FINAL REPORT

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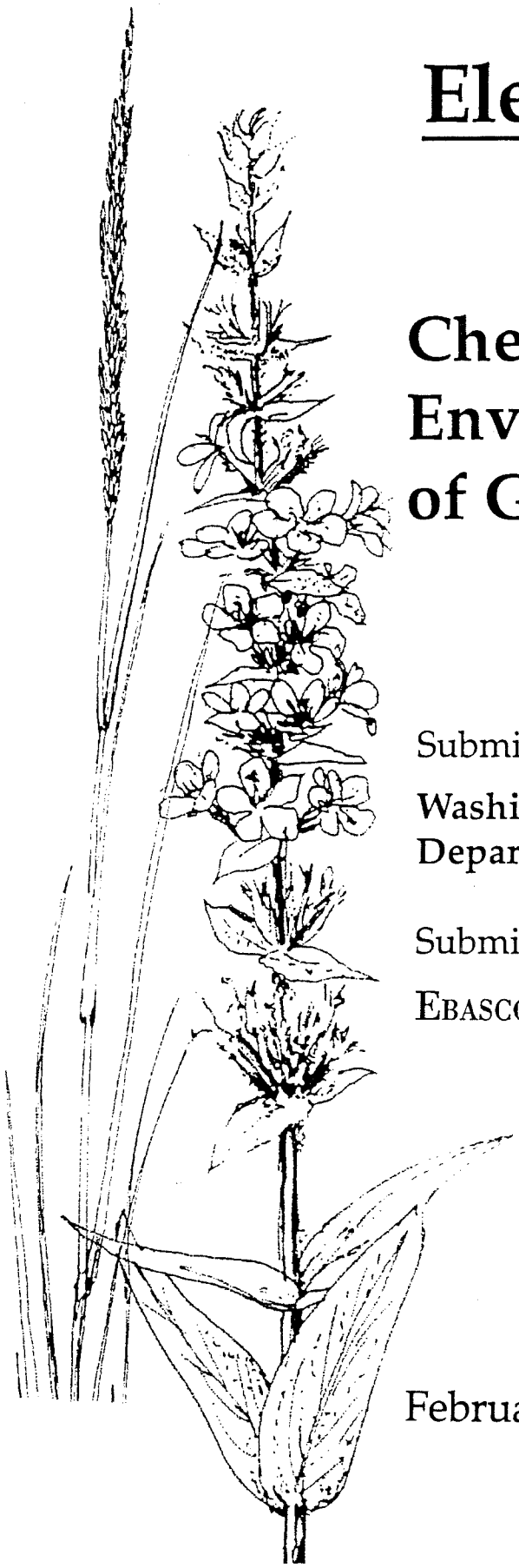

Chemical Methods Only: Environmental Effects of Glyphosate and 2,4-D

Submitted to

Washington State
Department of Ecology

Submitted by

EBASCO ENVIRONMENTAL

February 1993

NOXIOUS EMERGENT PLANT

Environmental Effects of Glyphosate and 2, 4-D

work performed by:

Herra Environmental Consultants, Inc.
1414 Dexter Avenue North, Suite 200
Seattle, Washington 98109

and

Ebasco Environmental
10900 NE 8th Street, Suite 500
Bellevue, Washington 98004

February 1993

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SECTION 1

ENVIRONMENTAL EFFECTS OF GLYPHOSATE

1.0 INTRODUCTION

This report examines the potential environmental effects of using glyphosate to control emergent wetland vegetation. The two aquatic weeds that are of most concern in Washington state are purple loosestrife (*Lythrum salicaria*), a freshwater plant, and cordgrass (*Spartina* spp.), a marine plant. The toxic effects (both acute and chronic) of glyphosate on nontarget plants, animals, and aquatic organisms are described. In addition, secondary effects such as possible increases in erosion resulting from the loss of vegetation, water quality degradation caused by release of nutrients from decaying vegetation, and habitat damage are also described.

Rodeo®, manufactured by Monsanto Agricultural Products Company, is the only glyphosate formulation that is commercially available and registered for aquatic use in Washington state. Rodeo® contains 53.8 percent glyphosate as the isopropylamine salt and 46.2 percent inert ingredients i.e., water (Monsanto 1992). Consequently, this report focuses on those studies that describe the environmental effects of Rodeo® or its active ingredient, glyphosate. However, in situations where no data are available for Rodeo®, existing information for Roundup® (a glyphosate herbicide containing 41 percent of the isopropylamine salt of glyphosate and 59 percent inert ingredients, including a proprietary surfactant) are presented. The surfactant in Roundup® is considered more toxic than the active ingredient, glyphosate. Mitchell et al. (1987a) report that Roundup® is significantly more toxic to fish than is Rodeo® (96-hour LC_{50} values for rainbow, chinook, and coho salmon ranged from 12 to 60 mg/L with Roundup®, compared to 600 to 1,070 mg/L with Rodeo®). Therefore, environmental effects information for Roundup® may be considered as a conservative estimate of the potential effects from Rodeo® before it is mixed with a nonionic surfactant. Label instructions require that Rodeo® be used with a surfactant.

The environmental effects of glyphosate and other degradation products of the parent compound depend on the transport and fate of these substances within the environment, which in turn are governed by the chemical and physical properties of these substances and the chemical, physical, and biological characteristics of the receiving environment. The acute and chronic effects of glyphosate and its degradation products on plants and animals depend on their persistence, bioaccumulation, biotransformation and elimination

properties. Environmental impacts are also dependent on the frequency of exposure, duration of exposure, and concentrations of glyphosate in the environment.

Toxicity information presented here focuses on the active ingredient, N-(phosphonomethyl)glycine, or technical glyphosate. Rodeo® must be used with a nonionic surfactant, which may increase the toxicity of glyphosate spray formulations, as discussed in the section on environmental effects of adjuvants.

2.0 TOXICITY OVERVIEW

The toxic effects of glyphosate and its degradation products are partially dependent on the transport, fate, and transformation of these substances in the environment. Because glyphosate has a very low vapor pressure and is stable in water and sunlight, the transport, fate, and environmental effects of glyphosate are strongly related to microbial biodegradation rates in both soil and water.

Glyphosate is a systemic, nonselective, broadleaf herbicide that inhibits plant growth by interfering with production of amino acids. Because of these properties, glyphosate is generally lethal on contact with broadleaf plants. Glyphosate residues in soils generally are not absorbed by plant roots, and residues do not adversely affect germination or establishment of plants. Adverse impacts to nontarget plants within the treatment area would include death unless nontarget vegetation is avoided during glyphosate application. In addition, because glyphosate phytotoxicity is variable among different species depending on absorption and translocation rates as well as other physiological factors, adverse impacts to nontarget vegetation also are likely to be variable. In other words, different plant species may have different tolerance levels to glyphosate.

Glyphosate and its primary degradation product, aminophosphonic acid (AMPA) are generally nontoxic to animals (U.S. EPA 1988). Most studies indicate that glyphosate is quickly and completely degraded in periods of days to weeks by microbial organisms in both terrestrial and aquatic environments. Some studies of both terrestrial and aquatic environments have, however, indicated that low levels of glyphosate may persist up to a year or more (O'Keefe 1985; Torstensson et al. 1989). However, normal application rates used for control of emergent vegetation do not result in doses of glyphosate shown to cause acute effects, and because glyphosate is biodegradable and nonbioaccumulative, acute lethal effects are unlikely. In addition, in studies where glyphosate persists for longer periods, levels are generally orders of magnitude below those shown to cause adverse effects to terrestrial or aquatic organisms.

Because glyphosate only persists for longer periods (up to a year or more) at low levels and does not generally bioconcentrate or bioaccumulate, the risk of long-term exposure and associated chronic effects is low. Animals may be exposed to glyphosate by

ingesting treated vegetation, by grooming activities, and by drinking water containing glyphosate. Glyphosate generally has very low bioconcentration factors in animals and does not appear to strongly bioaccumulate. Once glyphosate concentrations in the environment are reduced, or the animal is removed from the contaminated environment, glyphosate is quickly excreted by most animals. In addition, long-term, chronic toxicity studies of rats and dogs indicate that glyphosate does not appear to have any mutagenic, carcinogenic, teratogenic, developmental, or reproductive effects on animals.

However, other pesticides or herbicides (e.g., Sevin®) in the receiving environment may interact with glyphosate, thereby exerting an additive, synergistic, or antagonistic effect on glyphosate toxicity to plants and animals. There is no information currently available on the interaction or combined toxicity of Sevin® and glyphosate. However, Sevin® appears to be several times more toxic to plants and animals than glyphosate. For instance, Brown et al. (1979) report that LC_{50} values for channel catfish fry in a 4-day experiment ranged from 0.14 to 1.56 mg/L of Sevin®, whereas Folmar et al. (1979) report an LC_{50} value of 130 mg/L glyphosate for similar-sized fry in a 4-day experiment.

3.0 POTENTIAL IMPACTS TO SOIL AND SEDIMENT

3.1 PERSISTENCE AND ACCUMULATION

Both glyphosate and AMPA generally undergo rapid biodegradation (days to weeks) and completely degraded in most temperate soils. Soil half-lives and glyphosate accumulation are strongly related to biodegradation rates as well as factors influencing degradation processes (e.g., adsorption, temperature, pH, and soil moisture). Because microbial degradation occurs in both aerobic and anaerobic conditions (Rueppel et al. 1977), glyphosate and its degradation products are not likely to persist or accumulate in soils or sediments in emergent wetlands and adjacent upland areas that may receive overspray or drift. However, recent studies have demonstrated low level persistence in marine sediments and upland soils (Kroll 1991; Torstensson et al. 1989).

3.2 STABILITY AND EROSION

Following the application of glyphosate, there may be a short interim period when little or no vegetation exists in the target area. Assuming that desirable emergent vegetation does not immediately replace eliminated vegetation and that decomposing vegetation provides little soil or sediment stability, there may be a short-term risk of soil erosion and sediment instability within the glyphosate application area. No long-term erosion or sediment transport impacts are likely once the treated area is recolonized by desirable emergent or shrub vegetation. The potential for soil instability and erosion in application areas depends on a number of factors including the wetland system, watershed position and topography, water regime and gradient, and the physical and chemical characteristics of the soil and sediment.

In general, purple loosestrife and cordgrass occur in natural and manmade wetland areas having low gradient and low erosive potential. Wetlands have important ecological and economic functions, including but not limited to sediment storage and retention, nutrient cycling, food chain support, fish and wildlife habitat, water quality protection, and flood control (Mitsch and Gosselink 1986). Wetland environments often are rich in fine sediments and organic materials, which are susceptible to wind and wave erosion. Whether these sediments are eroded after herbicide treatment depends on sediment size,

location, prevailing wind direction and speed, and surrounding vegetation. For instance, in many cases winds are disrupted by surrounding trees that reduce wind speed and erosion potential. Erosive wind forces in smaller wetlands are lower because of the smaller area of open water exposed to winds. In undisturbed wetlands, soil and sediment instability and erosion generally are unlikely because native trees and vegetation provide a buffer around the treatment area.

In intertidal salt marshes treated with glyphosate to control cordgrass species, there is likely to be some short-term risk of sediment erosion and transport. Evans (1986) indicates that when cordgrass is killed by herbicide application, the roots stabilizing fine surface sediments decay over a period of several years, exposing sediments to wave erosion. Wind, wave, and rainstorm events may cause erosion, transport, and redistribution of fine sediments. The amount of erosion depends on the size of the storm event, the size of the treated area, the magnitude of tidal exchanges, and the length of exposure to erosive forces. If such events occur, sediments containing glyphosate residues could be transported and deposited in clean areas. The risk of erosion impacts can be mitigated and minimized by expediting regrowth and recolonization using revegetation techniques for desirable species.

Similar scenarios for wind and wave erosion during storm events exist for lakeshore wetlands if there are large areas of open space, permitting unimpeded winds to generate waves that may erode fine-grained sediments in the treatment area. As with the intertidal salt marsh scenario described above, there is some temporary, short-term risk of erosion and transport of glyphosate-contaminated sediments. Fine-grained silt and clay-sized sediments with adsorbed glyphosate (or its degradation products) may be suspended in the water column by wind-generated waves and transported outside the treated area. Because microbial activities will, over time, decrease the amount of glyphosate and its degradation products, it is unlikely that transport of contaminated sediments pose a risk to the receiving environment. Clean sediments may be transported to the treated area and deposited. Potential erosion impacts depend on water level fluctuations as well as the size of the treated area, the depth of water above barren sediment, decomposition rates of noxious vegetation, reestablishment rates of desirable vegetation, and proximity to adjacent vegetation. The last three factors affect sediment stability and strongly influence erosion potential. Unless water level fluctuations are extreme because of reservoir management operations or natural seasonal variation, potential erosion impacts are likely to be negligible to low. The short-term risk of erosion impacts can be reduced by

planting desirable emergent vegetation or shrubs to promote stabilization of sediments in treated areas.

Because of the erosive potential of flowing water, erosion and sediment transport impacts are more likely on riverbanks and unlined canal banks. In riverine environments, large treated bank areas may become destabilized and susceptible to cutting, slumping, or sloughing prior to regrowth and stabilization by desirable vegetation. In irrigation canal systems, because removal of nuisance emergent vegetation (e.g., loosestrife) restores irrigation canals and ditches to their unvegetated design capacity, it is unlikely that glyphosate treatment contributes to significant adverse erosion and sediment transport impacts. However, as treated vegetation decomposes, sediments trapped and retained by the living purple loosestrife may be released and transported downstream. Potential erosion impacts can be mitigated or minimized by using erosion control and revegetation techniques. Without mitigation, large quantities of sediment with adsorbed glyphosate residues could be transported to downstream areas in riverine systems. Clean sediments from upstream areas may likewise be deposited in the treated area. Neither transport of glyphosate-contaminated soils into clean areas nor transport of clean soils into glyphosate treatment sites is likely to have adverse effects on soil organisms or sediment quality, because low levels of glyphosate residuals in soils are rapidly reduced through biodegradation in most cases.

3.3 MICROFLORA

Microflora in the soil and sediment are important to decomposition, mineralization, soil fertility, and nutrient cycling processes (i.e., denitrification, nitrification, and ammonification) in terrestrial and aquatic environments. Biodegradation and inactivation of glyphosate by microorganisms are also the primary mechanisms in the breakdown and detoxification of glyphosate in soil and water (Grossbard 1985).

Laboratory studies using pure cultures of microorganisms have reported adverse effects on microflora from exposure to glyphosate (Grossbard 1976, 1974; Quilty and Geoghegan 1976, 1975). In an extensive examination of 46 cultures (Cooper et al. 1978), severe adverse effects on saprophytic soil fungi, which are important decomposers of plant cellulose, were observed only at very high concentrations of glyphosate (500 μg/mL). Forty-three percent of the isolates were killed at a glyphosate concentration of 500 μg/mL, whereas only 21 and 17 percent of the isolates failed to grow in media containing

100 and 50 μg/mL of glyphosate, respectively. By contrast, in an earlier test (Cooper et al. 1978), 91 percent of soil fungi were inhibited to varying degrees by exposure to media containing 50 μg/mL of glyphosate. The fungi used in this earlier study were unidentified, random isolates from soil.

Laboratory experiments, which are most useful for herbicide registration purposes, may not be completely reliable for predicting effects of glyphosate on microflora activity under field conditions. Greaves et al. (1980) criticize pure culture tests because:

- Isolated culture organisms may be atypical of natural soil forms
- Organisms are often stimulated to artificially high metabolic rates in culture media
- Organisms are removed from their typical ecological associations
- Interpretation of laboratory results is difficult, and extrapolation to *in situ* conditions is not possible.

Responses of microflora exposure to glyphosate in field studies are considerably different from those observed in pure laboratory cultures. For example, numbers of fungi propagules in peat and treated soils under field conditions were higher than those in controls (Roslycky 1982; Quilty and Geoghegan 1976). At concentrations of 300 μg/g of glyphosate, cellulolytic and proteolytic bacteria exhibited transient increases, whereas nitrifying bacteria were little affected (Kruglov et al. 1980; Quilty and Geoghegan 1976). Roslycky (1982) concludes that notable increases in actinomycetes at concentrations as high as 1,000 μg/g may indicate that microbes use glyphosate as a substrate.

Muller et al. (1981) report that denitrification does not appear to be adversely affected in soils at glyphosate application rates up to 2.6 kg/ha. This rate is close to the rate of 2.8 kg/ha used to control emergent vegetation in prairie pothole wetlands (Henry 1992). In addition, ammonification and nitrification processes are not inhibited at glyphosate concentrations of up to 100 μg/g. Marsh et al. (1977) report that nitrogen mineralization was increased in 100 ppm glyphosate-treated soils. These results indicate that glyphosate may be used as an energy source and may stimulate rather than inhibit fungal and bacterial growth.

Although many species of microorganisms are inhibited by glyphosate in pure culture, this effect is selective, variable in magnitude, and frequently dose-related (Grossbard 1985). Grossbard (1985) suggests that microorganisms generally tolerate glyphosate in soil much better than in pure cultures because glyphosate is strongly adsorbed to soil and is thus less likely to result in fungus mortality, although it may extend the lag phase or reduce growth rates. Concentrations that produce inhibitory effects in laboratory studies are several orders of magnitude higher than maximum concentrations typically observed in the field (i.e., 0.162 to 0.600 mg/L) (Feng et al. 1990; Henry 1992; Newton et al. 1984). In addition, glyphosate behavior *in situ* is considerably different from behavior in laboratory experiments.

Based on the information available, it is unlikely that microflora biodegradation and nutrient cycling activities will be adversely affected by concentrations of glyphosate in the receiving environment resulting from emergent vegetation control applications.

4.0 POTENTIAL IMPACTS TO AIR

Because glyphosate is essentially odorless and has a very low vapor pressure (Monsanto 1985), it can be considered nonvolatile. Therefore, air quality is unlikely to be adversely affected by glyphosate applications. However, crops or other nontarget plants could be adversely affected if they come in contact with glyphosate resulting from overspray or drift losses during application. To avoid drift and potential impacts to nontarget plant species, Monsanto (1990) specifies that glyphosate should not be applied if conditions are gusty or winds exceed 5 mph.

5.0 WATER

Some glyphosate is introduced to surface waters during application to emergent vegetation. The amount of glyphosate introduced to surface waters depends on the application rate, foliar interception, overspray, and drift. In addition, glyphosate may be introduced to surface waters from target vegetation washed by rainfall or tidal inundation when there is insufficient foliar absorption time (i.e., less than 6 hours).

5.1 SURFACE WATER

As noted in the fate and transport section, the physical and chemical characteristics of glyphosate (e.g., solubility, volatility, and mobility) contribute to its degradability in terrestrial and aquatic environments. Because aquatic environments sometimes have fewer microorganisms, biodegradation of glyphosate in surface waters generally is slower than in soils. Nonetheless, biodegradation is generally rapid (i.e., days to weeks) in freshwater and saltwater.

The degree of water quality degradation is dependent on the amount of glyphosate introduced to a given receiving water. Because interception by foliage reduces the amount of glyphosate reaching the water surface, overspray and drift are the primary sources of glyphosate introduction to surface waters. Other sources of glyphosate to surface waters, such as runoff and washoff from treated plant surfaces, are generally negligible (refer to fate and transport section). Dilution, dissipation, and biodegradation quickly diminish the concentration of glyphosate in the water column, minimizing degradation of surface water quality. For instance, the concentrations of glyphosate observed in the surface waters of prairie pothole wetlands during emergent vegetation control activities ranged from 0.140 to 0.600 mg/L 12 hours after application (Henry 1992). Eight days after glyphosate treatment, concentrations had fallen considerably, ranging from 0.035 to 0.490 mg/L. Feng et al. (1990) report a maximum concentration of 0.162 mg/L in streams intentionally oversprayed with glyphosate. Therefore, surface water quality impairment from glyphosate in the water column is not expected to occur during purple loosestrife or cordgrass control activities.

In another field study, Goldsborough and Beck (1989) found that glyphosate dissipated rapidly in small forest ponds, the half-lives varying from 1.5 to 3.5 days. They observed initial glyphosate concentrations from 215 to 352 μg/L 30 minutes following application at a treatment rate of 0.89 kg/ha. Pond surface area ranged from 0.2 to 0.7 hectares and depth ranged from 0.9 to 1.5 meters.

These field studies indicate that glyphosate dissipation is rapid in water. The results of Goldsborough and Beck (1989) appear to indicate that glyphosate treatment may contribute to increases in total digestible (Kjeldahl) nitrogen (TKN) and total phosphorus. Inorganic nitrogen (NO_3+NO_2-N) decreased from 0.14 to <0.01 mg/L in pond water 30 days after treatment. By contrast, TKN and total phosphorus increased from 1.6 to 2.5 and 0.04 to 0.29 mg/L, respectively. The bioavailability of the various nitrogen and phosphorus fractions to algae and the potential impacts on water quality require further study.

Although there is no evidence showing that normal application rates result in glyphosate concentrations in surface water causing human or environmental health risks, the Rodeo® product label (Monsanto 1990) recommends that glyphosate not be applied within 0.5 miles of surface water intakes. Because of the rapid biodegradation and dissipation of glyphosate in surface water, it is unlikely that normal applications of glyphosate in municipal wastewater treatment facilities (e.g., sewage lagoons or oxidation ditches) will result in effluent glyphosate concentrations that pose risks to the environment.

