i.e., sedation, elevated blood pressure, death. The ED_{10} is the dose effective in 10% of animals.

Median lethal dose (LD_{50})-The dose of a chemical, biological agent, or other substances that causes death in 50% of defined test animals.

Metabolism-The sum total of the biochemical reactions that a chemical undergoes in an organism. The processes include biochemical (enzymatic) reactions in the cells of the body that convert nutrients to energy and structural materials of the body; reactions that change wastes so they can be removed; and reactions that convert foreign substances, such as some pesticides to forms that can be excreted. The latter processes are of importance for some pesticides.

MOS-Acronym for margin of safety.

NOAEL-Acronym for no-observed-adverse-effect level.

No-observed-adverse-effect level (NOAEL)-The dose rate or concentration at and below which no adverse effects can be detected. (See threshold, LOAEL) If the estimated dose of an herbicide to a worker is very low compared to the NOAEL for the most sensitive effect found in the laboratory, no harmful effect is to be expected.

Persistence-The duration of measurable concentrations of a pesticide in soil, foliage or other media. (See half-life)

Pesticide-Any chemical (or biological product) intended to control or kill pests. Herbicides, insecticides, fungicides are all pesticides. The term is sometimes incorrectly used to mean only insecticide, for example “pesticides and herbicides”.

Q-The value used to express risk to aquatic species. It is the ratio of the critical toxicity value (similar to the no observed adverse effect level) to the expected dose based on exposure analysis.

Registration-The process by which government (e.g., Canadian federal government) authorities determine that a pesticide is suitable for use. Standards of public and worker safety, environmental impact, and usefulness must all be met.

Risk-The probability (likelihood) that some adverse or undesirable effect will take place in the future, as a result of some specified activity. Risk may relate to health, finances or any other kind of undesirable impact. Real risk may be so small that it cannot be distinguished from zero, or so great that it is a certainty. In the context of pesticides, risk is the probability that use of the pesticide will result in some specified harmful effect on workers or the public. Risk assessment is the process of estimating that probability.

Safety Factor-See Margin of Safety.

Subacute-Extending over a few days to perhaps a month. This and related terms do not carry defined time periods; consequently there is overlap in the way they are used. (See acute, subchronic, chronic).

Subchronic- For experimental studies, relatively long term, but not as long as a chronic study. Typically three to six months. (See acute, subacute, and chronic).

Threshold-The lowest dose that will produce a given effect. As a practical matter, the threshold is little different from the NOAEL.

Toxicant-A toxic agent; a poison

Toxicity-The whole pattern of harmful effects (injury, illness and other undesirable effects) that a chemical can cause. It is a property of the chemical; it does not change.

Toxicology-The group of scientific disciplines that identifies and studies the adverse effects of chemicals on biological systems, whether in the laboratory or in the field.

Toxin-A poisonous substance (poison) produced by a living organism.

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