However, there may be some short-term, secondary effects on surface water quality from glyphosate treatment of noxious emergent plants. Dissolved oxygen and nutrient levels may be temporarily affected during decomposition of treated vegetation. Decomposition of dense stands of purple loosestrife and cordgrass following glyphosate application may result in temporarily reduced dissolved oxygen levels and elevated levels of nutrients (nitrogen, phosphorus, and carbon) in treated areas. Increases in nitrates, phosphates, and dissolved carbon may contribute to algal blooms. Addition of limiting nutrients, such as phosphorus or nitrogen, in surface waters are known to contribute to algal blooms (Wetzel 1975).

By contrast, release of nutrients (e.g., nitrogen, phosphorus, and carbon) from decaying emergent vegetation in treated areas may be beneficial to primary productivity and food chain support, particularly in estuarine ecosystems. No studies have been conducted to

demonstrate the effects on dissolved oxygen levels, nutrient levels, or the rate of primary productivity as a result of glyphosate application and the accompanying decomposition of vegetation.

5.2 SEA-SURFACE MICROLAYER

The sea-surface microlayer has been generally described as that layer of water lying between the surface film or nanolayer (0.03 μm) and the surface millilayer (0.1-1.0 mm) (Word et al. 1986). The sea-surface microlayer provides habitat for a wide variety of biota including bacteria, protozoans, metazoans, and the eggs or larvae of fish, lobster, and crab. Because of this important habitat, there is concern that the introduction of toxicants into the aquatic environment may eventually find their way into the microlayer.

The sea-surface microlayer is rich in naturally occurring organic matter and nutrients which are derived from the natural production that occurs in subsurface waters. Samples collected from the upper 350 μm of the sea surface indicate that the majority of material present consists of nonpolar compounds (Word et al. 1986). This material is generally composed of relatively insoluble surfactants. These compounds migrate to the sea surface and become oriented according to their hydrophilic or hydrophobic fractions (Word et al. 1986).

Levels of inorganic materials are also elevated in the surface microlayer (Word et al. 1986). Surface samples collected by Szekielda et al. (1972) indicate that up to 80 percent of the materials present may be composed of silica sand particles rather than organic materials. Hardy et al. (1985) report a residence time of 1.5 to 15 hours in the sea-surface microlayer for metals associated with particulates. While surface winds (3.6 m/sec) were observed to decrease the microlayer concentrations and residence time of these particulates, microlayer enrichment was observed.

Application of glyphosate within or adjacent to the aquatic environment could result in direct contact with the water surface or indirect contact via precipitation and runoff or tidal action. As noted previously, the toxicity of glyphosate (or other herbicides) is a function of the available concentration over a given period of time. Dilution within the water column would be expected to rapidly reduce the surface microlayer concentration and thus the toxicity of glyphosate. Because glyphosate is highly soluble in water, it would not be expected to accumulate appreciably in the surface microlayer. However,

because compounds such as glyphosate are strongly adsorbed onto particulate matter, fate and transport mechanisms similar to those observed for particulate metals may operate in the surface microlayer. Thus, based on the observed residence time of particulate metals, persistence of glyphosate in the surface microlayer for up to 15 hours might be anticipated as a worst case.

No sea-surface microlayer studies have been conducted to examine the presence or toxicity of glyphosate (Hardy 1992 personal communication; Gardiner 1992 personal communication). Because those life forms and life stages of biota present in the surface microlayer are susceptible to toxicants, adverse impacts on various species may occur if glyphosate enters surface waters located in the vicinity of application areas. Such impacts might be expected to occur, at least over short periods of time, until dispersion and dilution reduce the levels of glyphosate in the surface microlayer.

5.3 GROUNDWATER

Although highly soluble in water (1.2x104 mg/L), glyphosate does not have a high potential for leaching into groundwater (U.S. EPA 1988). Soil mobility is limited due to the rapid degradation and strong soil sorption characteristics of glyphosate and its degradation products. Therefore, routine applications are unlikely to contribute to degradation of groundwater quality. No studies have been conducted to examine or document groundwater quality degradation from glyphosate use.

6.0 POTENTIAL IMPACTS TO BIOTA

6.1 PLANTS

Acute and chronic effects of glyphosate exposure to target and nontarget plants, including macrophytes, phytoplankton, and algae, are identified in this section. A brief discussion is presented on factors that may influence glyphosate toxicity (e.g., uptake and translocation) to target and nontarget plants. Toxicological characteristics are summarized in Table 1.

6.1.1 Terrestrial and Aquatic Macrophytes

Because glyphosate is a nonselective, systemic herbicide, exposure of target and nontarget broadleaf terrestrial and aquatic macrophytes is generally lethal. However, the phytotoxic effects of glyphosate among species and among individuals of the same species are variable and depend on several factors. Factors that influence the effectiveness of glyphosate on target and nontarget plant species include application rates, timing of application, contact time, method of application, physiological factors (e.g., uptake and translocation rates), and life history stage and age of the plant.

The phytotoxicity of glyphosate to purple loosestrife and cordgrass is variable. In the only study on the effectiveness of glyphosate on purple loosestrife, total control of purple loosestrife was achieved on small test plots (20 m^2) with an application rate of 1.7 kg/acre in mid-August (Malecki and Rawinski 1985). By contrast, extensive information exists regarding the effectiveness of glyphosate in controlling cordgrass. Studies in Great Britain report variable success in using glyphosate to control cordgrass, with mortality varying from zero to 50 percent (Nature Conservancy Council undated). Furthermore, the council reports that no herbicide, including glyphosate, effectively killed young cordgrass (seedlings to 3 years). Treatment of young cordgrass was ineffective because the young plants grow close to the substrate, are exposed for shorter period between tides, and have a small leaf area. Among the factors influencing variations in mortality rate were application method, limited contact time between tides, and runoff from leaves.

Table 1. Acute and chronic effects of glyphosate in plants.

Organism	Effect	Duration	Concentration (ppb)	Life Stage	Reference
Green Algae					
Euglena gracilis	CLS	6 days	203,000	Log phase	Richardson et al. 1979
	PGR	6 days	203,000	Log phase	Richardson et al. 1979
	PSE	1.67 hours	5,100	Log phase	Richardson et al. 1979
	PSE	4 days	510,000	Log phase	Richardson et al. 1979
	PSE	1.67 hours	10,100	Log phase	Richardson et al. 1979
	PGR	4 days	200,000	Log phase	Richardson et al. 1979
Chlamydomonas eugametos	PGR	2 days	17,000	Log phase	Grossbard 1985
Dunaliella bioculata	PGR	2 days	100	Log phase	Hess 1980
Selenastrum capricornutum	EC50-BM	14-21 days	4	Log phase	Turbak et al. 1986
	EC50-OX	NR	10	Log phase	Turbak et al. 1986
	EC50-GR	7 days	13,800	Log phase	Heydens 1991
Chlorella sorokiniana	PGR	NR	2,000	Log phase	Christy et al. 1981
Chlorella pyrenoidosa	EC50-GR	NR	590,000	Log phase	Maule and Wright 1984
Chlorococcum hynosporum	EC50-GR	NR	68,000	Log phase	Maule and Wright 1984
Blue-green Algae					
Anabaena flos-aquae	EC50-GR	7 days	15,000	NR	Heydens 1991
Diatoms					
Skeletonema costatum	EC50-GR	7 days	640	NR	Heydens 1991
Navicula pelliculosa	EC50-GR	7 days	42,000	NR	Heydens 1991
Periphyton					
Lotic	PGR	NR	>300	NR	Austin et al. 1991
Lentic	EC50-GR	NR	42,000	NR	Goldsborough and Brown 1988

Continued....

Table 1. Acute and chronic effects of glyphosate in plants (continued)

Organism	Effect	Duration	Concentration (ppb)	Life Stage	Reference
Submerged and Floating Macrophytes					
Elodea canadensis	EC50-GR	NR	>1,000		Forney and Davis 1981
Myriophyllum spicatum	EC50-GR	NR	>1,000		Forney and Davis 1981
Water Milfoil	EC50-GR	NR	>1,000		Forney and Davis 1981
Vallisneria americana	EC50-GR	NR	>1,000		Forney and Davis 1981
Lemna spp. (duckweed)	EC50-GR	NR	>1,000		Forney and Davis 1981
Lemna minor	PHY	NR	1,690		O'Brien and Prendeville 1979

CLR - Measurable change in chlorophyll content
EC50 - Median effective concentration
BM - Measurable change in biomass
OX - Measurable change in oxygen production
GR - Measurable change in growth
NR - Not reported
PHY - Physiological effects: changes in organic functions or functions of organs
PRG - Changes in population growth
PSE - Photosynthesis effect: change in plant productivity

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Other studies on the efficacy of glyphosate have cited physiological causes for variations in phytotoxicity. Boerboom et al. (1990) found that clones of birdsfoot trefoil exhibited differential rates of glyphosate translocation. These authors report that tolerance to glyphosate was correlated to 5-enolpyruvylshikimate 3-phosphate synthase enzyme activity. They further conclude that differential rates of tolerance to glyphosate among individuals may be due to differential rates of spray retention, absorption, and translocation. D'Anieri et al. (1990) also suggest that variations in the translocation among species may explain some of the variability in glyphosate phytotoxicity among different species of plants. D'Anieri et al. (1990) report an interactive effect among glyphosate translocation efficiency, water stress, and application timing (e.g., during fall dormancy) in red maple and loblolly pine seedlings.

Results of these studies suggest that glyphosate phytotoxicity to target and nontarget plants appears to depend on several factors, including plant physiological status, plant retention, uptake and translocation rates, timing of application, and application rate. The magnitude of adverse effects on nontarget plants also depends on the extent of overspray and drift losses during glyphosate application, as well as species-specific uptake and translocation rates. Any adverse effects on nontarget plant species are expected to be short-term, because glyphosate biodegrades rapidly and does not appear to inhibit seed germination or establishment immediately after application (Chakravarty and Chatarpaul 1990).

6.1.2 Submerged Aquatic Vegetation

Limited studies have been conducted on the effects of glyphosate on submerged aquatic plants. In a laboratory growth inhibition study, no adverse effects were observed on the growth of *Elodea canadensis* (elodea), *Myriophyllum spicatum* (water milfoil), or *Vallisneria americana* (wild celery) exposed to glyphosate concentrations of up to 1,000 ppb for 3 to 6 weeks (Forney and Davis 1981). The authors conclude that glyphosate was essentially nontoxic to the tested species. These results appear consistent with the findings of other investigators who report that submerged aquatic plants are either resistant (Evans 1978) or affected by only very high glyphosate concentrations (Peverly and Crawford 1975).

Although available information on the effects of glyphosate on submerged aquatic plants suggests that there is no risk to this group of plants, additional information regarding

impacts on ecologically important species, such as eelgrass, is needed. No information is currently available on the effects of glyphosate on eelgrass. However, information on the potential impacts of glyphosate on eelgrass and other submerged aquatic vegetation is likely to come from a *Spartina alterniflora* control study being conducted in Willapa Bay, Washington by the U.S. Fish and Wildlife Service and the University of Washington School of Fisheries.

6.1.3 Algae, Periphyton, and Phytoplankton

Results of laboratory and field studies of both freshwater and marine algae, periphyton, and phytoplankton generally indicate that glyphosate is slightly toxic to practically nontoxic to these plant groups (see Table 1). However, there is considerable intraspecific and interspecific variation among studies on the reported population growth effects of glyphosate. For example, glyphosate growth inhibition of various algal species ranges from a level of 100 mg/L in a 2-day exposure of the green alga *Dunaliella bioculata* (Felix et al. 1988) to a level of 203,000 mg/L for the flagellated euglenoid (*Euglena gracilis*) in a 6-day exposure experiment (Richardson et al. 1979). Some of the variation in toxicity is explained by the increased sensitivity or susceptibility of test organisms in laboratory cultures and environments. Several studies report adverse effects for low levels (<1 ppm) of glyphosate (Heydens 1991; Felix et al. 1988; Turbak et al. 1986). However, glyphosate concentrations that cause adverse effects on algae and periphyton are generally several orders of magnitude greater than concentrations that would occur with normal application for aquatic weed control. Moreover, toxicities observed in the laboratory have not been corroborated by similar results in field studies (Garnett 1991; Heydens 1991). Tooby (1985) attributes the lack of toxicity *in situ* to the rapid loss of chemical from the water, primarily through adsorption and dilution.

Growth rates and biomass accrual of lotic periphyton (i.e., plants attached to surfaces in moving water environments) in experimental streams were not adversely affected by low concentrations of glyphosate (1 to 300 ppb) (Austin et al. 1991). The authors report that periphyton biomass accrual appears to be stimulated by glyphosate addition, indicating that glyphosate may be used as a source of phosphorus. Bothwell (1985) demonstrated that lotic periphyton growth rates are saturated at very low phosphorus levels (0.003 ppm). Austin et al. (1991) conclude that there may be some potential for glyphosate to contribute to cultural eutrophication in oligotrophic, nutrient-limited ecosystems.

Therefore, the potential for glyphosate to be used as a source of phosphorus and to contribute to cultural eutrophication requires further study.

Growth inhibition of algae, periphyton, and phytoplankton are unlikely to occur because maximum observed glyphosate concentrations in the receiving environment are generally lower than levels that inhibit growth, and these concentrations decrease rapidly over a period of days to weeks. If growth inhibition of algae, periphyton, and phytoplankton occurs, it is likely to be only temporary due to the rapid biodegradation and dissipation of glyphosate in the environment.

Because of the importance of freshwater and marine algae, periphyton, and epiphytes to the food chain, additional studies are needed to evaluate potential chronic effects of glyphosate (e.g., mutagenicity, bioconcentration, and bioaccumulation). Moreover, quantification of environmental concentrations resulting from routine emergent weed control applications is needed, particularly for marine environments. Additional toxicity information, particularly for critically important species such as eelgrass, is also needed. There appears to be no information available on potential bioaccumulation, effects on species composition, or trophic transport. However, based on glyphosate's chemical characteristics (i.e., high solubility in water and low octanol/water partition coefficient), it is not likely to bioaccumulate. Finally, information is needed on secondary effects due to nutrient release from decomposing vegetation after glyphosate treatment, particularly in nutrient-limited systems.

7.0 POTENTIAL IMPACTS TO ANIMALS

Acute and sublethal effects of glyphosate exposure on various animal groups are summarized in this section. Information is presented on the bioconcentration and bioaccumulation of glyphosate in these animal groups, and gaps in data on the chronic and acute effects of glyphosate and its degradation products are summarized.

7.1 MAMMALS

Wetland environments that support purple loosestrife and cordgrass are important to a number of terrestrial and aquatic mammals. Riverine, lakeshore, estuarine, and some manmade wetlands (e.g., retention/detention facilities and irrigation canals) may be important habitat areas for large and small mammals. Glyphosate applications to control noxious emergent weeds in these environments may adversely affect animals by altering habitat quality and quantity.

The effects of glyphosate applications on large and small mammals is primarily related to alteration of habitat, food quality, food preference, exposure concentration, persistence, and accumulation. Adverse effects are measured and evaluated by examining the effects of exposure on reproductive success, growth, survival, and population. Overspray and drift from treated areas may adversely affect adjacent nontarget plant communities. These adverse effects may be manifested as changes in vegetational species composition, density, diversity, and abundance. Herbivorous animals such as small mammals and deer may be attracted to changes in vegetational communities resulting in abundance and diversity of preferred foods or cover types.

The response of mammals to herbicide application can be predicted if data are available on the range of habitats occupied by a species and its density in these habitats (Morrison and Meslow 1983). In other words, the potential adverse effects of noxious emergent weed control activities using glyphosate depend on the numbers and types of mammals using treated areas; the availability of other suitable habitats if unfavorable alterations in habitat occur; and the effects of exposure on reproduction, growth, and survival of these animals. Adverse effects from glyphosate treatment on preferred habitat areas are likely to be temporary, pending regrowth of desirable emergent vegetation in the treated area.

All of the existing acute and chronic toxicity information on glyphosate effects on small mammals in field conditions is based on Roundup® application in coniferous forest ecosystems. Roundup® is not registered for aquatic use in Washington state. However, because no data are available for Rodeo®, these results are presented here to provide an indication of the potential acute and chronic effects of glyphosate (Rodeo®) on small mammals under field conditions.

Sullivan (1990) and Ritchie et al. (1987) report short-term increases in the abundance and quality of food and cover (i.e., grasses and herbs) for deer mouse and Oregon vole 1 year after Roundup® application. No adverse effects on reproduction, growth, or survival have been observed in Oregon vole or deer mouse in test plots treated with Roundup® (Sullivan 1990; Sullivan and Sullivan 1981). Based on these data, it appears unlikely that Rodeo®, which has been proven less toxic than Roundup® to fish (acute toxicity basis), will cause acute or chronic effects in small mammals (Mitchell et al. 1987a).

7.1.1 Acute Effects

Available acute toxic effects data for glyphosate are based primarily on analyses using rats and rabbits. Acute oral and dermal glyphosate toxicities occur at high concentrations (see Table 2). The single dose acute oral toxicities (i.e., LD_{50}) of glyphosate in rabbits and rats range from 3,800 mg/kg for rabbits to 5,600 mg/kg for rats (USDA 1981; Monsanto 1983). Acute dermal toxicity of glyphosate to rabbits is also very high, ranging from >5,000 to >7,940 mg/kg (Weed Science Society of America 1983; Monsanto 1983). The variation in dermal toxicity is a result of different testing methods. The higher dermal toxicity value (>7,940 mg/kg) represents the minimum lethal dose for skin absorption, whereas the lower value (>5,000) is reported as an LD_{50} (median lethal dose). In all cases, these concentrations are several orders of magnitude above the potential dose from exposure to the maximum observed concentrations in streams and ponds resulting from normal application rates (0.162 to 0.600 mg/L) (Henry 1992; Feng et al. 1990; Newton et al. 1984).

Significant gaps exist in the available acute toxicity information on pure glyphosate effects on mammals that are likely to inhabit or use emergent wetlands. For example, no information is available on the toxicity of glyphosate or its degradation products to raccoons, bears, mustelids (e.g., minks, weasels, and ermines), foxes, elks, beavers, muskrats, shrews, or other terrestrial or aquatic mammals. However, based on

Table 2. Acute and chronic toxicity of glyphosate to mammals.

Organism	Nature of Exposure	Effect	Concentration (mg/kg)	Toxicity Category	Reference
Rabbit	Single dose acute oral (8 day)	LD50	3,800	Practically nontoxic	U.S. Dept. of Energy 1983
Rat	Single dose acute oral (8 day)	LD50	5,400	Practically nontoxic	Weed Sci. Soc. Am. 1983
	Single dose acute oral (8 day)	LD50	>5,000	Practically nontoxic	Monsanto 1983
	Single dose acute oral (8 day)	LD50	4,040	Practically nontoxic	USDA 1981
Rabbit	Single dermal application - 24-hr exposure	MLD	0-300	Practically nontoxic	Weed Sci. Soc. Am. 1983
	Single dermal application - 24-hr exposure	LD50	>5,000	Practically nontoxic	Monsanto 1983
Rat	0, 30, 100 or 300 ppm dietary for 2 yrs; chronic	No significant differences	0-300		Monsanto 1983
Dog	0, 30, 100 or 300 ppm dietary for 2 yrs; chronic	No significant differences	0-300	No carcinogenic or histopathologic effects	USDA 1981
Mouse	0, 30, 100 or 300 ppm dietary for 2 yrs; chronic	No significant differences	0-300	No evidence of increase in cytoplasmic vacuolation	USDA 1981

Continued...

Table 2. Acute and chronic toxicity of glyphosate to mammals (continued).

Organism	Nature of Exposure	Effect	Concentration (mg/kg)	Toxicity Category	Reference
Rat	3 gen. reprod. dietary intake 30, 100, or 300 ppm	No significant differences	0-300	No reproductive or teratogenic effects	USDA 1981
Rat	Teratology test	NOEL	3,500	No teratogenic effects	U.S. EPA 1982
Rabbit	Fetotoxicity test	NOEL	175		U.S. EPA 1982
Mouse	0, 100, or 300 ppm dietary for 18 mos.	No significant differences	0-300	No correlation between treatment and carcinogenicity	USDA 1981
Mouse	Dominant lethal test		5 or 10	No mutagenic effects	USDA 1981

MLD - Minimum lethal dose
NOEL - No-observed-effects level

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glyphosate toxicities to rats and rabbits, it appears unlikely that glyphosate will adversely affect terrestrial or aquatic mammals.

7.1.2 Chronic Effects

Laboratory studies of the chronic effects of glyphosate show that it is slightly to practically nonirritating to the eyes of rabbits and also practically nonirritating to their skin (Monsanto 1983). Ninety-day studies on the subchronic oral toxicity of glyphosate to rats and dogs indicate that high oral doses of up to 2,000 ppm do not significantly affect behavior, survival, or body weight. No significant reproductive, teratogenic, mutagenic, or carcinogenic effects from exposure to concentrations of glyphosate of up to 300 ppm are reported in long-term (2-year) laboratory studies of rats, dogs, rabbits, and mice. Therefore, based on these studies, it appears unlikely that glyphosate will have adverse chronic effects on mammals that inhabit or use emergent wetlands.

7.2 BIRDS

Acute and toxic effects of glyphosate on birds have been evaluated using bobwhite quails and mallard ducks. The diet of gallinaceous birds, such as the bobwhite quail, consists primarily of vegetation and seeds. Bobwhite quails, which are smaller and ingest more food per unit of body weight than their larger gallinaceous allies, are used to assess glyphosate toxicity because they are more likely to be exposed to higher doses of glyphosate than are other members of this group (Heydens 1991). Therefore, the bobwhite quail is a good indicator of potential adverse effects of pesticide exposure to this group of birds. Similarly, mallard ducks are used to evaluate the potential toxicity of glyphosate to dabbling ducks. The diet of dabbling ducks consists primarily of terrestrial and aquatic vegetation and some aquatic insects.

7.2.1 Acute Effects

Acute toxicological characteristics of glyphosate in birds are summarized in Table 3. The toxicity of glyphosate to bobwhite quail chicks and mallard ducklings has been assessed using both single dose acute oral (followed by an 8-day observation period) and dietary intake tests. Bobwhite quail chicks and mallard ducklings were exposed to dietary glyphosate concentrations of up to 4,640 mg/kg. No mortality or treatment-related signs (e.g., abnormal weight gain) were observed in either species. Because there were no

Table 3. Acute and chronic toxicity of glyphosate to birds.

Organism	Effect	Nature of Exposure	Concentration (mg/kg)	Toxicity Classification*
Bobwhite quail chicks	LD50	Single dose acute oral (8 day)	>4,640	Practically nontoxic
Bobwhite quail chicks	LC50	8-day dietary	994 mg/kg-day or >4,640 ppm	Practically nontoxic
Mallard duck chicks	LC50	8-day dietary	1,106 mg/kg-day or >4,640 ppm	Practically nontoxic
Bobwhite quail adult female	None	One-generation study of reproduction effects	>1,000 ppm or 96 mg/kg-day	Practically nontoxic
Mallard duck adult female	None	One-generation study of reproduction effects	>1,000 ppm or 126 mg/kg-day	Practically nontoxic

Source: Heydens 1991
* Toxicity classification follows Christensen 1976

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adverse reactions at these exposure levels, the LD_{50} and LC_{50} values are reported as >4,640 mg/kg and >4,640 ppm, respectively. The latter figure is equivalent to an average dietary intake of 994 mg/kg-day for bobwhite quail and 1,106 mg/kg-day for mallards, well above potential exposures at normal application rates (Heydens 1991).

This information indicates that glyphosate is practically nontoxic to bobwhite quails and mallard ducks and presumably to other members of the gallinaceous and dabbling duck bird groups. Because lethal toxicities are unlikely to be reached, no adverse effects on gallinaceous or dabbling duck bird populations are expected from normal applications of glyphosate for control of aquatic weeds. Additional lethal toxicity information is needed on other birds groups that use wetland areas, such as perching birds and shorebirds. These birds nest and feed in emergent wetlands and may have different tolerances to glyphosate because of different metabolic rates and diets.

7.2.2 Chronic Effects

Reproductive effects of glyphosate on bobwhite quails and mallard ducks have been studied. Females of these two species were exposed to dietary concentrations of up to 1,000 ppm throughout a one-generation study (17 weeks). The numbers of eggs laid, eggs cracked, viable embryos, and normal hatchlings, as well as eggshell thickness and post-hatchling growth and survival, were measured to assess the reproductive effects of exposure to glyphosate. No measurable effects were identified at any of the dietary concentrations evaluated. The highest evaluated concentration, 1,000 ppm, is equivalent to an average daily intake of 96 and 126 mg/kg in quails and mallard ducks, respectively (Heydens 1991).

Based on this information, it appears unlikely that adverse effects on successful reproduction will occur as a result of routine glyphosate applications for the control of noxious emergent vegetation.

Additional information is needed on other potential chronic effects of glyphosate on birds. No information is currently available on mutagenic, carcinogenic, or teratogenic effects of glyphosate on birds. Although glyphosate is unlikely to bioconcentrate in birds (based on its low water/octanol partition coefficient and the bioconcentration factors determined for fish), additional information is needed on the bioconcentration and accumulation of glyphosate in birds.

7.3 FISH

7.3.1 Acute Effects

Acute toxicities of glyphosate in fish have been reported by several investigators. The range of glyphosate concentrations causing acute toxic effects are summarized in Table 4. Reported lethal concentrations for glyphosate vary considerably among studies for individuals of one species and for different species. For instance, 96-hour LC_{50} values for fathead minnows range from 97 to 127 mg/L. Similarly, lethal concentrations for rainbow trout vary from 130 to 240 mg/L. In all cases, the reported lethal concentrations for salmonids and other fishes are classified as slightly to practically nontoxic (see Table 4). Furthermore, these concentrations are one to three orders of magnitude higher than the highest concentration (0.600 mg/L) observed for a 2.8 kg/ha (2.5 lb/acre) treatment application rate using Rodeo® to control cattails in prairie pothole wetlands (Henry 1992). Other investigations report similar concentrations of glyphosate in water, ranging from 162 to 352 ppb for application rates of 2.0 to 3.3 kg/ha (Feng et al. 1990; Goldsborough and Beck 1989; Newton et al. 1984). Still other investigations report that no acute toxicity hazards are likely at usual application rates (Mitchell et al. 1987a; Tooby 1985).

Mitchell et al. (1987a) report lethal concentrations for rainbow trout (*Salmo gairdneri*), chinook salmon (*Oncorhynchus tshawytscha*), and coho salmon (*O. kisutch*) of 680 to 1,070 mg/L, 750 to 1,440 mg/L, and 600 to 1,000 mg/L, respectively, for a mixture of Rodeo® and Valent X-77 surfactant (formerly Ortho X-77). Higher toxicities were observed for the Rodeo®/X-77 mixture than for pure glyphosate, indicating that the surfactant is more toxic than glyphosate. For example, lethal concentrations to rainbow trout were 1,100 mg/L for Rodeo® (i.e., pure glyphosate), compared to 680 mg/L for Rodeo®/X-77. These results indicate that pure glyphosate was practically nontoxic to rainbow trout, chinook salmon, and coho salmon (Mitchell et al. 1987a). (Note that the Rodeo® label requires a nonionic surfactant be used with Rodeo®.)

Variation in acute toxicity to salmonids and other species may be caused by several factors including variation in water quality characteristics (e.g., pH, hardness, alkalinity, temperature, and conductivity) and differences in toxicity testing methods. Differences in water quality can affect acute toxicity of pesticides (Rand and Petrocelli 1985). Although inconclusive, some authors have indicated or suggested that glyphosate toxicity to

Table 4. Acute and chronic effects of glyphosate in fish.

Organism	Effect	Duration	Concentration (ppb)	Life Stage	Toxicity Classification*	Reference
Rainbow trout	LC50	4 days	130,000	0.8 grams	Practically nontoxic	Johnson and Finley 1980
	LC50	1 day	140,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	1 day	240,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	140,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	140,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	240,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	240,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	AVO	4 days	10,000	Fry	Moderately toxic	Folmar 1976
Bleak	LC50	4 days	16,000	Fingerling	Slightly toxic	Linden et al. 1979
Bluegill	LC50	1 day	150,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	1 day	230,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	1 day	240,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	135,000	0.9 grams	Practically nontoxic	Johnson and Finley 1980
	LC50	4 days	140,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	140,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	220,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
Channel catfish	LC50	1 day	130,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	130,000	0.5-2.2 grams	Practically nontoxic	Folmar et al. 1979
	LC50	4 days	130,000	2.2 grams	Practically nontoxic	Johnson and Finley 1980
Common carp	LC50	1 day	12,800	4.5 cm, 110 grams	Slightly toxic	Li and Chen 1981
	LC50	4 days	115,000		Practically nontoxic	Li and Chen 1981
Fathead minnow	LC50	1 day	97,000	0.5-2.2 grams	Slightly toxic	Folmar et al. 1979
	LC50	4 days	97,000	0.5-2.2 grams	Slightly toxic	Folmar et al. 1979
	LC50	4 days	97,000	0.5-2.2 grams	Slightly toxic	Johnson and Finley 1980
	LC50	NR	127,000	0.6 grams	Practially nontoxic	Henry 1992

Continued...

Table 4. Acute and chronic effects of glyphosate in fish (continued).

Organism	Effect	Duration	Concentration (ppb)	Life Stage	Toxicity Classification*	Reference
Flagfish	LC20-D	2 hours	2,940	8 days posthatch	Moderately toxic	Holdway and Dixon 1988
	LC20-D	2 hours	29,600	8 days posthatch	Slightly toxic	Holdway and Dixon 1988
Grass carp	LC50	1 day	26,000	Fingerling 1st yr	Slightly toxic	Tooby et al. 1980
	LC50	2 days	24,000	Fingerling 1st yr	Slightly toxic	Tooby et al. 1980
	LC50	4 days	15,000	Fingerling 1st yr	Slightly toxic	Tooby et al. 1980

LC50 - Median lethal concentration (96-hr)
LC20-D - Lethal concentration to 20 percent of the population
* Toxicity classification follows Christensen 1976
NR - Not reported

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salmonids is related to changes in dilution water hardness, pH, alkalinity, temperature, and conductivity. Folmar et al. (1979) reported increased acute toxicity (96 hr-LC_{50}) to rainbow trout and bluegill relative to increases in water temperature and pH (6.5 to 7.5). However, no increase in acute toxicity was observed when pH changed from 8.5 to 9.5. By contrast, Mitchell et al. (1987a) reported no significant differences ($p < 0.05$) in glyphosate toxicity as a function of dilution water pH, hardness (4.5 to 85 mg/L $CaCO^3$), and conductivity. However, they concluded that the toxicity of a Rodeo®/X77 mixture to coho and chinook salmon and rainbow trout did appear to decrease in waters with relatively high pH (7.8), hardness (77 mg/L $CaCO^3$), and conductivity (130 umhos/cm). Wan et al. (1989) also reported decreasing acute toxicity (96 hr-LC_{50}) to coho, chum, pink, and chinook salmon and rainbow trout relative to increasing pH and hardness. They reported 96 hr-LC_{50} values of 10-27 mg/L glyphosate in slightly acidic (pH 6.3), soft water (3.5-10 mg/L $CaCO^3$) and 96 hr-LC_{50} values of 148-211 mg/L glyphosate in slightly basic (pH 8.2), hard water (86 mg/L $CaCO^3$). In addition, they concluded that hardness and pH of dilution waters appear to be the key factors influencing acute toxicity of glyphosate to salmonids. Mitchell et al. (1987a) suggest that differences in toxicity testing methods among studies may explain the wide range in lethal concentrations reported in the literature.

Another source of variation in the reported lethal concentrations may be varying percentages of impurities in different batches of glyphosate. Glyphosate reportedly may contain up to 0.1 ppm N-(nitrosoglyphosate), for which no toxicity information exists (USDA 1992).

Differences in the kind of surfactant used in a glyphosate formulation (i.e., surfactant plus technical grade glyphosate) also have been suggested as a potential source of variation in lethal concentrations (Henry 1992; Mitchell et al. 1987a; Hildebrand et al. 1982). Several researchers suggest that the surfactant used in a glyphosate formulation has additive effects on the lethal toxicity level and that the surfactant itself is more toxic to salmonids than technical glyphosate alone (Wan et al. 1989; Mitchell et al. 1987a; Hildebrand et al 1982). Wan et al. (1989) report lethal concentrations in salmonids exposed to pure glyphosate ranging from 16 to 184 mg/L, depending on the dilution water type. By contrast, they report lethal effects in salmonids for surfactant concentrations ranging from 1.6 to 3.3 mg/L. Similar findings are reported by Mitchell et al. (1987a) as discussed above.

Additional lethal toxicity information is needed to more accurately assess the potential effects of glyphosate on salmonids and other fishes, particularly on larvae and fry, which may be more vulnerable to pollutants. Available acute toxicity information concentrates primarily on freshwater fishes. Information is lacking on the acute toxicity of glyphosate on all life history stages of marine fishes. No information is currently available on glyphosate toxicity to herring, ocean perch, sturgeon, or sea-run cutthroat trout. For accurate assessment of the potential effects of glyphosate on fish populations, information on the acute toxicity to larvae, fry, fingerlings, smolts, and adults is needed.

7.3.2 Chronic Effects

Few studies have been conducted on the chronic effects of glyphosate on fish. Some studies identify chronic effect threshold concentrations for rainbow trout and salmon fry and fingerlings in freshwater (Table 4). All of these studies found that threshold effect concentrations were above receiving water concentrations expected for usual application rates of glyphosate to control emergent vegetation. In addition, concentrations of glyphosate causing chronic effects are between 2.5 and 5 times those allowed by label restrictions. The maximum glyphosate concentration following label requirements (Monsanto 1991) for control of purple loosestrife over a one-acre area containing 6 inches of water would be approximately 2 mg/L, which is five times lower than the concentration resulting in rainbow trout avoidance behavior. This assumes even diffusion of the glyphosate over the entire area. Similarly, the maximum glyphosate concentration for Spartina control that would occur by following the manufacturer specifications (up to 7.5 pints Rodeo®/acre) would be approximately 4 mg/L, which is 2.5 times lower than the concentration causing avoidance in rainbow trout.

Information on chronic toxicity effects of glyphosate on osmoregulation and smoltification of salmonids is limited to findings of a study of Roundup®. Unlike Rodeo®, Roundup® contains glyphosate, inert ingredients, and a surfactant (although the Rodeo® label requires use of a surfactant). Mitchell et al. (1987b) examined the effects of Roundup® exposure on the osmoregulation of coho salmon smolts. They found that yearling smolt survival after 24 hours in seawater was unaffected by exposure to Roundup® concentrations of up to 10 times those encountered in water immediately after aerial application (2.78 mg/L). Furthermore, they report no abnormal responses in smolts when a 10-day freshwater recovery period was permitted between herbicide and seawater exposures. These investigators suggest that coho smolt osmoregulation and survival

would not be affected by Roundup® applied at rates specified on the product label. Based on this information, it appears likely that application of Rodeo® without a surfactant will not adversely affect smoltification, smolt survival, or completion of the freshwater-to-saltwater transition phase of salmonid or other anadromous fish life cycles. However, the use of Rodeo® without a surfactant would likely be considered a violation of label requirements.

Chronic toxicity information on salmonids and other fishes also appears to be limited to studies of the toxic effects of Roundup®. Morgan and Kiceniuk (1992) examined the effects of 1- and 2-month exposures to various concentrations of Vision® (the same formulation as Roundup®) on growth, carcinogenicity, feeding, and agonistic behavior in rainbow trout hatchery fingerlings. They found no significant adverse affects on the feeding or agonistic behavior of rainbow trout exposed to Vision® concentrations of 4.25, 8.0, and 45.75 ppb. In addition, exposures to these concentrations resulted in no significant adverse effects on growth in length or body weight or in the number of gill lesions. These investigators conclude that the sublethal levels of Roundup® test concentrations are relatively nontoxic. As in the smoltification toxicity results, these results suggest that Rodeo® (which has been found less toxic than Roundup® in acute toxicity studies), is unlikely to adversely affect salmonid behavior, growth, or cancer rates.

Glyphosate bioaccumulation and excretion in fish has not been extensively studied. Sacher (1978) reports a bioconcentration factor of 1.6 for bluegill sunfish exposed to 0.6 ppm glyphosate for a period of 28 days. However, Sacher (1978) reports much lower bioconcentration factors for rainbow trout, largemouth bass (*Micropterus salmoides*), and channel catfish (*Ictalurus punctatus*), of 0.03, 0.04, and 0.18, respectively. These bioconcentration factors resulted from exposure to 10 ppm of glyphosate for a period of 14 days, followed by a 35-day period in clean water. In clean water, the rainbow trout and catfish eliminated approximately 50 percent of the initial concentration within 4 to 6 days. Brandt (1984) reports similar findings, with bioconcentration factors in fish tissue ranging from 0.2 to 0.3 after fish were exposed to glyphosate for 10 to 14 days. The tested fish species are not named. These low bioconcentration factors indicate that there is a low bioaccumulation potential for glyphosate and therefore a low potential for adverse, long-term impacts on fish (Reinert and Rodgers 1987). Tooby (1985) concludes that based on these bioconcentration factors, depuration rates, and the low

octanol/water partition coefficient of glyphosate, bioaccumulation is unlikely to occur in the tested fish species. Based on these data, glyphosate appears to be nonaccumulative.

Although limited information on the chronic effects (e.g., reproductive, carcinogenic, behavioral, and physiological effects) of Roundup® formulations exists, additional information is needed to assess the potential mutagenic, teratogenic, carcinogenic, and reproductive effects of pure glyphosate on salmonids and other fishes. Additional data on the latent secondary effects of glyphosate exposure, such as immunosuppression, susceptibility to parasites or diseases, and reduced fitness, are also lacking. Significant information gaps also exist on chronic toxicities to marine fish species. No information is currently available on the effects of glyphosate degradation products on fish.

7.4 AQUATIC INVERTEBRATES

7.4.1 Acute Effects

Lethal toxicities of glyphosate to various freshwater and marine organisms are summarized in Table 5. Lethal toxicities range from 22 ppm for a harpacticoid copepod to 1,157,000 ppb for a freshwater leech (*Nepahelopsis obscura*), indicating that glyphosate is only slightly toxic (10,000 to 100,000 ppb) to practically nontoxic (100,000 to 1,000,000 ppb) for most of those freshwater and marine invertebrates tested. Glyphosate has been classified as an insignificant hazard (LC_{50} >1,000 ppb) for a few organisms. Tolerance to glyphosate and lethal toxicities vary depending on the life history stage of the organism, receiving environment conditions, and toxicity testing methods. Based on these data, it appears unlikely that lethal toxicities will occur in either freshwater or marine environments for most organisms during application of glyphosate to control emergent aquatic vegetation.

Henry (1992) found no significant differences in the survival of six species of invertebrates between treated and control wetlands at an application rate of 2.8 kg/ha. Although low levels of glyphosate may persist in some soils and sediments for up to a year, glyphosate generally has a short half-life (days to weeks) in water and sediment (where many invertebrates live), and maximum concentrations of glyphosate in the receiving waters are typically well below lethal levels. Therefore, it is unlikely that populations of freshwater or marine invertebrates will be adversely affected through emergent vegetation control activities.

Table 5. Acute and chronic effects of glyphosate in freshwater and marine aquatic invertebrates.

Organism	Effect	Concentration (ppb)	Life Stage	Toxicity Classification*	Reference
Freshwater					
Daphnia magna (Waterflea)	EC50-MOR	545,000	1st instar	Practically nontoxic	Henry 1992
Chironomous riparius (Midge)	EC50-MOR	1,308,000	4th instar	Insignificant hazard	Henry 1992
Chironomus plumosus	EC50-IM	55,000		Moderately toxic	Folmar et al. 1979
Chironomous riparius (Midge)	96-hr EC50-M	727,000	Adult	Practically nontoxic	Henry 1992
Stagnicola elodes (Pond snail)	EC50	242,000	Adult	Practically nontoxic	Henry 1992
Nephaelopsis obscura (Leech)	LC50	1,157,000	Adult	Insignificant hazard	Henry 1992
Anopheles quadrimaculatus (Mosquito)	LC50	673,000	4th instar	Practically nontoxic	Holck and Meek 1987
Red swamp crayfish	LC50	47,310	Immature, 25-40 mm	Slightly toxic	Holck and Meek 1987
Marine					
Palaemonetes vulgaris (Grass shrimp)	TL50-MOR	281,000	NR	Practically nontoxic	Heydens 1991
Mysidopsis bahia (Mysid shrimp)	LC50	>1,000,000	NR	Insignificant hazard	Heydens 1991

Continued...

Table 5. Acute and chronic effects of glyphosate in freshwater and marine aquatic invertebrates (continued).

Organism	Effect	Concentration (ppb)	Life Stage	Toxicity Classification*	Reference
Marine (continued)					
Crassostrea virginica (Atlantic oyster larvae)	EC50	>10,000	Egg/larvae	Slightly toxic	Heydens 1991
Tripneustes esculentus (Sea urchin)	96-hr EC50-I	>1,000,000	NR	Insignificant hazard	Heydens 1991
Uca pugilator (Fiddler crab)	LC50	934,000	NR	Practically nontoxic	Heydens 1991

* Toxicity classification follows Christensen 1976
LC50 - Median lethal concentration; 96-hr unless indicated otherwise
EC50- 48 hr. unless indicated otherwise
IM - Immobility
TL - Tolerance limit to produce effect in 50 percent of test organisms
NR - Not reported

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7.4.2 Chronic Effects

There are limited studies on the chronic or sublethal effects of glyphosate on aquatic invertebrates. A few studies present information on the behavioral, mutagenic, and bioconcentration characteristics of glyphosate. Except for some depression in the copulation rate of *Rivulogammus pulex*, Kikuth (1979) found no evidence of behavioral abnormalities in *R. pulex* or *Carcinogammus roesilli* exposed to a glyphosate concentration of 16 μg/L. *R. pulex*, which inhabits clean, fast-flowing streams, was considered a more sensitive species. Another behavioral trait that has been used to assess adverse effects of chemical pollution in water is invertebrate drift. Tooby (1985) found no effect on the drift of midge (*Chironomus* spp.) larvae exposed to glyphosate concentrations of 2 mg/L in artificial streams. Based on these data, it appears unlikely that adverse behavioral effects will result from the expected low concentrations of glyphosate in the water column.

Glyphosate does not appear to be mutagenic. Heydens (1991) reports normal larval development in fertilized eggs of the Atlantic oyster (*Crassostrea virginica*) exposed to concentrations of glyphosate ranging from 0.24 to 10 ppm for 48 hours. Although glyphosate does not appear to be a mutagen based on this information, additional information on the mutagenicity of glyphosate is needed to confirm this conclusion.

Bioconcentration factors have been calculated for a marine mollusk and a freshwater crayfish. The marine mollusk *Rangia cuneata* was exposed to a glyphosate concentration of 0.54 ppm for 35 days, followed by a 42-day depuration (recovery) period (Heydens 1991). The average tissue glyphosate concentration at the end of the recovery phase was 2.4 ppm. Approximately 50 percent of the accumulated glyphosate was excreted during the depuration period. The bioconcentration factor was calculated as 9.6 (Heydens 1991). (Heydens notes that a bioconcentration factor greater than 100 indicates the potential to bioaccumulate.) Heydens (1991) calculated a bioconcentration factor of 0.27 for crayfish based on a similar experiment, using an initial glyphosate concentration of 0.53 ppm, an exposure period of 28 days, and a depuration period of 44 days. Both experiments used initial glyphosate concentrations similar to the maximum glyphosate concentration (0.600 mg/L) in water observed by Henry (1992) following emergent vegetation control activities. Therefore, these bioconcentration factors appear representative of the glyphosate levels and effects that are likely to result from application

for control of loosestrife and cordgrass emergent vegetation. Based on these findings, it appears that glyphosate will not bioconcentrate or bioaccumulate in deposit feeders (e.g., mollusks) or detritivores (e.g., crabs, crayfish, and shrimp). While it appears that adverse biological affects to shellfish populations are unlikely from glyphosate applications, it is not known how residual levels, if they occur, will affect commercial shellfish production in areas such as Willapa and Padilla bays.

Considering the information presented here, it appears unlikely that application of glyphosate to control emergent aquatic vegetation will cause adverse behavioral, mutagenic, or bioaccumulation effects on aquatic invertebrates. However, additional information is needed on the potential carcinogenic, teratogenic, and reproductive effects of glyphosate.

8.0 ADDITIONAL INFORMATION NEEDS

8.1 SOIL AND SEDIMENT

Because some studies indicate that low levels of glyphosate may persist for up to a year or more, additional research is needed to quantify and evaluate glyphosate degradation and persistence in both emergent freshwater and emergent saltmarsh sediments.

8.2 WATER

Glyphosate application may contribute to increases in bioavailable nitrogen and phosphorus fractions thereby contributing to eutrophication and associated adverse effects to water quality. The potential effects of glyphosate applications on nutrient availability, particularly phosphorus, requires further study.

8.3 PLANTS

8.3.1 Submerged Aquatic Vegetation

Although available information on the toxicity of glyphosate to submerged aquatic plants suggests there is no risk to this group of plants, additional information is needed on the potential effects to ecologically important species such as eelgrass. Such additional information is likely to come from a *Spartina alterniflora* control study being conducted in Willapa Bay, Washington by the U.S. Fish and Wildlife Service and the University of Washington School of Fisheries.

8.3.2 Algae, Periphyton, and Phytoplankton

Based on the available data, it appears unlikely that glyphosate application will cause growth inhibition and other adverse chronic or acute effects to algae, periphyton, and phytoplankton. However, because of the position of these organisms at the base of the food web and their importance to marine and freshwater ecosystems, additional information is needed to confirm and support these findings. In particular, additional

information is needed on the persistence and chronic effects of repeated glyphosate treatments on these plants.

8.4 ANIMALS

8.4.1 Mammals

Significant gaps exist in the available acute toxicity information on pure glyphosate effects on mammals that are likely to inhabit or use emergent wetlands. For example, no information is available on the toxicity of glyphosate or its degradation products to raccoons, bears, mustelids (e.g., minks, weasels, and ermines), foxes, elks, beavers, muskrats, shrew, or other terrestrial or aquatic mammals. However, based on glyphosate toxicities to rats and rabbits, it appears unlikely that glyphosate will adversely affect terrestrial or aquatic mammals.

8.4.2 Birds

Additional acute toxicity information is needed on other bird groups that use wetland areas, such as perching birds and shorebirds. These birds nest and feed in emergent wetlands and may have different tolerances to glyphosate because of different metabolic rates and diets.

Also, additional information is needed on the potential mutagenic, carcinogenic, and teratogenic effects of glyphosate on birds. Furthermore, the potential bioconcentration and bioaccumulation of glyphosate in birds requires additional study. No information is currently available on these topics.

8.4.3 Fish

More information is needed to assess the potential impacts of water quality parameters, such as pH, hardness, conductivity, and temperature on glyphosate toxicity to fish. Although limited information on the chronic effects (e.g., reproductive, carcinogenic, behavioral, and physiological effects) of Roundup® formulation exists, there is little information on the chronic effects of Rodeo® or Rodeo®/nonionic surfactant mixtures to salmonids or other fishes. In addition, there is no information on the latent secondary effects of glyphosate to fish, such as immunosuppression, susceptibility to parasites or

diseases, and reduced fitness. Significant information gaps also exist on both the chronic and acute toxicity of glyphosate to all life phases of marine fish species. No information is currently available on the effects of glyphosate degradation products to fish or on the effects of repeated applications to fish.

8.4.4 Amphibians and Reptiles

There is no available information on the acute chronic effects of glyphosate to amphibians and reptiles. Because many amphibian species are dependent on wetlands for successful reproduction and juvenile development, the potential effects of glyphosate to amphibian reproduction and growth are of particular importance.

8.4.5 Invertebrates

Although glyphosate does not appear to be a mutagen based on existing toxicity data, additional research on the potential mutagenicity of glyphosate and Rodeo®/nonionic surfactant mixtures is needed to confirm this conclusion. In addition, further information is needed on the potential carcinogenic, teratogenic, and reproductive effects of glyphosate and Rodeo®/nonionic surfactant mixtures to ecologically and commercially important invertebrates such as amphipods, Dungeness crab, mussels, clams, and oysters.

8.5 GENERAL

A potential source of the variation in reported lethal and chronic toxicities of glyphosate may be varying percentages of impurities in different batches of glyphosate. Glyphosate reportedly may contain up to 0.1 ppm N-(nitrosoglyphosate), for which no toxicity information exists (USDA 1992).

9.0 SUMMARY AND CONCLUSIONS

Although glyphosate may persist at low levels for up to a year or more in soils and sediments, glyphosate generally undergoes rapid biodegradation (days to weeks), and is thus unlikely to adversely affect soil or sediment quality or soil microflora. However, there may be some risk of short-term soil or sediment instability and erosion following treatment in emergent wetlands, prior to regrowth or recolonization of treated areas by desirable vegetation. Revegetation and replanting techniques can be used to promote site stabilization and to mitigate these potential impacts.

Air quality is unlikely to be adversely affected by glyphosate application due to its low volatility.

Glyphosate application is not expected to affect surface water or groundwater quality. However, additional information is needed to determine the potential for glyphosate to contribute to cultural eutrophication in oligotrophic surface waters. Information is not available on the potential impacts of glyphosate on the sea-surface microlayer or its inhabitants.

Because it is a systemic, broadleaf plant herbicide, glyphosate is generally toxic to target and nontarget plants that it contacts. However, the phytotoxicity of glyphosate varies depending on several factors, including foliar retention, plant absorption, translocation, and timing of application.

Available information indicates that phytoplankton generally are not adversely affected by low levels of glyphosate. However, because of the importance of phytoplankton to the food web, additional information is needed to assess the potential toxic effects of glyphosate on freshwater and marine phytoplankton found in emergent wetlands. Field studies on the environmental effects of glyphosate on commercially or ecologically important Pacific Northwest species, such as eelgrass, are also needed.

Adverse impacts on nontarget plant species may result from overspray and drift during glyphosate application, but these potential impacts can be mitigated using careful application practices.

Glyphosate appears to have acute or chronic effects on animals only at very high concentrations. These concentrations are unlikely to be reached as a result of using glyphosate to control purple loosestrife and cordgrass. Therefore, it is unlikely that significant short- or long-term, acute or chronic effects on animals will result from an aquatic weed control program that uses glyphosate.

Although glyphosate appears to be nonbioaccumulative and is unlikely to adversely affect most animals, additional information is needed to assess the potential effects of glyphosate on some animal groups. For instance, no information is available on the mutagenic, carcinogenic, or teratogenic effects of glyphosate on birds. There are large information gaps on the chronic and acute effects of glyphosate on all life stages of marine fishes. Additional information is also needed on the effects of glyphosate on commercially important species, such as anadromous salmonids.

Rodeo®, the only formulation of glyphosate registered for use in aquatic environments in this state, requires the use of a surfactant. The potential impacts of surfactants on the Environment are discussed in a separate section on adjuvants.

10.0 ENVIRONMENTAL EFFECTS OF ADJUVANTS

Adjuvants are chemicals that are added to a pesticide mixture prior to application to enhance the performance of the original pesticide formulation. They encompass a wide variety of products including acidifiers, attractants, buffers, defoaming agents, deposition aids, extenders, spray colorants, spreader-stickers, surfactants, and thickeners that perform one or more of the following functions:

- Improve foliage wetting and coverage
- Reduce evaporation rate of the spray
- Improve weatherability of spray deposit
- Enhance penetration and translocation
- Adjust pH
- Improve compatibility of mixtures.

This report describes the environmental fate and effects of the following three agricultural adjuvants:

- R-11 Spreader Activator
- LI-700 Penetrating Surfactant
- Valent X-77 Spreader.

These three adjuvants are marketed primarily as surface active agents, or surfactants. Their main function is to provide an even coverage of the herbicide on the surface of the plant, although they may also improve herbicide penetration. For aquatic plants, whose foliage often has a waxy surface that resists wetting, surfactants are commonly needed to reduce surface tension and promote wetting.

Of the two herbicides being considered for control of aquatic weeds in Washington state (glyphosate and 2,4-D), surfactant use is required only for glyphosate. Glyphosate functions by disrupting photosynthesis, respiration, and the synthesis of nucleic acids in plant tissue. It must absorb through the plant foliage (i.e., leaves and stems) to be effective. Studies have shown that 20 to 30 percent of glyphosate applied to leaf vegetation is absorbed within 12 hours of application (Garnett 1991). Therefore, use of

surfactants to promote even coverage on plant foliage will improve the effectiveness of glyphosate. In comparison, 2,4-D is usually applied in a petroleum hydrocarbon carrier, such as kerosene, or is applied in granular form (e.g., Aquakleen®). Thickeners such as NalcoTrol® are sometimes used with 2,4-D at a rate of about 0.5 percent to reduce wind drift.

Pesticide adjuvants must be approved by the U.S. Environmental Protection Agency under 40 CFR 180.1001. Because chemical adjuvants are not part of the original pesticide formulation and are not considered active ingredients, they are not subject to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) requirements for pesticide registration (40 CFR 152). As a result, much of the environmental fate and toxicity information available for pesticides is not currently available for many of the adjuvants. Only limited toxicity data are available for the three adjuvants (R-11, LI-700, and X-77) of interest for aquatic weed control in Washington state. While all adjuvants are of some interest, this report, because of its limited scope, summarizes compound-specific information that is available from the manufacturers or reported in the scientific literature for three of the most widely available nonionic surfactants. Where compound-specific information is unavailable, information regarding other similar adjuvants is presented.

11.0 ENVIRONMENTAL FATE OF ADJUVANTS

This section summarizes available information on the chemical characteristics of each of the three adjuvants that affect its environmental persistence and mobility (e.g., solubility and biodegradation).

11.1 CHEMICAL CHARACTERISTICS

R-11 Spreader Activator. R-11, distributed by Wilbur-Ellis Company, contains octylphenoxypoly(ethoxyethanol), isopropyl alcohol, and compounded silicone. It is an alkylphenol ethoxylate nonionic surfactant with the following general chemical formula:

R-11 is a clear liquid with an alcohol-like odor. It is soluble in water and has a specific gravity of 1.0.

LI-700 Penetrating Surfactant. LI-700 is manufactured by Loveland Industries, Inc. It contains a mixture of phosphatidylcholine and methylacetic acid. Phosphatidylcholine, otherwise known as lecithin, is a naturally occurring complex lipid, or phosphoglyceride (Lehninger 1975). Phosphoglycerides are constituents of most plant and animal cell membranes. The phosphatidylcholine molecule contains both positive and negative charges and is referred to as zwitterionic. The chemical formula for phosphatidylcholine is shown below:

$$CH_2(R)CH(R')CH_2OPO(OH)O(CH_2)2N(OH)(CH_3)_3$$

where R and R' represent fatty acid groups.

LI-700 is a dark brown liquid with a vinegar-like odor. It is miscible in water and has a specific gravity of 1.0302. Although it is not classified as corrosive, because it contains methylacetic acid, LI-700 is often used as an acidifying agent. LI-700 is incompatible with strong oxidizers.

Valent X-77 Spreader. X-77 spreader activator, distributed by Valent U.S.A. Corporation contains alkylarylpoly(oxyethylene), glycols, free fatty acids, and isopropyl alcohol. Like R-11, X-77 is an alkylphenol ethoxylate nonionic surfactant. It is highly

volatile and flammable. Valent U.S.A. Corporation declines to reveal the chemical formulation for X-77. The generic formula for alkyarylpoly(oxyethylene) surfactants is shown below:

11.1.1 Solubility

Information concerning the solubility of the adjuvants R-11, LI-700, and X-77 is not available. Because surfactants tend to concentrate at interfaces rather than in the bulk solution, it is likely that when applied in aquatic environments for weed control, the concentration of these adjuvants would be greatest near the water surface. Both R-11 and X-77 are soluble in water (Wilbur-Ellis 1992; Valent 1991). However, LI-700, which is composed primarily of phosphatidylcholine (a zwitterionic surfactant that has an extremely low solubility in water [Tadros 1984], is likely to be less soluble than either R-11 or X-77).

Surfactant molecules are composed of hydrophilic and hydrophobic groups. The presence of the hydrophilic group provides the surfactant's water solubility. In the two nonionic alkylphenol ethoxylate surfactants (R-11 and X-77), the hydrophilic group is composed of polyoxyethylene $(CH_2CH_2O)n$ and the hydrophobic group is composed of alkylphenol. Nonionic surfactant solubility generally increases with increasing polyoxyethylene content (Kirk-Othmer 1983).

Because heating reduces the hydration of the hydrophilic portion of the surfactant molecule, solubility of the ethoxylates is inversely related to temperature (Tadros 1984). As dehydration occurs, surfactant molecules aggregate into micelles. The temperature at which micelle formation occurs, as evidenced by an increase in turbidity of the solution, is known as the cloud point. Nonionic surfactants are typically most effective at temperatures between 10 °C below and 20 °C above the cloud point (Tadros 1984).

The addition of electrolytes such as sodium chloride also reduces the solubility of nonionic surfactants (Meguro et al. 1987), which suggests that these surfactants may be less effective when used in marine environments.

11.1.2 Biodegradation

No product-specific biodegradation studies have been conducted using R-11, LI-700, or X-77. Studies using other alkylphenol ethoxylates similar to R-11 and X-77 have shown that with proper acclimation, substantial biodegradation of these surfactants can occur under both field and laboratory conditions (Swisher 1987). However, biodegradation of surfactants with alkylphenol hydrophobes (similar to the R-11 and X-77 adjuvants) containing either linear or branched alkyl groups is generally much more difficult than that of other nonionic surfactants. The degree and rate of biodegradation is influenced by several factors, including the number of ethylene oxide units per mole of surfactant, shape of the alkyl group, location of the benzene ring attachment in the alkyl chain, and temperature (Swisher 1987; Sivak et al. 1980). In general, biodegradation is more rapid for nonionic surfactants with chemical structures that exhibit less branching of the alkyl groups, fewer numbers of ethylene oxide units, and benzene ring attachment to a primary carbon in the alkyl chain (Arthur D. Little 1977; Swisher 1987).

Studies indicate that biodegradation of nonionic ethoxylate surfactants occurs via three main pathways (Swisher 1987):

- Cleavage of the hydrophilic and hydrophobic units
- Attack at the far end of the hydrophobe, with biodegradation proceeding inward
- Attack at the far end of the ethoxylate chain.

The major degradative pathway appears to be shortening of the ethoxylate chain (Swisher 1987).

The alkylphenol ethoxylates are generally considered to undergo fairly complete primary degradation (i.e., degradation to the extent that the surfactant characteristics are eliminated). However, ultimate degradation, or the complete conversion to carbon dioxide and water, is slow and may not be complete (Swisher 1987). Incomplete breakdown of the ethoxylates results in the generation of chemical intermediates, primarily polyethylene glycol (Swisher 1987). Studies have shown that polyethylene glycol is biodegradable, but breakdown occurs relatively slowly. Because its toxicity is much lower than that of its precursors (Conway et al. 1983), polyethylene glycol does not

appear to be a significant problem in the receiving environment. Additional information on the persistence of polyethylene glycol is needed to assess its environmental impact.

Short ethoxylates have also been identified as potential intermediates (Swisher 1987). Because the aquatic toxicity of these short ethoxylates is relatively high, there is some concern about potential environmental effects. However, insufficient data are available to determine whether the formation of these intermediate by-products represents a problem for aquatic use.

No information concerning the breakdown of phosphatidylcholine surfactants such as LI-700 was found during this review. Because phosphatidylcholine is a naturally occurring lipid, it is likely that this product is subject to biodegradation. The rate and degree of breakdown are unknown.

Data summarized by Swisher (1987) demonstrate that biodegradation rates and effectiveness for the octylphenol ethoxylates vary considerably (see Appendix A). Three to 100 percent reductions in surfactant concentration occurring over a period of 1 to 77 days have been reported (Swisher 1987). Most of these studies were conducted under laboratory conditions with higher concentrations of the microorganisms than might typically be present under field conditions. Therefore, it is likely that degradation may occur more slowly in the aquatic environment.

Acclimation can reduce the time for breakdown to occur. Lashen et al. (1967) studied surfactant degradation in water samples from the Schuylkill River (Delaware) and Ohio River using a tertiary octylphenol ethoxylate (t-OPE_{10}). In laboratory tests, biodegradation of the surfactant in unacclimated river samples took at least 1 week compared to several days using acclimated samples.

Lashen et al. (1966) studied the breakdown of a tertiary octylphenol ethoxylate (t-OPE_{10}) in a continuous-flow activated sludge system. The results indicate that 90 to 100 percent removal of the surfactant was achieved after 10 to 15 days of acclimation with only 65 percent removal of the polyethoxylate chain. No information is available for degradation of the phenolic compounds.

Studies also indicate that microorganisms capable of degrading surfactants may not be ubiquitous. In laboratory tests of an octylphenol ethoxylate (OPE_9) in an activated sludge

system using indigenous bacteria, Mann and Reid (1971) report that less than 20 percent of the surfactant was degraded after 46 days of operation. When activated sludge from a municipal wastewater treatment plant was added to the system, biodegradation of the surfactant increased to about 80-90 percent.

Temperature also plays a significant role in biodegradation. Mann and Reid (1971) conducted field trials on a trickling filter system and found that degradation of a tertiary octylphenol ethoxylate increased from a low of 20 percent between March and June to 80 percent between September and October, dropping back to 20 percent in December. In comparison, removal rates for linear primary alcohol ethoxylates were 80-90 percent during the winter months.

Birch (1984) also documented temperature-related changes in bacterial growth rates in a laboratory activated sludge unit. The activated sludge system was operated at temperatures ranging from 6 °C to 15 °C with sludge retention times of 2 to 10 days. At 15 °C and 2 to 10 day retention times, degradation of tertiary octylphenol ethoxylate ranged from about 90 to 95 percent. However, at 6 °C and retention times of 2 and 4 days, surfactant removal decreased to 5 percent and 50 percent, respectively. These results indicate that lower growth rates that occurred at the lower temperatures resulted in lower surfactant removal.

11.2 IMPURITIES

Because synthesis of pure nonionic surfactants is difficult, most commercial surfactants typically contain a number of impurities (Meguro et al. 1987). Impurities that are commonly present in the alkylphenol ethoxylates include (Tadros 1984):

- Polyethylene glycol (present at concentrations of 0.2 to 3 percent, depending on the molecular weight of the surfactant)

- Aldehydes (present at concentrations of 10 to 30 ppm in ethylene oxide, which is used in the production of ethoxylate surfactants)

- Inorganic catalyst residues (i.e., sodium, potassium, acetate, and phosphate, present at levels up to 0.1 percent if the ethoxylate is not filtered).

In addition, X-77 reportedly can contain trace amounts of ethylene oxide (Valent 1991). Ethylene oxide is considered to be a probable human carcinogen by the International Agency for Research on Cancer (IARC). However, because ethylene oxide is highly volatile, it is not likely to persist in the aquatic environment and probably poses a greater threat to workers handling this material than to aquatic organisms.

11.3 ENVIRONMENTAL EFFECTS

As explained earlier, registration requirements for adjuvants are not as stringent as those for pesticides. Consequently, only limited toxicity information (i.e., mammalian and aquatic organism acute toxicity) is available for most adjuvants. Available toxicity information for the three adjuvants of concern is summarized in Table 6.

11.3.1 Aquatic Toxicity

The acute toxicity of alkylphenol ethoxylates (R-11 and X-77) generally ranges from 4 to 12 mg/L for fish and from 1 to 100 mg/L for invertebrates (Sivak et al. 1980). Aquatic toxicity is affected by the length of the ethoxylate and alkyl chains on the surfactant molecule. Longer ethoxylate chain lengths result in lower toxicity, while longer alkyl chain lengths result in greater toxicity (Sivak et al. 1980).

Product-specific acute toxicity data for the three surfactants of interest, summarized in Table 1 indicate that the two nonionic alkylphenol ethoxylate products (R-11 and X-77) are generally more toxic to aquatic organisms than is the phosphatidylcholine surfactant (LI-700). In particular, available data suggest that X-77 may be highly toxic to *Daphnia magna*. LI-700 is practically nontoxic to two species of fish (rainbow trout and bluegill sunfish) and to *Daphnia magna*.

Given the typical application rates of these surfactants (i.e., 0.12 to 0.5 gallons per 100 gallons of spray solution), it is unlikely that concentrations in the receiving water environment would exceed the acute toxicity thresholds. Using conservative assumptions (1 foot water depth and maximum application rates for both the adjuvants and the herbicide, i.e., 0.94 gal/acre of glyphosate in a 0.75 percent spray solution), the estimated water column concentration of these adjuvants is approximately 0.2 to 0.3 mg/L, which is up to 3 orders of magnitude lower than the acute toxicity threshold

Table 6. Physiochemical characteristics of adjuvants used with aquatic herbicides

	R-11	LI-700	X-77
CAS Number	9036-19-5 (a)		
Manufacturer	Wilbur-Ellis	Loveland Ind.	Valent U.S.A.
Application Rate (qt/100 gal)	2	0.5-2	0.25-2
TOXICITY DATA (LC50) (Concentrations in ppm)			
Fish			
Channel catfish (g)			4.4 (b)
Rainbow trout (g)	3.8 (c)	130 (c)	4.2 (c)
Bluegill sunfish (g)	4.2 (c)	210 (c)	4.3 (c)
			5.5 (d)
Toxicity Rating	Moderately toxic	Practically nontoxic	Moderately toxic
Aquatic invertebrates			
Daphnia magna (f)	19 (c)	170 (c)	1 (b)
			2 (c)
			1.4-2.6 (e)
Hyalella azteca (g)			4-7 (e)
Nephelopsis obscura (g)			8-16 (e)
Chironomus spp. (f)			7-14 (e)
			8.6 (h)
Toxicity Rating	Slightly toxic	Practically nontoxic	Slightly toxic to highly toxic
Marine organisms			
Grass shrimp (g)			10 (b)
Toxicity Rating			Moderately toxic
Mammals			
Oral			
Rat	NA	>5000	>5000
Rabbit	5840 (i)	NA	NA
Toxicity Rating	Practically nontoxic	Practically nontoxic	Practically nontoxic
Dermal			
Rat	NA	5000	NA
Rabbit	13000 (i)	NA	>5000
Primary Irritation Index	NA	6.2	4

Blank spaces indicate data not available.

(a) For octylphenoxypoly(ethoxyethanol)
(b) Bruce 1992 pers. comm.
(c) Monsanto no date
(d) Watkins et al. 1985
(e) Henry 1992
(f) 48-hr acute toxicity tests
(g) 96-hr acute toxicity test
(h) Buhl and Faerber 1989
(i) For isopropyl alcohol

levels. However, in some cases, e.g., treatment of Spartina, concentrations may be greater.

Mitchell et al. (1987a) report that a mixture of Rodeo® (a glyphosate herbicide) and X-77 (31 percent Rodeo®, 0.4 percent X-77, and 69 percent water v:v) was more toxic to rainbow trout than Rodeo® used alone. The 96-hour LC_{50} (lethal concentration, mean) values for Rodeo® calculated in terms of the total formulation and for the isopropylamine salt of glyphosate were 1,100 mg/L and 580 mg/L, respectively. In comparison, the LC50 values for the Rodeo®/X-77 mixture were 680 mg/L and 130 mg/L, respectively, based on the total formulation and the isopropylamine salt of glyphosate. These values indicate that both Rodeo® and Rodeo® mixed with the surfactant X-77 are practically nontoxic to rainbow trout.

Henry (1992) reports similar results with Rodeo® and Rodeo®/X-77 mixtures used on *Chironomus* spp. (midge larvae), *Daphnid* (water flea), *Hyalella azteca* (scud), and *Nephelopsis obscura* (leech). Although X-77 was generally more toxic than the Rodeo® alone, no synergistic effects were observed when the two chemicals were combined. Toxic effects were simply additive, with the Rodeo®/X-77 mixture exhibiting greater toxicity to all four aquatic species than Rodeo® alone:

	LC_{50} (mg/L)		
Species	Rodeo®	Rodeo®/X-77	X-77
Midge	1,308	293	10
Water flea	545	130	2
Scud	727	213	6
Leech	1,157	201	11

These data indicate that Rodeo® and Rodeo®/X-77 mixtures are practically nontoxic to aquatic organisms while X-77 alone is moderately toxic. The Rodeo®/X-77 mixture contained Rodeo®, X-77, and Chem-Trol® (a drift retardant) in proportions of 29:1:3 by volume.

Sublethal effects on aquatic organisms have been observed with other nonionic surfactants (i.e., linear alkylbenzene sulfonates and alcohol ethoxylates) at concentrations of 0.5

mg/L (Sivak et al. 1980). These concentrations are below the acute toxicity levels for these surfactants. Sublethal effects observed in fish include impaired swimming activity, altered breathing rate, and opercular movement. Sublethal effects, such as reduced swimming activity, reduced heart rate, inhibition of siphon retraction, reduced burrowing activity, inhibition of formation of byssal threads, and intestinal damage, have also been reported in aquatic invertebrates at concentrations greater than 1 mg/L of linear alkylbenzene sulfonates and alcohol ethoxylates (Sivak et al. 1980). However, no information on sublethal effects is available for the alkylphenol ethoxylates.

11.3.2 Mammalian Toxicity

Oral and dermal acute toxicity data for the three adjuvants addressed here are summarized in Table 6. All three adjuvants are classified as practically nontoxic based on rat and rabbit studies. Although no product-specific data are available, chronic toxicity studies using other alkylphenol ethoxylates administered at doses of 1,000 mg/kg/day over a 2-year period to rats and dogs showed no significant toxicity (Sivak et al. 1980).

The Material Safety Data Sheet (MSDS) provided by the manufacturers rate both X-77 and LI-700 as corrosive based on eye and skin irritation tests in rabbits. X-77 caused irreversible destruction of ocular tissue or irritation persisting for more than 21 days. The primary irritation index (dermal) for X-77 for a 24-hour exposure is 4.0 (range 0-8).

LI-700 produced corrosion, corneal opacity, and irritation persisting for 10 days. The primary irritation index (dermal) for LI-700 was reported to be 6.2 with moderate irritation persisting 72 hours after application. Evidence of tissue damage (i.e., corrosion) was observed in all animals at 7 days. Based on these data, LI-700 was classified in Toxicity Category I for both eye and dermal application.

No eye or dermal irritation rating is provided for R-11. The MSDS states that R-11 may cause eye irritation and corneal inflammation as well as skin irritation, scaling, or dermatitis.

11.4 SUMMARY

Because adjuvants are not regulated under FIFRA, little information is available concerning the environmental fate and effects of R-11, LI-700, and X-77 surfactants. Surfactants are typically soluble in water. Based on their chemical composition, it is expected that R-11 and X-77 are more soluble than LI-700. In addition, because surfactants characteristically concentrate at interfaces, these products may be expected to accumulate primarily at the water surface and thus may affect the sea-surface microlayer.

Data for other, similar surfactants indicate that these products are subject to biodegradation. Degradation to the extent that the surfactant characteristics are no longer evident occurs fairly rapidly. However, complete breakdown of the component hydrophilic and hydrophobic units may proceed more slowly. Intermediate degradation products include the relatively low toxicity polyethylene glycol and more highly toxic short-chain ethoxylates. Insufficient data are available to assess potential impacts from these intermediates.

Acute toxicity information based on limited studies with fish and aquatic invertebrates indicates that X-77 and R-11 are more toxic than LI-700. X-77 and R-11 are moderately toxic to fish and slightly toxic to highly toxic to aquatic invertebrates. LI-700 is practically nontoxic to both fish and aquatic invertebrates. Mammalian studies indicate that these products are practically nontoxic to rats and rabbits. However, no data exist to evaluate avian toxicity of these products.

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SECTION 2

ENVIRONMENTAL EFFECTS OF 2, 4-D

1.0 INTRODUCTION

Purple loosestrife *Lythrum salicaria L.* proliferates in wetland habitats invading wet meadows, pasture wetlands, cattail marshes, stream and river banks, lake shores, irrigation ditches, drainage ditches and stormwater retention basins. It prefers moist, organically rich soils. The species crowds out native vegetation, creating dense monocultural stands that provide poor habitat for native wildlife. In producing thick stands, it chokes out waterways, slowing flow and promoting siltation. Invasion of this noxious weed, therefore, results in degradation of natural habitats for both native vegetation and wildlife, as well as causing economic burden on farmers who must keep irrigation water flowing.

2,4-Dichlorphenoxyacetic acid (2,4-D) is used for plant control, including purple loosestrife, under a wide range of environmental conditions, including upland forests, range and pasture land, cropland, wetlands, and waterway management. The application of 2,4-D to control the growth of purple loosestrife is used in wet environments, primarily along irrigation canals, drainage ditches, ponds, and lakes. It acts as a growth regulator with hormone-like activity. At relatively low dosage or applied by low-volume methods, 2,4-D can be sprayed on plant leaves and translocates to kill the plant root. At a somewhat higher dosage, the herbicide can be applied directly to the soil to be absorbed by plant root systems. Potential adverse environmental impacts may occur in 2,4-D-treated aquatic systems and in natural and agricultural areas immediately adjacent to treated areas.

2.0 ENVIRONMENTAL TOXICITY REVIEW

The U.S. Environmental Protection Agency (EPA) document *Guidance for the Reregistration of Pesticide Products Containing 2,4-D as the Active Ingredient* (September 1988a) identifies 33 herbicide formulations, including acid, inorganic salt, and ester species. This review addresses only those formulations registered for aquatic use by the Washington Department of Agriculture (1992):

2,4-D dimethylamine salt (DMA)
2,4-D butoxyethyl ester (BEE) (low volatile ester)
2,4-D isooctyl (2-octyl) ester (low volatile ester)
2,4-D isooctyl 2-ethylhexyl ester (low volatile ester)
2,4-D sodium salt
2,4-D alkyl amine salt.

Chemical structures and formulae for these 2,4-D formulations are provided in the Environmental Fate portion of this review.

The two most commonly used forms of 2,4-D are the DMA and BEE formulations. The ester formulations are generally named for the associated alcohol used in their production. The type of ester (high volatile or low volatile) also depends on the alcohol used in the formulation, with low volatile esters formulated with long-chain alcohols. High volatile esters are not registered by the Washington State Department of Agriculture for use in Washington. Both the amine and the ester formulations commonly used in aquatic applications are rapidly converted to acid form in the environment.

2.1 EFFECTS ON PLANTS

Broadleaf plants readily absorb, translocate, and metabolize 2,4-D. The herbicide is most effective during active growth periods. When applied to the leaves, it is absorbed and moves to the root system to destroy the plant (translocation). When applied to the soil, it is absorbed directly into the root. The herbicide displays optimum dosage responses, where too little is not effective and too much produces contact injury to leaves, resulting in reduced translocation to the roots. Epinastic

bending (increased growth along the upper leaf surface resulting in curling downward) may follow application within minutes. If 2,4-D is applied to the upper foliage, most of it translocates to the shoot tips. If applied to lower leaves, it moves to the roots. For perennial species, such as purple loosestrife, applications that cover basal portions of the plant are most effective (Ashton and Crafts 1973).

Biochemical reactions to 2,4-D application differ depending on dose and plant type. Trace amounts appear to stimulate growth. At higher doses, meristems (formative plant tissue) are inhibited, length growth ceases, and swollen, tumorous stems and roots are produced. 2,4-D causes abnormal growth response, affecting respiration, food reserves, and cell division. As it moves to the root system, roots lose the ability to absorb water and salts, photosynthesis is inhibited, and the phloem becomes plugged (Ashton and Crafts 1973).

Regardless of the application formula, 2,4-D appears to be transformed into the anion form within the plant. As it moves from leaf to root, the ionic form is readily absorbed by all living cells, resulting in a constantly diminishing concentration. To be most effective upon application, the plant should be in an active growth stage, in which nutrients move quickly through the system (Ashton and Crafts 1973).

The amine formulations of 2,4-D result in a water soluble form that appears to work best in foliar application on perennial weeds, due to relatively slow absorption through plant leaves that results in better translocation to roots. Amine and inorganic salt formulations also are most appropriate for soil application because roots are best adapted to absorb polar materials. The ester formulations of 2,4-D are applied in an emulsified form, the oil-water emulsion making application more efficient. Emulsified formulations move through the leaf cuticle more quickly than do polar (e.g., amine) forms and are considered best for application to annual weeds (Portman and Losey 1979; WSSA 1989). Furthermore, Portman and Losey (1979) estimate that volatile formulations are approximately 20 percent more effective than amine salts.

Appropriate application rates for 2,4-D depend on target species and the application environment. Recommendations for aquatic application are provided in Table I.

2.2 EFFECTS ON ANIMALS

Applications of 2,4-D to wetland and riparian areas may affect aquatic and terrestrial animal species. Except in a few cases, animals are not adversely affected by exposure to 2,4-D at concentrations typically used for emergent plant control (low ppm levels). The ester formulations have been shown to affect animal organisms at lower concentrations than the amine and salt formulations (refer to Biota section for more detailed discussion of the effects of 2,4-D on animals).

Bioaccumulation of 2,4-D and its formulations is reported to be low (USDA 1984). In animals, 2,4-D is absorbed almost completely after ingestion and distributed to the blood, liver, kidney, heart, lungs, and spleen, and to a lesser extent to the muscles and brain. Because 2,4-D is generally more soluble in water than in organic solvents or lipids, it does not tend to accumulate in body tissue and most is rapidly eliminated by the kidney and excreted in urine once exposure ceases (Norris 1981). The ester formulations are usually hydrolyzed to an acid form in animals and excreted (USDA 1984). Studies also indicate that animals do not metabolize 2,4-D. Most is excreted unchanged in the urine, although Grunow and Böhme (1974) reported that a minor portion of the 2,4-D administered to rats was conjugated with the amino acids glycine and taurine, and with glucuronic acid.

Khanna and Fang (1966) reported that rats fed an oral dose of 3 to 30 mg/kg of ^{14}C-2,4-D (acid) excreted 93 to 96 percent of the 2,4-D within 24 hours, primarily via urine. Peak concentrations were observed 6 to 8 hours after receiving a 1 mg/kg dose, with no trace reported after 24 hours. Zielinski and Fishbein (1967) reported that 2,4-D and its isooctyl esters administered subcutaneously to mice was excreted in the urine, with no traces of the metabolite 2,4-dichlorophenol. Similar results have been reported in rats, pigs, calves and chickens fed 50-100 mg/kg of 2,4-D in salt formulation with 2,4-D reportedly eliminated mainly in the urine

Table 1. Recommended application of 2,4-D in aquatic environments.

Herbicide	Rate	Time	Comments
		Submersed Weeds in Lakes and Ponds	
2,4-D granules	19-038 lb ae/A	Spring-early summer	Apply granules uniformly over water surface by hand or with a rotary spreader
		Emersed, Marginal and Bank Weeds Along Irrigation Ditches	
DMA	1-3 la ae/A	Spring-early summer	Use at least 100 gal/A of water carrier for ground application and at least 7.5 gal/A for aerial application.
		Broadleaf, Floating, Emersed, Marginal, and Bank Weeds of Ponds and Lakes	
DMA	1-3 lb ae/A	Spring-early summer	Use at least 100 gal/A of water carrier for ground application and at least 7.6 gal/A for aerial application.
Isooctyl Ester	0.2-6 lb ae/A	Spring-early summer	Use with water or kerosene (150-300 gal/A). Thoroughly spray all foliage.
Isococtyl 2-Ethylhexyl	2-6 lb ae/A		Apply when leaves are fully developed above water line and plants are actively growing.

Source: Burrill et al. 1992; Pesticide labels

(Erne 1966 a,b). The plasma half-life of 2,4-D in these studies was estimated at approximately 3 to 12 hours (Erne 1966 a,b). In five men who ingested 5 mg/kg 2,4-D, elimination in the urine occurred with a half-life of 17.7 hours while about 82 percent of the administered 2,4-D was excreted unchanged as 2,4-D and 12.8 percent was excreted as conjugate (Sauerhoff et al. 1977). Similar results were reported by Kohli et al. (1974) in tests conducted on 6 men given 5 mg/kg 2,4-D. More than 75 percent of the 2,4-D was excreted within 96 hours without undergoing transformation in the body.

Trace amounts of 2,4-D have been detected in the milk of lactating rats 6 days after being administered an oral dose of 0.05 mg/kg 2,4-D, but the majority of the 2,4-D was excreted in the urine (92.1 percent of the administered dose) and feces (6.1 percent) within three days (Fedorova and Belova 1974). Within 24 hours of treatment, 16.8 percent of the dose was detected in the uterus, placenta, fetus, and amniotic fluid of pregnant rats. Bjorklund and Erne (1966) also reported that 2,4-D was capable of passing the placental barrier in pigs.

Studies also indicate that 2,4-D does not bioaccumulate in fish. Like mammals, fish rapidly excrete 2,4-D after exposure ceases. Elleghausen et al. (1980) reported a depuration half-life of 1.1 days for 2,4-D (^{14}C labeled and unlabeled) in catfish exposed to an initial concentration of 0.01 ppm for four days. Bluegill sunfish given intraperitoneal injections of 1.0 and 2.5 ppm 2,4-D DMA excreted approximately 90 percent of the herbicide as 2,4-D within 6 hours of dosing.

Bioconcentration factors (BCF), defined as the net accumulation of a chemical directly from aqueous solution by an aquatic organism, have also been measured for 2,4-D. They are calculated as the ratio of the concentration in the aquatic organism to the concentration in the water. Bioconcentration factors do not take into account accumulation from other potential pathways such as food. The term bioaccumulation generally refers to the net accumulation of a chemical by an organism as a result of uptake from all routes of exposure (e.g., food, water, air).

Weber (1977) developed a rating system for evaluating the bioaccumulation potential of pesticides based on BCF values (Table 2). This rating system has been used in this report to assess numerical BCF values for 2,4-D that are reported in the literature. A detailed description of BCF values for 2,4-D in aquatic organisms is presented in later sections under each category of organism.

Table 2. Classification of pesticides based on bioaccumulation factor.

Classification	Bioaccumulation Factor[a]	Characteristics
Nonaccumulative	≤10	Pesticide readily decreases when organism is removed from exposure.
Slightly accumulative	60-700	Pesticide is only gradually lost when organism is removed from exposure, or pesticide is degraded by the organism.
Moderately accumulative	700-8,000	Pesticide is only gradually lost when organism is removed from exposure, or pesticide is slowly degraded by the organism.
Highly accumulative	≥8,000	Pesticide is not lost when organism is removed from exposure, or pesticide is not significantly degraded by the organism.

Source: Weber 1977

a As defined by author, bioaccumulation factor = concentration in aquatic organism/concentration in water.

2.3 2,4-D MIXTURES WITH OTHER HERBICIDES

Formulations of 2,4-D may act in combination with other pesticides under three scenarios: 1) applied as a mixture, 2) broadcast in a separate application (e.g., areas where pesticides are applied for mosquito and aquatic vegetation control), or 3) accidentally combined as a result of overspray in marginal areas or of runoff from neighboring areas treated with different products. Herbicide mixtures may

result in antagonistic, synergistic, additive, or unchanged effects for the individual components.

A review of studies by Grossbard and Atkinson (1985) summarized the effects of combining glyphosate with 2,4-D as a function of formulation and organism tested. In studies reviewed, Cramer and Burnside (1980), O'Sullivan (1979), and O'Sullivan and O'Donovan (1980) identified antagonistic effects. O'Sullivan (1979) reported the 2,4-D ester formulation to be particularly antagonistic to cereals. Yang Zu Nang (1978) found glyphosate and 2,4-D mixtures to be antagonistic at higher application rates on mixed weeds in Indonesia. Moshier and Russ (1978) found positive synergistic effects on young *Ampelamus albidus* (honeyvine milkweed). Other studies reported that mixtures of glyphosate and 2,4-D give "good results," are "possibly synergistic," and are "apparently compatible" (Widyanto and Soerjani 1978; Tollervey et al. 1979; India Tea Research Assn. 1979; Proctor 1975). No information is available on toxicity of combined mixtures to animals.

Shearer and Halter (1980) reviewed studies conducted on 2,4-D mixtures with dalapon (Brooker 1976); with 2,4,5-trichlorophenoxyacetic acid (Holland et al. 1960, Matida et al. 1976); and with fenac and Banvel-D (Pierce 1968), none of which identified supra-additive toxicities to aquatic organisms. No information is currently available on the effects of 2,4-D combined with Sevin® (carbaryl), endothal, or fluordane.

However, studies have shown that 2,4-D can be more persistent in the presence of other pesticides, particularly the insecticides carbaryl (i.e., Sevin®) and parathion, and the herbicide dinoseb (Smith 1989). Carbaryl and parathion are likely to be used to control mosquitos in areas that would also be treated with 2,4-D for control of aquatic vegetation. Therefore, 2,4-D residues, particularly in soil and sediment, could remain for longer periods of time in these areas. Because few studies have been conducted using pesticide combinations, it is uncertain whether toxicity and persistence of 2,4-D could be enhanced by other pesticides that might be applied for other purposes.

3.0 POTENTIAL IMPACTS TO SOIL AND SEDIMENT

The application of 2,4-D to control the growth of purple loosestrife primarily affects freshwater environments where the plant is found. Deposition of 2,4-D in areas adjoining lakes, ponds, canals, and drainages may occur depending on the mode of application (e.g., aerial spray, hand spray, pellets) and weather conditions (primarily wind). Impacts on soil in these areas depend on the type of vegetative cover present (e.g., broadleaf, grass, or barren) and the stability of the soil with respect to wind and rainfall. Movement of 2,4-D through the environment may be facilitated by attachment to soil particles that are transported by water or wind.

As broadleaf weeds are destroyed in and adjacent to waterways, soil and sediment erosion may occur depending on soil and sediment erosivity and exposure to erosive forces. Because 2,4-D is selective to broadleaf plants, other plants such as grasses will likely be unaffected by treatment. Although erosion would probably increase after treatment due to the reduction in plant cover, other vegetation that remains after 2,4-D treatment would tend to ameliorate the potential erosion impacts. Furthermore, unless physically removed after treatment, the dead vegetation would function like a mulch material to help reduce erosion until the area is revegetated with other more desirable vegetation. A worst-case scenario could occur if the area does not revegetate before the dead vegetation completely decomposes and exposes the underlying soil/sediment.

With elimination of plant species providing soil and sediment stability, physical characteristics of the soil are the primary factors affecting soil erosivity. The two most important soil characteristics affecting water-influenced erosion are infiltration capacity and structural stability. Soil texture, organic content, and clay content (i.e., swelling clays) also influence infiltration capacity (Brady 1974). Structural stability depends on the ability of soil aggregates to withstand breakup caused by physical bombardment of water and wind. This depends on many factors, including both biological (mechanical binding action of microorganisms, cementing action of the intermediate products of microbial synthesis and decay, and

cementing action of the more resistant stable humus components) and the organic/inorganic component interaction that provides bridging between organic matter and soil clays (Brady 1974).

Simplified erodibility factors (K) based purely on soil texture for different topsoil and subsoil regimes have been developed by the Soil Conservation Service (USDA 1978a). These K factors can be used as approximate erosivity estimates. The K values listed in Table 3 are used in predicting rainfall erosion losses with the universal soil loss equation (USDA 1978b) and may be used as relative indicators of erosivity across different soil texture classifications.

Table 3. Soil erodibility factors.

Surface Layer Texture	Estimated K Value
Clay, Clay loam, loam, silty clay	.32
Fine sandy loam, loamy very fine sand, sandy loam	.24
Loamy fine sand, loamy sand	.17
Sand	.15
Silt loam, silty clay loam, very fine sandy loam	.37

Source: Barfield et al. 1981

The loss of soil by wind erosion involves detachment and transport mechanisms. Detachment results from abrasion by both wind and entrained particles. Transport may cause soil particles to travel along the land surface (saltation) or to travel parallel to or upward from the land surface (suspension). Soil moisture is the primary factor in determining erosion by wind. Other soil characteristics include mechanical stability of dry soil clods and aggregates, presence of a stable soil crust, and bulk density and size of erodible soil fractions (Brady 1974). Once detached, finer-grained particles are most likely to move in wind. Unless 2,4-D-treated areas become barren or completely denuded, wind erosion is likely

to be negligible. Furthermore, revegetation of treated areas with plantings can be used to mitigate potential erosion.

Stream, ditch, and canal banks that rise steeply from the water are subject to erosion from rain, overland flow contribution, and fluctuating channel flows. Wind erosion may affect bank stability regardless of slope. Canals typically are constructed with 3:1 bank slopes and are designed to convey peak demand flows without eroding. Irrigation canals can be lined with a variety of materials including earth, blended earth (clay mix to reduce seepage loss), gravel, asphalt, concrete, or geotextile. Although vegetation may invade the channel over time, vegetation-lined channels are typically not constructed because plant growth can reduce the canal's conveyance capacity. The main objective in canal design is to minimize seepage losses from the canal and to maximize the conveyance capacity. Therefore, nuisance plant growth is actively removed by the irrigation district. Plant removal operations are usually performed at the end of the irrigation season. The general procedure involves filling and sealing the canal, after which the area is treated with a herbicide (Dickensheets 1993 personal communication). The purpose of 2,4-D application would be to restore irrigation water flow by eliminating dense stands of purple loosestrife. Because irrigation canals are typically designed to operate at capacity under unvegetated conditions, removal of nuisance purple loosestrife is unlikely to result in destabilization of irrigation canals. However, depending on site-specific conditions, erosive processes, and the amount of sediment trapped by loosestrife, removal of loosestrife may contribute to limited sediment erosion and transport.

The removal of broadleaf weeds along lake and reservoir banks may affect shoreline stability with respect to wind and wave erosion. Fluctuating water levels resulting from reservoir management expose bank soil to erosive forces. With a loss of aquatic vegetation and exposure to wind or rain, bank sediments may be redistributed along the reservoir bottom. Natural lakes experiencing less fluctuation in water stage experience a greater impact from wave action developed from wind across long open stretches and less impact from direct wind exposure. Because of 2,4-D selectivity for broadleaf plants, it is expected that resistant plant

species would take the place of purple loosestrife and provide a stable shoreline vegetative cover. However, the presence and numbers of resistant plant species would likely vary from site to site and could result in variable rates of revegetation.

Studies have shown that 2,4-D has low soil adsorption capability. Soil partition coefficients (K_d) range from about 0.3 to 1.2 in sand, sandy loam and silt loam soils (Dynamac 1988). Compounds with Kd values less than 5 are generally considered to be highly mobile in soil. High soil mobility combined with a relatively high solubility in water (300-3,000,000 mg/L) for the salt and 2,4-D acid indicate that these formulations would not be transported in soil and sediment. Adsorption on pure clays and silts is negligible (Grover and Smith 1974; Weber et al. 1965). However, higher soil organic content increases 2,4-D adsorption (Moreale and VanBladel 1980). Transport may result if organically rich sediment with adsorbed herbicide becomes mobile in flowing water systems or wind.

Although the 2,4-D BEE formulation exhibits lower water solubility (12 mg/L), studies have shown that 2,4-D rapidly hydrolyzes to the acid form in water. Therefore, 2,4-D BEE should also be transported primarily in soluble form, rather than in association with sediment. One exception may be the pellet form of 2,4-D BEE which has primarily been used to control submerged aquatic vegetation. These pellets are designed to sink to the bottom where they dissolve and release 2,4-D to the plant roots. In some cases, use of 2,4-D BEE pellets has resulted in long-term persistence of 2,4-D in hydrosoil, measured at 10 months post-treatment (Smith and Isom 1967). It is uncertain whether the increased sediment persistence was due to overapplication or poorly formulated pellets (Shearer and Halter 1980). As described by Wilkinson (1964), 2,4-D release characteristics from pellet formulations is influenced by a variety of factors, including pellet size, ambient temperature, and sediment characteristics.

The persistence of 2,4-D in sediments depends on plant uptake rates, microorganism metabolic capabilities, and water/sediment chemical characteristics. Under relatively static conditions, microbial degradation of deceased plant life may

deplete the oxygen supply to the point where aerobic conditions are no longer present. Oxygen depletion in sediments inhibits aerobic microbial decomposition processes. DeLaune and Salines (1985) reported that degradation of 2,4-D residues occurred about six times slower under anaerobic conditions compared to aerobic conditions.

Studies have shown that 2,4-D residues in soil biodegrade with half-lives on the order of days to weeks (McCall et al. 1981; Grover 1973; Smith 1972 and 1978). Biodegradation rates appear to be initially enhanced as a result of multiple herbicide applications, where microorganisms have acclimated to 2,4-D as a food source (Kaufman and Edwards 1982). Persistence and accumulation of 2,4-D in soils from repeated application have not been identified as problems (Smith et al. 1989). However, studies have shown that 2,4-D can persist in aquatic sediments. For example, half-lives as long as 50 days have been reported in flooded soils (Kuwatsuka and Miwa 1989) and residues persisting as long as 300 days after treatment have been reported in lake sediments (Dynamac 1988).

4.0 POTENTIAL IMPACTS TO AIR

2,4-D effects related to the air pathway consist of transport through volatilization as well as mechanical overspray and/or wind drift. Volatilization plays a minor role, because only the non- to low-volatile formulations may be used in the state of Washington. Salts (inorganic and amine) are least volatile, followed by acids, with esters ranking as most volatile. Volatilization allows ester formulations to enter the plant through leaf stomata. In water, the ester hydrolyzes quickly and volatilization is not a viable transport mechanism.

Overspray and drift due to wind during application are the major mechanisms for potential impact on nontarget organisms. Some product labels require use of low pressure (10-40 psi) power spray equipment. Application is permitted only when the air is calm (5 mph or less). Coarse sprays and thickening agents minimize drift.

Deleterious effects of overspray involve destruction of crops and native plants immediately adjacent to shorelines and irrigation canals. Insect, bird, and mammal populations are not likely to be affected at the concentrations used in weed applications, or in the relatively restricted areas of their use. Spray application from aircraft, in which controls necessary to minimize drift are more difficult to achieve, carries the greatest potential for impacts on nontarget species. 2,4-D spray applied to waterways can be better delivered by hand or with spray booms mounted on trucks or boats. Drift also can be reduced by using a thickener. Dynamac (1988) reported that under stable wind conditions, drift did not occur beyond 100 feet when applied at a 10-foot elevation using a 0.028 microfoil boom; under a 15-20 mph crosswind, application at 70- to 120-foot elevations resulted in a maximum 200-foot drift. Using a roadside sprayer with thickener added, little herbicide effect was found beyond 10 feet.

The air pathway also provides a route for potential nuisance impacts associated with 2,4-D application due to odor. 2,4-D is described as phenolic to odorless

(EPA 1988a), with a taste and odor threshold in water of 3.13 mg/L (EPA 1987). Although unlikely, if 2,4-D application contributes to odor problems, it may become a regulatory concern of the Washington Department of Ecology. The Ecology air quality program does not regulate herbicide use but may become involved if herbicide application interferes with enjoyment of air quality on private property through creation of odor problems (Willenburg 1992 personal communication).

5.0 POTENTIAL IMPACTS TO WATER

In aquatic environments, 2,4-D is usually ionized to the free anion through salt dissociation and ester hydrolization (Shearer and Halter 1980). 2,4-D found as a free anion is readily degraded by microorganisms under appropriate conditions (i.e., favorable pH, nutrient levels, and temperature). In general, these conditions include neutral to basic pH values (greater than or equal to 7) and relatively warm temperatures, which may be found in shallow, exposed irrigation canals and ditches, as well as shallow lakes during summer months. Sherwood (1992 personal communication) reported that the pH in irrigation ditches typically fluctuates from 6 to 10. Lows generally occur in the early morning, gradually increasing to 10 by midday in canals containing heavy algae and submersed aquatic plant growth.

The degradation rate of 2,4-D in a riverine environment increases with nutrients, sediment load, and dissolved organic carbon (Nesbitt and Watson 1980a,b). However, studies have shown that in the absence of the appropriate microorganisms, 2,4-D can persist for extended periods of time in an aquatic environment. For example, DeMarco et al. (1967) observed no change in 2,4-D concentration after a 100-day incubation period in simulated impoundments containing 50 ppb 2,4-D in sterilized, light-shielded water. In addition, no breakdown of 2,4-D was reported by Aly and Faust (1964) in lighted laboratory flasks containing 3 ppm 2,4-D in aerobic lake water after a 120-day incubation period.

In large-scale field studies of 2,4-D persistence in lakes, relative herbicide concentrations were correlated to water movement patterns. In one application, 2,4-D concentrations were measured at 0.5 to 1.5 ppm 8 hours following treatment, at 0.05 to 0.5 ppm 24 hours following treatment, and then at trace to undetectable levels from 2 weeks thereafter over the 6-month monitoring period. A second application in a more static water regime resulted in concentrations exceeding 5 ppm for 5 days, diminishing to about 0.6 ppm in 2 weeks, and

undetectable after 4 weeks (Wojtalik et al. 1971). In Lake Okanogan, British Columbia, addition of 2,4-D pellets resulted in typical residues of 0.02 to 0.05 ppm, declining to nondetectable levels 3 to 8 days following treatment (Lim and Lozoway 1979). Transport of 2,4-D was observed from cropland into an estuary environment in a pesticide runoff study conducted in Padilla Bay, Washington (Mayer and Elkins 1990). The herbicide was detected in bay water samples ranging from 0.1 to 1.3 μg/L following rain events (treatment rates were not reported).

Water column residues of 2,4-D formulations are highly dependent on application rate, dilution capacity (volume, mixing), and microbial degradation. Persistent exposure in the low ppm range could affect invertebrate populations and some fish populations, depending on specific formulation. Product label instructions typically specify application of 1 to 6 pounds acid equivalent (a.e.) per acre for liquid formulations of 2,4-D used to control floating, emersed, and marginal weeds along irrigation ditches and ponds/lakes (see Table 1). If applied to 1 acre-foot of water and assuming no interception by plant foliage, these application rates correspond to concentrations of approximately 0.4 to 2.2 mg/L of 2,4-D in the receiving water body. These concentrations are the expected maximum levels that could occur immediately after application. However, these concentrations are expected to rapidly dissipate due to mixing with water from untreated areas. In addition, biodegradation and photodegradation would also contribute to the breakdown of remaining 2,4-D residuals.

Typical application rates for granular formulations (e.g., Aqua-Kleen®, Weedtrine®) used to treat submerged plants are considerably higher (19-38 lb a.e./acre) than the rates used to control floating, emersed, and marginal weeds. The granular formulations which sink fairly rapidly, are designed to distribute the 2,4-D along the pond/lake bottom for uptake by the plant roots. Therefore, although applied at higher rates, these formulations would be expected to have little effect on water quality.

Reductions in dissolved oxygen concentrations caused by decomposing plant material could reduce water quality and adversely affect fish and other aquatic organisms. Pesticide labels for 2,4-D products prohibit application to more than one-third to one-half of a lake or pond in any one month to control oxygen depletion (for fish viability). Available information indicates that, except at very high concentrations (>100 mg/L), 2,4-D does not inhibit the growth of many microorganisms (Kim and Maier 1986; Balajee and Mahadevan 1990). Therefore it is unlikely that application of 2,4-D along the banks of wastewater lagoons would interfere with lagoon operations. However, results from a study by Mishra and Pandey (1989) indicate that 2,4-D applications for control of aquatic vegetation could stimulate the growth of toxic algae such as *Anabaena*.

Although irrigation canals are typically designed for low velocity (i.e., 1-2 feet per second), 2,4-D residues can travel long distances from application areas. Irrigation canal treatment with approximately 2 to 3 pounds acid equivalent per acre of an oil-soluble amine salt formulation (not registered for irrigation canal application) resulted in maximum concentrations of 14 to 25 ppm at distances of 0.5 to 4.25 miles downstream from the treatment site (Frank 1969).

Due to the relatively high potential mobility of 2,4-D in soils, shallow groundwater can be affected. Research has shown that residues from proper application do not tend to concentrate below the first foot of the soil profile. A review of 2,4-D contamination in groundwater identified approximately 100 detections in 1,700 samples, usually attributed to concentrated point source loading from spills or poor management (Dynamac 1988). The highest concentration found was 36.5 μg/L. The highest nonpoint source-related concentration was 4.2 μg/L, with most findings less than 1 μg/L. Two examples of 2,4-D loading directly to aquifers include discharge of rinse water from herbicide mixing tanks and application of pellets (Tardon beads) to frozen ground which subsequently washed off an adjacent hillside into an irrigation canal that passed through a ground water recharge zone (BLM 1987).

The federal drinking water standard for 2,4-D is 70 μg/L, established as a maximum contaminant level (MCL). The Model Toxics Control Act Method B establishes a risk-based cleanup level of 160 μg/L for groundwater (not specified for surface water). The 2,4-D concentrations found in groundwater during the survey cited above do not exceed either of these values. No herbicide management policy was found specifically addressing sensitive areas for wellhead protection or groundwater recharge. However, all product labels reviewed contain language alerting the user to the toxic effects of 2,4-D and describing precautions to mitigate contamination of domestic, irrigation, and animal feed sources.

6.0 POTENTIAL IMPACTS TO BIOTA

The following section provides a summary of information based on bioassay studies to determine ecotoxicological effects on plant and animal species with respect to various 2,4-D formulations. Available bioassay results identifying LC_{50} and LD_{50} values are presented for the formulations of concern. The validity and applicability of each study cited has not been evaluated during the course of this review. Additional qualitative information is also cited concerning observed physical effects and behavioral reactions to 2,4-D application in laboratory and field studies. Most of this information has been summarized at length by USDA (1984) and Shearer and Halter (1980).

6.1 PLANTS

Toxicity discussions of plants are divided between, macrophytes and phytoplankton. An extensive review of literature was performed by Shearer and Halter (1980) regarding aquatic application of 2,4-D. Shearer and Halter (1980) also provide a detailed discussion on 2,4-D effects on phytoplankton. Toxicity studies were not reviewed regarding 2,4-D toxicity to higher plants as part of this environmental effects review, although effects of unspecified formulations on nontarget species are documented.

6.1.1 Macrophytes

In general, broadleaf plants are susceptible to 2,4-D, while grasses and grains are resistant. 2,4-D is therefore toxic to many nontarget broadleaf plants, including both crops and native vegetation. Adverse effects depend on rate of application, number of applications over a given period, and relative susceptibility of individual species. Plant susceptibility to 2,4-D has been qualitatively described for a number of weeds, crops, and other terrestrial plants (Table 4).

Table 4. Plant susceptibility to 2,4-D.

Weeds - foliar application (Portman and Losey 1979)

Easily killed:

Alder*	Pigweed	Lambsquarters
Black Mustard	Dandelion	
Broadleaf Plantain*	Russian Thistle	

Usually susceptible at higher rates or with repeat applications:

Buttercup	White Clover
Canada Thistle	Yarrow

Control not practicable:

Annual Bluegrass	Reed Canary Grass*	Sandbur
Barnyardgrass	Russian Knapweed	Western Bracken
Moss	Cattails*	Wild Oat
Quackgrass	Fiddleneck (mature)	

Crops (Portman and Losey 1979)

Easily killed or injured:

Grapes	Red clover	Mint
Tomatoes	Alfalfa (young)	Sugar beets
Beans	Tree fruits	Hops
Lentils	Peppers	Strawberry
Peas and vetches	Sweet clover, crimson clover	

Less easily killed or injured:

Corn	Bulb crops
Potatoes	Flax
Asparagus (mature)	

Generally not killed or injured:

Grains (wheat, oats, barley, rye)
Grasses (annual and perennial)

Terrestrial Plants - 2,4-D Ester (BLM 1987)

Susceptible:

Willow

Susceptible to intermediate:

Cottonwood	Snowberry
Quaking Aspen	Lupine
Serviceberry	

Resistant:

Indian ricegrass	Western wheatgrass
Bluegrass	Idaho fescue
Thickspike wheatgrass	Spike fescue

* Identified with wetland environments

The use of 2,4-D as a weed killer is described on individual product labels, with appropriate mixing instructions for the specified application. If applied correctly, minimal damage would occur to nontarget species, and drift would be restricted to areas immediately bordering wetlands, ponds, and channel banks. However, because it is a nonspecific broadleaf herbicide, 2,4-D may adversely affect crops and other nontarget species exposed to overspray and drift. Potential adverse impacts on crops and other nontarget species can be mitigated by using careful application techniques, thereby minimizing drift and overspray.

The effectiveness of 2,4-D in controlling target weeds was evaluated by Hume (1988) in wheat field applications. A number of mechanisms were identified enabling susceptible species (lamb's-quarters and stinkweed) to survive over many years of 2,4-D application. These mechanisms involve timing of herbicide application relative to plant life cycle stages and considering intermittent germination and dormancy periods. Other factors include plasticity, seed dispersal, hardiness, and tolerance. Weather conditions during application also play an important role. Because residual 2,4-D may be absorbed by plant roots, it may inhibit regrowth or recolonization of treated areas by desirable vegetation prior to microbial degradation of residuals. Additional information is needed on the potential effects of 2,4-D residuals on seed germination, seedling establishment, and regrowth of desirable vegetation in treated areas.

6.1.2 Algae

Studies performed in both the field and laboratory (Tables 5 and 6) indicate that algal response to 2,4-D application varies from no effect (<1 ppm), to stimulation of growth at low concentrations (5-200 ppm), to growth inhibition (100-700 ppm), to temporary loss of species (400-1,200 ppm) (Shearer and Halter 1980; USDA 1984). It is not clear whether algal inhibition occurs as a result of cell death or lack of cell division. Laboratory results indicate that algal survival, growth, and productivity are not adversely affected at moderately high concentrations (>200 ppm) of acid and sodium salt formulations (Shearer and Halter 1980). Field observations indicate that low concentrations appear to stimulate algal growth (2

Table 5. Toxicity of 2,4-D formulations to phytoplankton: Field studies.

2,4-D Formulation	Results	Reference
DMA	2 ppm DMA increased algal production. 1 ppm did not. Water chemistry changes also noted.	Scott et al. 1978
DMA	2 ppm noted a heavy algal bloom in tank during experiment. Not present in control pool.	Stallings and Huckins 1978
DMA	20-40 lb. active ingredient per acre- 6 of 12 genera not present 24 hours post treatment but back at 2 wks. 4 of 12 genera more abundant at 24 hrs. Chlorophyll a questionable at 24 hrs. but no different at 2 wks. 14-C lower for 2 wks but of questionable significance and origin. "Plankton" retained 2,4-D residues for 6 months post treatment.	Wojtalik et al. 1971
BEE (?)	Algal growth stimulated after milfoil mortality *Chlorella* sp.	Robinson, as cited by Capp 1978
BEE	Field observations-no difference as a result of herbicide treatment, based on 19 plankton tows.	Whitney et al. 1973

Source: Shearer and Halter 1980

Table 6. Toxicity of 2,4-D formulations to phytoplankton: Laboratory studies.

2,4-D Formulation	Results	Reference
BEE	0.001-1.0 ppm - no effects on growth 4.0 ppm-10% increase in number of culture cells showing poor growth.	Butler et al. 1975
BEE	Depending on the algal strain, 36-87% of the 2,4-D was apparently metabolized to non-2,4-D compounds in 2 weeks.	Butler et al. 1975
BEE	No adverse effect on growth at 10-50 ppm, stimulation at 10 ppm. Growth inhibited by 25% at day 7 by 100 ppm. Cell pathology noted at 100 ppm but returned to normal after 7 days in clean water.	Poorman 1973
Sodium salt	No effects on algal growth at 100-300 ppm. Progressive decreases at 400-1200 ppm.	Singh 1974
Sodium salt	Cell division synchronization, cell weight, and cell diameter unaffected at all exposure levels. Respiration and photosynthesis inhibited above 244 ppm. No effect on inorganic nutrient uptake at 244 ppm.	Bertagnoli and Nadakavukaren 1974
Acid	Growth drastically at 110 and 220 ppm treatment levels in all 3 species.	Lembi and Coleridge 1975
Acid	*C. microporum*: Good growth at 5-50 ppm, none at 60-100 ppm. *B. cinnibarinus*: Fair growth at 10-40 ppm, poor at 41-70 ppm. *A. nidulans*: Good growth at 10-170 ppm, poor at 100-500 ppm. *S. marcescens*: Good growth at 50-700 ppm, poor at 900-1000 ppm. Uptake of 14C-2,4-D found to be "low", 10X reduction in residue observed after washing algal cells with water.	Voight and Lynch 1974
Acid	No effects on survival or growth at concentrations up to 200 ppm.	Morre 1974
Acid	Growth limited above 5 ppm in 1 strain of *Nostoc* sp.; growth limited above 50 ppm in 1 strain of Nostoc sp.; growth limited above 100 ppm in 17 algal strains; growth limited above 500 ppm in 3 algal strains.	Venkataraman and Rajyalakshmi 1972
Acid	No toxic effects noted at concentrations up to 200 ppm.	Vance and Smith 1969

Source: Shearer and Halter 1980

ppm) or have no effect (1 ppm). At higher concentrations, temporary loss of algal populations may occur. Recovery may take a few days to a few weeks (Pierce 1961). Because algae and phytoplankton appear to be adversely affected only at high 2,4-D concentrations, adverse effects on the food chain as a result of emergent weed control activities are unlikely.

Mishra and Pandey (1989) conducted laboratory experiments examining the effects of 2,4-D on some nitrogen-fixing cyanobacteria isolated from a rice paddy field. These investigators found that growth and nitrogen fixation of *Nostoc linckia, N. calcicola*, and an unidentified *Nostoc* species, and *Anabaena doliolum* was stimulated at concentrations up to 100 μg/mL (mg/L) 2,4-D, which is higher than the typical dose of about 40 μg/mL that would be expected to occur from the recommended field application rate for rice paddy fields. Furthermore, the authors reported lethal doses to these nitrogen fixers at very high 2,4-D concentrations ranging from 1,500 to 2,000 μg/mL (mg/L). Because the typical field dose of 2,4-D used to control weeds in rice paddies is only about 40 μg/mL, it appears unlikely that microbial nutrient cycling processes would be adversely affected by 2,4-D applications. However, to confirm this conclusion, additional information is needed on the effects of the various formulations of 2,4-D to microbes. In addition, these data suggest that 2,4-D applications to control aquatic vegetation could stimulate the occurrence of toxic algal blooms (e.g., *Anabaena*) in freshwater systems.

A BCF value of 6.8 was measured for 2,4-D acid in the algae *Chlorilla fusca* (Reinert and Rodgers 1987) which, according to Weber's (1977) system for evaluating pesticides would indicate that 2,4-D is non-accumulative.

6.2 MICROORGANISMS

Available information on the effects of 2,4-D on microorganisms is limited to a few laboratory studies. Using batch cultures of unspecified bacteria, Orhon et al. (1989) found that growth inhibition of 50 percent of the test organisms occurred at

a concentration of 400 mg/L 2,4-D. The 2,4-D formulation used in this experiment was not reported.

In an experiment using three species of ectomycorrhizal fungi, which form important symbiotic relationships with many vascular plants, facilitating nutrient uptake and improving resistance to stress (Harley and Smith 1983), Estok et al. (1989) reported significantly reduced growth rates with various concentrations of 2,4-D. They observed reduced growth rates in all three species of ectomycorrhizal fungi (*Censcoccum geophilum*, *Pisolithus tinctorius*, and *Hebeloma lonicaudum*) at concentrations $\geq$1,000 ppm 2,4-D. In addition, total growth inhibition was observed in all three species at $\geq$5,000 ppm 2,4-D. Reduced growth rates occurred in the two more sensitive species (*P. tinctorius* and *H. lonicaudum*) at concentrations $\leq$100 ppm 2,4-D, but the authors concluded that growth conditions in laboratory media generally tend to predispose the bioassay fungi to herbicide toxicity. By contrast, Ibola (1978) reported that 2,4-D had little effect on growth rates of three other species of ectomycorrhizal fungi at a concentration of 10 ppm, although growth was inhibited at higher concentrations and was completely suppressed at $\geq$1,000 ppm. The 2,4-D formulation was not reported in either of these ectomycorrhizal fungus studies.

6.3 ANIMALS

Many studies have been performed on various animal groups to measure toxicity and other impacts related to exposure to 2,4-D. The following sections summarize existing information concerning both calculated LD_{50} and LC_{50} values and observed adverse effects due to measured exposures. Animal groups are divided into mammals, birds, fish, and invertebrates. Reports of data and information are limited to the six 2,4-D formulations listed in the discussion of toxicity characteristics.

6.3.1 Mammals

Available toxicity information for two of the six 2,4-D formulations (sodium salt and acid) is presented in Table 7. Formulation-specific information was not found for the other mixtures. The ecotoxicological categories for acute oral applications (Brooks 1973) are listed in Table 8.

The studies cited in Table 7 indicate that sodium salt and acid formulations produce slight to moderate acute toxic effects on mice, rats, guinea pigs, and rabbits; mice were the most sensitive to both formulations. Dog and mule deer studies also indicate slight to moderate acute toxic effects. Lethal concentrations (LD_{50}) for these formulations of 2,4-D ranged from 100 to 1,000 ppm.

Chronic effects of 2,4-D include gastrointestinal disturbance, weight loss, muscle weakness, and loss of coordination in mammals. Some formulations cause mild eye, skin, and respiratory irritation. Formulations mixed with oil are more likely to cause skin reactions. As a dry powder, 2,4-D can affect the eye (USDA 1984). Data regarding the carcinogenicity of 2,4-D are not conclusive (EPA 1988a; IARC 1982). Several studies of laboratory animals have identified some embryotoxic and fetotoxic responses (USDA 1984). These effects include decreased birth weight, decreased litter size, or decreased fertility, often at doses that cause adult toxicity.

Chronic dietary toxicity tests with the sodium salt formulation identified no adverse effects on mice fed doses of up to 93 mg/kg daily for 3 weeks to 3 months (Bovey and Young 1980) or for rats fed doses of up to 400 mg/kg daily for 30 days, although death occurred at 500 mg/kg (Hill and Carlisle 1947).

6.3.2 Birds

Available toxicity information for four of the six 2,4-D formulations is assembled in Table 9. Toxicity rankings are provided for all bioassay results found in the literature. Studies were conducted to measure subacute dietary toxicity as well as

Table 7. Toxicity of 2,4-D formulations to mammals.

2,4-D Formulation	Organism	LD_{50} (ppm)	Toxicity Ranking	Reference
DMA	---	---	---	---
BEE	---	---	---	---
Isooctyl (2-octyl)ester	---	---	---	---
Isooctyl (2-ethylhexyl)ester	---	---	---	---
Sodium salt	Mice	375	Moderate	Hill & Carlisle 1947
		375	Moderate	Rowe & Hymas 1954
		360	Moderate	Liktionov et al. 1973
	Rats	730	Slight	Liktionov et al. 1973
		666	Slight	Hill & Carlisle 1947
		805	Slight	Rowe & Hymas 1954
		2,000	Slight	Schillinger 1960
	Guinea pigs	551	Slight	Rowe & Hymas 1954
		1,000	Slight	Hill & Carlisle 1947
	Rabbits	800	Slight	Rowe & Hymas 1954
Acid	Mice	375	Moderate	Hill & Carlisle 1947
		368	Moderate	Rowe & Hymas 1954
	Rats	375	Moderate	Rowe & Hymas 1954
		>500	Slight	McLaughlin 1951
		666	Slight	Hill & Carlisle 1947
	Guinea pigs	469	Moderate	Rowe & Hymas 1954
		1,000	Slight	Hill & Carlisle 1947
		<325	Moderate	McLaughlin 1951
	Rabbits	800	Slight	Hill & Carlisle 1947
	Dog	100	Moderate	Drill and Hiratzka 1953 in Rowe & Hymas 1954
	Mule Deer	400-800	Moderate-Slight	Tucker & Crabtree 1970 in Mullison 1981

Source: USDA 1984

Table 9. Toxicity of 2,4-D formulations to birds.

2,4-D Formulation	Organism	LD_{50} (ppm)	Toxicity Ranking	Reference
DMA[3]	Bobwhite	>5,000 (5-day)[1]	Practically nontoxic	EPA 1988b
	Japanese quail	>5,000 (5-day)[1]	Practically nontoxic	
	Ring-necked pheasant	>5,000 (5-day)[1]	Practically nontoxic	
	Mallard	>5,000 (5-day)[1]	Practically nontoxic	
BEE[3]	Bobwhite	>5,000 (5-day)[1]	Practically nontoxic	EPA 1988b
	Japanese quail	>5,000 (5-day)[1]	Practically nontoxic	
	Ring-necked pheasant	>5,000 (5-day)[1]	Practically nontoxic	
	Mallard	>5,000 (5-day)[1]	Practically nontoxic	
Isooctyl (2-octyl)ester	---	--- ---	---	
Isooctyl (2-ethylhexyl)ester	---	--- ---	---	
Sodium salt	Mallard	>2,025[2,3]	Practically nontoxic	EPA 1988b
Acid	Mallard	>1,000[2]	Practically nontoxic	Tucker & Crabtree 1970
	Pheasant	340-654[2]	Moderate	Tucker & Crabtree 1970
	Coturnix quail	530-842[2]	Slight	Tucker & Crabtree 1970
	Pigeon	530-842[2]	Slight	Tucker & Crabtree 1970
	Chuckal	200-400[2]	Moderate	Hudson et al. 1984
	Chicken	358-817[2]	Slight	Rowe & Hymas 1954
	Japanese quail	>5,000[1,3]	Practically nontoxic	EPA 1988b

Source: USDA 1984

1 Dietary (subacute)
2 Acute oral
3 Acceptable for use in hazard assessment, but does not meet EPA requirements for pesticide reregistration

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acute oral toxicity. The ecotoxicological categories for both applications are listed in Table 10.

Table 8. Ecotoxicological categories for mammals.

LC_{50}(ppm)	Toxicity Ranking
<10	Very highly toxic
10 - 50	Highly toxic
>50 - 500	Moderately toxic
>500 - 2,000	Slightly toxic
>2,000	Practically nontoxic

Source: Brooks 1973

The studies cited in Table 9 indicate that in subacute dietary toxicity studies, applications of DMA and BEE are practically nontoxic to bobwhite quail, Japanese quail, ring-necked pheasant, and mallard ducks. Data used to support the reregistration of 2,4-D base the toxicity evaluation for birds on these subacute studies. Acute oral test results provided in Table 9 indicate avian responses to 2,4-D exposure result in practically nontoxic to moderately toxic effects. Although none of these studies meet the specified requirements for pesticide registration, they are considered adequate for use in a hazard assessment and are included here to present available information on avian toxicity of 2,4-D.

6.3.3 Fish

Available toxicity information for each of the six 2,4-D formulations is assembled in Table 11 (isooctyl ester information is consolidated). Toxicity rankings are provided for those bioassays conducted over 96-hour time periods, as established by EPA. The ecotoxicological categories identified for fish are listed in Table 12. The studies cited indicate that 2,4-D formulations are practically nontoxic to highly toxic, with ester forms exhibiting the greatest toxicity. Data used to support the reregistration of 2,4-D indicate that DMA is considered practically nontoxic, BEE

Table 10. Ecotoxicological categories for birds.

LD_{50} (ppm)		
Acute Oral	Dietary	Toxicity Ranking
<10	<50	Very highly toxic
10 - 50	50 - 500	Highly toxic
>50 - 500	>500 - 1,000	Moderately toxic
>500 - 2,000	>1,000 - 5,000	Slightly toxic
>2,000	>5,000	Practically nontoxic

Source: Brooks 1973

Table 11. Toxicity of 2,4-D formulations to fish.

2,4-D Formulation	Organism	LC_{50} (96-hr) (ppm)	Toxicity Ranking	Reference
DMA	Chinook salmon	>100[1]	Practically nontoxic	EPA 1988b
	Rainbow trout	>100-420[2]	Practically nontoxic	EPA 1988b
		100	Slightly toxic	Johnson & Finley 1980
	Channel catfish	>100-395[2]	Practically nontoxic	EPA 1988b
		193	Practically nontoxic	Schultz 1973
	Fathead minnow	266-800[2]	Practically nontoxic	EPA 1988b
		320-1,530[1]	Practically nontoxic	EPA 1988b
		335	Practically nontoxic	Schultz 1973
		245-458	Practically nontoxic	Johnson & Finley 1980
	Bluegill	>100-335[2]	Practically nontoxic	EPA 1988b
		177	Practically nontoxic	Schultz 1973
		123-230	Practically nontoxic	Johnson & Finley 1980
	Smallmouth bass	236[1]	Practically nontoxic	EPA 1988b
		185-300	Practically nontoxic	Johnson & Finley 1980
BEE	Rainbow trout	0.65[2]	High	EPA 1988b
	Bluegill	0.76-1.2[2]	High-Moderate	EPA 1988b
	Fathead minnow	3.3[2]	Moderate	EPA 1988b
		2.5-4.2	Moderate	Johnson & Finley 1980
		5.6	Moderate	Mount & Stephan 1967
	Channel catfish	0.78-1.35[2]	High-Moderate	EPA 1988b
	Bleak	3.2-3.7	Moderate	Linden et al. 1979
Isooctyl ester	Cutthroat trout	>50	Slight	Woodward 1982
Isooctyl (2-ethylhexyl) ester	Rainbow trout	51-96[2]	Slight	EPA 1988b
	Bluegill	19.5[2]	Slight	EPA 1988b
Sodium salt	Mullet	32	Slight	Tag El-Din et al. 1981
Acid	Cutthroat trout	57-72	Slight	Johnson & Finley 1980
	Lake trout	35-56	Slight	Johnson & Finley 1980
		62-110[2]	Slight-Practically nontoxic	EPA 1988b
	Rainbow trout	110[2]	Practically nontoxic	EPA 1988b
	Bluegill	180[2]	Practically nontoxic	EPA 1988b
	Fathead minnow	133[1]	Practically nontoxic	EPA 1988b

Sources: USDA 1984; Shearer and Halter 1980

1 Acceptable for use in hazard assessment, but does not fulfill EPA requirements for reregistration
2 Fulfills EPA requirements for reregistration

Table 12. Ecotoxicological categories for aquatic organisms.

LC_{50}(ppm)	Toxicity Ranking
<0.1	Very highly toxic
0.1 - 1	Highly toxic
>1 - 10	Moderately toxic
>10 - 100	Slightly toxic
>100	Practically nontoxic

Source: Brooks 1973

The studies cited indicate that 2,4-D formulations are practically nontoxic to highly toxic, with ester forms exhibiting the greatest toxicity. Data used to support the reregistration of 2,4-D indicate that DMA is considered practically nontoxic, BEE is slightly to highly toxic, and the isooctyl (2-ethylhexyl) ester is slightly toxic to freshwater fish.

In a study using three water types (soft/slightly acidic, hard/alkaline, and a mixture of the two), bioassays were performed on juvenile coho and pink salmon and rainbow trout (Wan et al. 1991). Water types used were city tap (total alk 4.8 mg/L, pH 6.3), British Columbia lake (total alk 80.5 mg/L, pH 8) and a mixture (total alk 38.9 mg/L, pH 7.5). Ninety-six-hour LC_{50} results indicate different responses to DMA and isooctyl ester (with petroleum solvent plus emulsifier) and water type (Table 13). The ester formulation appears to be 3 to 20 times more toxic than the amine under all conditions.

In Washington state, hard water/higher pH conditions generally are found in eastern Washington lakes (relative to western Washington lakes). Because ester formulations appear to be more toxic to juvenile coho and pink salmon and rainbow trout in hard water environments, potential adverse effects on these species are more likely to occur in eastern Washington lakes. By contrast, DMA appears to be toxic to juvenile salmonids of these species in both soft and hard water only at much higher concentrations. It therefore appears unlikely that the use of DMA would have adverse acute effects in either eastern or western Washington lakes.

An earlier study by Wan et al. (1990) evaluated the effects of two separate 2,4-D esters (BEE and 2,4-DP), a formulated product containing both esters plus a proprietary carrier (Weedone CB with Carrier A), Weedone CB plus diesel oil, diesel oil alone, and Carrier A alone. Test conditions were similar to those of the Wan (1991) study summarized above. Results are summarized in Table 14. These data indicate all herbicide formulations and mixtures to be highly toxic to salmonids. Diesel oil and Carrier A appear to be less toxic than the herbicides, both alone and in mixtures with Weedone.

Table 13. Effects of water characteristics on 2,4-D toxicity to salmonids.

Fish Species	Dilution Water	96-hr LC_{50} (ppm)	
		DMA	Isooctyl ester
Coho salmon	Soft	472	156
	Intermediate	493	158
	Hard	662	63
Pink salmon	Soft	291	30
	Intermediate	363	70
	Hard	438	21
Rainbow trout	Soft	409	167
	Intermediate	511	164
	Hard	744	79

Source: Wan et al. 1991

Table 14. Effects of water characteristics on 2,4-D product constituents to salmonids.

		96-hr - LC_{50} (ppm)					
Fish Species	Dilution Water	BEE	DP	Weedone	Weedone+	Diesel	Carrier A
Coho salmon	Soft	1.1	1.5	10	2.2	10,299	19.8
	Intermediate	1.5	2.2	8.3	6.6	33,216	20.7
	Hard	4.3	1.8	9.0	5.6	3,333	18.5
Pink Salmon	Soft	0.4	0.8	1.7	1.2	74	19.8
	Intermediate	0.9	0.8	1.9	2.4	123	19.7
	Hard	1.1	0.6	2.1	2.2	32	17.7
Rainbow Trout	Soft	0.8	0.9	5.0	2.0	3,017	14.8
	Intermediate	1.3	1.4	3.7	5.8	2,186	20.1
	Hard	2.2	1.8	6.0	5.5	2,447	22.2

Weedone+ = Weedone plus diesel

Source: Wan et al. 1990

In this study, the softer, more acidic water appears to increase BEE toxicity to all three salmonid species (in contrast to the harder, more alkaline water with the isooctyl ester in the Wan (1991) study. The authors conclude that in soft water conditions, toxicity to juvenile salmonids increases from Weedone < Weedone plus diesel < 2,4-DP < BEE. Across the three water types, 2,4-DP and BEE toxicities cannot be considered significantly different. The addition of 10 percent diesel oil to Weedone significantly increases toxicity to salmonids only in the soft water tested.

Chronic effects of exposure to 2,4-D in fish have been observed, including behavioral and metabolic responses. At a relatively high concentration of BEE (110 ppm), impaired swimming performance was observed in green sunfish within 1 hour following exposure (Morre 1974). In a field study evaluating behavior of sunfish guarding their nests during DMA surface spray application (average 4 mg/L in the water), fish abandoned their nests in six of nine trials and in three of six trials, depending on species (Bettoli and Clark 1992). Abandonment typically occurred within 30 seconds following application and lasted on the order of minutes. Some abandonment also occurred as a result of application of water spray without herbicide. In this study behavioral response to herbicide exposure was not conclusive. Similarly, Bain et al. (1990) were unable to detect significant responses of largemouth bass to aquatic herbicides in a field study (Bettoli and Clark 1992).

A study conducted on fish ponds in India evaluated the effects of 2,4-D application (commercial grade, unknown formulation) on bottom fauna productivity and on bottom- and column-feeding fish species (Sarkar 1991). Application of 2,4-D at rates ranging from 0.5 to 10.5 kg/ha (0.45 and 2.2 lb/acre) over a 1-year period increased bottom fauna biomass (see invertebrates section below). Survival rates of each fish species (common carp, mrigal, catla, and rohu) were measured at the end of the 1-year period when ponds were drained (Table 15).

Table 15. Effects of 2,4-D on survival and yield of four species of fish reared in ponds.

2,4-D Treatment (kg/hectare/yr)	Fish Species: Rohu	Catla	Mrigal	Common Carp	Total
			Survival (%)		
Control	71.0	62.5	61.0	55.0	
0.5	71.0	64.7	60.5	58.5	
2.5	70.8	64.0	62.7	60.1	
4.5	81.0	73.4	69.8	71.5	
6.5	61.5	56.0	74.1	73.8	
8.5	50.0	54.4	75.8	75.1	
10.5	46.8	50.5	78.7	78.9	
			Yield (kg/hectare)		
Control	321.80	215.78	495.11	285.67	1,318.36
0.5	338.71	200.71	497.65	387.49	1,454.56
2.5	334.53	197.40	520.75	403.87	1,456.55
4.5	362.31	206.62	642.90	588.71	1,840.54
6.5	262.61	185.39	722.81	640.07	1,810.88
8.5	210.00	170.55	751.33	676.84	1,808.72
10.5	197.95	159.23	793.30	690.57	1,841.05

Source: Sarkar 1991

Survival rates of fish from ponds with 2,4-D application at rates of 0.5 and 2.5 kg/ha (0.45 and 2.2 of column-feeding fish (rohu and catla) decreased 13.4, 29.6, 31.3 percent and 10.4, 13.0, 19.2 percent, respectively.

lb/acre) did not significantly differ from control values. At the 4.5 kg/ha (4.0 lb/acre) application rate, survival rates of rohu, catla, mrigal, and common carp increased by 17.4, 17.4, 14.4, and 30.0 percent of control, respectively. At 6.5, 8.5, and 10.5 kg/ha (5.8, 7.6, and 9.4 lb/acre) application rates, the survival ratesSarkar (1991) concluded that there is no clear evidence of a direct influence from 2,4-D on fish growth in ponds, but the effect of 2,4-D as an activator to growth and reproduction of soil organisms is probable. Sarkar also concluded that 2,4-D may be acted upon by bottom fauna as a growth substrate. Bottom fish then consume increased bottom fauna and significantly increase yield. Yield of bottom-feeding fish (common carp and mrigal) was significantly higher than that of planktivorous fish (catla and rohu) following exposure to higher 2,4-D treatments. Sarker assumed that planktivorous fish viability was affected by phytoplankton die-off resulting from higher 2,4-D exposure (both fish species are planktivorous). Because 2,4-D appears to adversely affect phytoplankton only at higher concentrations (400 to 1,200 ppm), it is unlikely that planktivorous fish are adversely affected by 2,4-D.

Tainting of fish flesh was reportedly of great concern in the early 1960s (Shearer and Halter 1980). Of particular concern was the potential presence of 2,4 dichlorophenol (2,4-DCP) as a breakdown product and product impurity. Folmar (1979) conducted a study on rainbow trout in which three groups were exposed to 0.05, 0.09, or 0.15 ppm DMA for 4 hours, then held in clean water for 1, 4, or 7 days following exposure. Taste tests indicated that fish exposed to the highest concentration were inferior to the control fish following all recuperation periods. Other treated fish were deemed acceptable in taste but not as good as the control fish. Although 2,4-D residue concentrations were low to unmeasurable, tainting of fish flesh may have resulted from metabolic by-products.

Based on the classification scheme developed by Weber (1977) and the bioconcentration factors (BCF) compiled by Reinert and Rodgers (1987), it appears that BEE and DMA are unlikely to bioaccumulate in fish. Nonequilibrium, whole-body BCF values for BEE calculated for channel catfish and bluegill in aquaria studies ranged from 2 to 14 and 6 to 21, respectively (Rodgers and Stalling 1972). However, BCF values estimated from the octanol-water partition coefficient for 2,4-D BEE by Chiou et al. (1977) and Veith et al. (1979) are considerably higher than those measured in fish, 408 and 162, respectively. These estimated BCF values suggest that BEE is slightly accumulative (based on Weber's classification system) (see Table 1).

BCF values for other formulations of 2,4-D have not been measured in fish. However, Reinert and Rodgers (1987) report that due to its relatively high solubility in water (3,000 g/L), 2,4-D DMA would be expected to have an extremely low BCF value. A BCF of 51.2 has been estimated for 2,4-D acid based on its log-octanol water partition coefficient (Reinert and Rodgers 1987), which based on Weber's (1977) rating system, would be considered non-accumulative.

Studies have shown that 2,4-D BEE in fish is hydrolyzed to the free acid form and rapidly excreted (Rodgers and Stalling 1972). After exposure to 1 ppm 2,4-D BEE under laboratory conditions, muscle and visceral tissue of rainbow trout and bluegill contained only 2,4-D acid; no 2,4-D BEE was detected. In rainbow trout, 2,4-D concentrations in muscle tissue increased to 1.2-4.4 ppm after 3 hours, declining to 0.1 ppm within 24 hours, and were nondetectable (<0.05 ppm) after 48 hours. In bluegill, concentrations in muscle tissue peaked at 1.7 to 46.6 ppm after 2 hours, declining to 0.1-3.5 ppm within 24 hours. Unlike the rainbow trout, 2,4-D concentrations in bluegill continued to be detected after 48 hours (0.1-0.2 ppm), but decreased below detection levels after 120 hours. In separate tests in which channel catfish and bluegill were exposed to 1 ppm 2,4-D acid for 120 hours, residues in fish tissue remained below the detection limit (0.01 ppm) over the entire test period (Rodgers and Stalling 1972).

Persistence of 2,4-D residues in fish has also not been observed after field application of 2,4-D formulations for aquatic wee control. Although elevated 2,4-D concentrations have been measured shortly after treatment operations in some cases where 2,4-D BEE pellets were used for weed control, these concentrations rapidly decline after exposure ceases. Smith and Isom (1967) reported that only one fish sample out of eleven collected in a reservoir treated with 2,4-D BEE pellets contained detectable levels of 2,4-D (0.15 ppm). Residues in the remaining fish were below the analytical detection limit (0.14 ppm).

Hoeppel and Westerdahl (1983) monitored the persistence of 2,4-D DMA and BEE residuals in fish in Lake Seminole, Georgia after four 27-acre test plots were treated to control Eurasian water milfoil. Two plots were treated with 22.5 and 45 kg acid equivalent (a.e.) per ha (20 and 40 lb/acre) of 2,4-D DMA (Weedar 64®) and two were treated with similar amounts of 2,4-D BEE (granular AquaKleen®) via aerial spray. Fish samples (large mouth bass, sunfish, bluegill, catfish, and gizzard shad) were collected up to 69 days after treatment. None of the fish from the two 2,4-D DMA plots contained detectable concentrations (0.1 mg/kg) of 2,4-D in muscle tissue. However, 2,4-D was detected in 18 of the 20 gizzard shad and 4 of 24 game fish collected from the 2,4-D BEE treatment plots through day 13. The highest concentration in the shad muscle tissue (3.85 mg/kg) was observed on day 1, declining to 1 mg/kg by day 28. In game fish, the highest muscle tissue concentration (5.1 mg/kg) was observed 4 days after treatment, declining to 0.1 mg/kg by day 13.

Schultz and Gangstad (1990) studied fish uptake of 2,4-D in ponds located in Florida and Georgia that were treated with 2.24 to 8.96 kg a.e./ha 2,4-D DMA (2-8.0 lb/acre). In the Florida ponds, the highest residues (0.005-0.08 mg/kg) in fish (largemouth bass, channel catfish, bluegill, and redear sunfish) occurred 1 day after treatment. Within 3 to 7 days, there were no detectable residues in any of the fish caught. However, 14 days after treatment, one fish contained detectable residues of 2,4-D, which was believed to have been caused by release of 2,4-D from decaying vegetation. In the Georgia ponds, the highest residue of 2,4-D (0.075 mg/kg) was observed in one of three bluegills collected 14 days after the

pond was treated with 8 lb/acre 2,4-D DMA. No residues of 2,4-D were detected in fish caught from the Georgia ponds 3 and 7 days after treatment.

6.3.4 Invertebrates

Available toxicity information for five of the six 2,4-D formulations is assembled in Table 16 (no information was found for the isooctyl [2-octyl] ester form). Toxicity rankings included in Table 16 for aquatic organisms, provided for bioassays conducted over 48-hour time periods (EPA 1988b; Brooks 1973), are defined in Table 12.

The studies cited in Table 16 indicate that toxicity characteristics depend on both 2,4-D formulation and species of interest. Toxicity rankings range from practically nontoxic to highly toxic. Data used to support the reregistration of 2,4-D indicate that the DMA and BEE formulations are moderately to highly toxic and the isooctyl (2-ethylhexyl) ester is highly toxic to aquatic invertebrates (EPA 1988b). Based on suggested application rates for liquid formulations of 2,4-D, it is expected that the peak acid equivalent concentrations in 1 acre-foot of water would range between approximately 0.4 and 2.2 ppm which exceed the LC_{50} values reported for many aquatic invertebrates (see Table 16). Therefore, it appears that normal application of 2,4-D for control of nuisance aquatic vegetation may have toxic effects on aquatic invertebrates.

Effects of 2,4-D on bottom fauna of fish ponds were measured by Sarkar (1991). Commercial grade 2,4-D (unknown formulation) was applied at the rates of 0.042, 0.200, 0.375, 0.542, 0.708, and 0.875 kg/ha (0.038, 0.18, 0.33, 0.48, 0.63, and 0.78 lb/acre) as liquid, uniformly splashed over pond surfaces. Subsequent applications were made every 30 days for a total of 12 applications. During every month, bottom fauna were collected with a bottom sampler from eight areas (1 square meter each) in ponds and analyzed.

Populations of bottom organisms, measured in mass/area, were reported for total annual herbicide applications of 0.5, 2.5, 4.5, 6.5, 8.5, and 10.5 kg/ha (0.45,

Table 16. Toxicity of 2,4-D formulations to aquatic invertebrates.

2,4-D Formulation	Organism	LC_{50} (48-hr) (ppm)	Toxicity Ranking	Reference
DMA	*Gammarus fasciatus*	>100[2] (96-hr)	Practically nontoxic	EPA 1988b
	Daphnia magna	4.0->100[2]	Moderate-practically nontoxic	EPA 1988b
	Cypridopsis vidna	8.0[1]	Moderate	EPA 1988b
	Chironomus plumosus	>100[2]	Practically nontoxic	EPA 1988b
	Palaemorets kadiakensis	0.15[1]	High	EPA 1988b
BEE	*Gammarus fasciatus*	6.1[2] (96-hr)	Moderate	EPA 1988b
		0.44[2]	High	EPA 1988b
		3.1-11	Moderate-slight	Sanders 1970
		1.0-8.6 (24-hr)	Moderate	Sanders 1970
		3.1-11 (96-hr)	Moderate-slight	Sanders 1970
	Gammarus lacustris	0.44 (96-hr)	High	Sanders 1970
	Pteronarcys californica			
	nymph	1.6 (96-hr)	Moderate	Sanders & Cope 1968
	adult	>1000 (96-hr)	Practically nontoxic	Pimental 1971
	Daphnia magna	5.6	Moderate	Sanders 1970
		1.7-6.4[2]	Moderate	EPA 1988b
		4.5-9.1	Moderate	Johnson & Finley 1980
	Asellus brevicaudus	2.6[2] (96-hr)	Moderate	Johnson & Finley 1980
	Nitocra spinipes	3.1 (96-hr)	Moderate	Linden et al. 1979
	Cypridopsis vidna	2.2[2]	Moderate	EPA 1988b
	Asellus brevicaudus	1.8	Moderate	Sanders 1970
		3.6	Moderate	Sanders 1970
	Chironomus plumosus	0.39-0.79[2]	High	EPA 1988b
	Palaemorets kadiakunsis	1.4	Moderate	Sanders 1970
	Oronectes nous	100	Practically nontoxic	Sanders 1970
Isooctyl (2-octyl)ester	---	---	---	---
Isooctyl (2-ethylhexyl) ester	*Gammarus fasciatus*	1.9-3.0 (96-hr)	Moderate	Johnson & Finley 1980
	Daphnia magna	0.5[2]	High	EPA 1988b
Sodium salt	*Daphnia magna*	1042	Practically nontoxic	Presing 1981
Acid	*Cyclops vernalis*	37	Slight	Bunting & Robertson 1975
	Gammarus fasciatas	3.2	Moderate	Sanders 1970
	Daphnia magna	>100	Practically nontoxic	Sanders 1970

Sources: USDA 1984; EPA 1988b

1 Acceptable for use in hazard assessments but does not fulfill EPA requirements for reregistration
2 Fulfills EPA requirements for reregistration

2.23, 4.0, 5.8, 7.6, and 9.4 lb/acre). At 2,4-D application rates of 0.5 and 2.5 kg/ha, the mean biomass was not significantly different from the control (no herbicide application). The higher 2,4-D treatments resulted in mean biomass increases of 22.15, 20.40, 21.30, and 21.10 percent of the control values, respectively. Dominant forms of bottom fauna identified during the study included *Chironomus lobaticceps* (27.9 percent) *Branchiura sowerbyi* (17.5 percent), *Planorbis exustus* (5.3 percent), unidentified *Odonata* (4.3 percent), *Viviparus bengalensis* (5.0 percent), *Lymnaea leuteola* (6.0 percent), *Branchiodrilus hortensis* (14.0 percent), and *Chaoborus* sp. (20 percent). However, effects on species diversity from the 2,4-D applications were not reported.

Purple loosestrife has been used as a perennial source of nectar and pollen in the production of honey. Although of 2,4-D should be applied prior to the flowering stage, honeybees may become exposed; product labels suggest treatment when weeds are young and actively growing, before the bud or early bloom stage. Morton et al. (1972) evaluated subchronic oral exposure of various 2,4-D formulations to newly emerged honeybees (Table 17). Although significantly longer half-lives resulted from ingestion of 10 and 100 ppm body weight of DMA, no other significant toxic effects were noted.

Available information on the bioaccumulation of 2,4-D in aquatic invertebrates is variable. BCF values of 1 to 603 and 8,267 to 10,825 have been reported for 2,4-D BEE in zooplankton and benthic organisms, respectively (Reinert and Rodgers 1987). According to Weber's system, 2,4-D BEE would be considered slightly accumulative in zooplankton and highly accumulative in benthic organisms.

Table 17. Subchronic oral exposure effects of 2,4-D to honeybees.

2,4-D Formulation	2,4-D Concentration (ppm body wt)	Honeybee Half-life (days)
DMA	0	25.8
	10	44.4
	100	43.2
	1,000	15.6
BEE	0	39
	10	45.2
	100	48.2
	1,000	15.6
Isooctyl ester	0	30.2
	10	27.2
	100	36.4
	1,000	28.9

Source: Morton et al. 1972

7.0 ADDITIONAL INFORMATION NEEDS

7.1 SOIL AND SEDIMENT

Further study is needed to determine whether the pellet form of 2,4-D BEE persists in aquatic sediments. Although earlier studies indicated that when applied in pellet form, BEE may persist for longer periods of time, it was unclear whether this was caused by overapplication or poorly formed pellets.

7.2 WATER

Decomposition of affected vegetation after 2,4-D application may result in increased bioavailable nitrogen and phosphorus. Increases in nutrients could adversely affect water quality and contribute to increased eutrophication. These potential effects require additional investigation.

7.3 PLANTS

7.3.1 Algae

Available information indicate that at low concentrations, 2,4-D can stimulate the growth of cyanobacteria and algae. Further investigation is required to determine whether application of 2,4-D to control aquatic vegetation could stimulate algal blooms.

7.4 ANIMALS

A number of additional studies are required by EPA for the reregistration of 2,4-D as summarized below (see Appendix 2; HEC and Ebasco 1992 for additional detail):

- Avian acute oral and subacute dietary toxicity (2,4-D acid and inorganic salts)

- Freshwater fish and invertebrate toxicity, fish early life stage and aquatic invertebrate life-cycle (2,4-D acid, inorganic salts, amines, and esters) [Note: Of particular concern is the available information indicating that typical field application rates would result in invertebrate mortality.]

- Acute toxicity (oral, dermal, and inhalation) in various animal species (2,4-D amine and ester formulations).

- Eye and dermal irritation (2,4-D acid, amine, and ester); dermal sensitization (2,4-D amine and ester).

- Subchronic, 90-day feeding in rodents and non-rodents (2,4-D amine and ester) and chronic toxicity for non-rodents (2,4-D acid).

- 21-day dermal toxicity (2,4-D acid, amine, and ester)

- Teratogenicity in rats (2,4-D amine and ester) and rabbits (2,4-D acid, amine, and ester).

- Mutagenicity testing (2,4-D acid, amine, and ester).

- General metabolism and dermal neurotoxicity (2,4-D acid, amine, and ester).

- Nontarget area phytotoxicity studies (seed germination, aquatic plant growth, and vegetative vigor).

- Bioaccumulation (crops, fish, and aquatic organisms).

8.0 SUMMARY AND CONCLUSIONS

Formulations of 2,4-D may be applied either as a liquid spray or as a solid matrix incorporated into the soil. Applications to purple loosestrife are normally performed as an aerial spray by hand or vehicle such as boat, truck, or helicopter equipped with a spray boom. Subsequent movement of 2,4-D through the environment depends on wind conditions and the flow of water in the receiving body.

In the state of Washington, only low-volatile 2,4-D formulations are permitted for use, which limits exposure to nontarget species in close proximity to application areas. Overspray can be minimized by using appropriate spray nozzle diameters, adding thickeners, spraying close to the ground, and restricting application to periods when winds are low .

2,4-D biodegrades in soil, sediment, and water and is generally not considered persistent in aerobic environments. Half-lives in soil and water generally range from several days to several weeks. Accumulation of 2,4-D has not been observed in soils receiving repeat exposures at typical field application rates (Smith et al. 1989). Available data indicates that half-lives in soils previously exposed to 2,4-D are lower than in previously untreated soils due to acclimation of soil microorganisms. Similarly, in laboratory studies, the initial lag time prior to the onset of biodegradation is typically shorter in cultures that have received prior exposure to 2,4-D (Aly and Faust 1964). However, studies have shown that 2,4-D can persist in aquatic sediments, with half-lives as long as 50 days reported in flooded soils (Kuwatsuka and Miwa 1989) and residues persisting as long as 300 days in lake sediments (Dynamac 1988). Earlier studies suggest that sediment persistence may be associated with the use of 2,4-D BEE pellet formulations, which sink to the bottom after application and slowly dissolve, releasing 2,4-D over a longer period of time. However, it is unclear whether the longer persistence observed in these earlier studies was caused by improper application or poorly formulated pellets (Shearer and Halter 1980).

Transport of 2,4-D away from application areas may be facilitated by attachment to soil or sediment particles that are subsequently moved by wind or water. 2,4-D exhibits relatively low soil partition coefficients (0.3 to 1.2) with sand and silty loam which indicates that it would not tend to bind to these soils. However, studies have shown that the presence of soil organic matter enhances 2,4-D affinity to the matrix. In freshwater environments where 2,4-D is likely to be applied for purple loosestrife eradication, water-mediated transport will likely result in higher potential impacts on nontarget species, since wet sediments are not easily transported by wind. 2,4-D may be distributed in water systems either by advective forces (open-channel flow) or by diffusion. Studies performed in irrigation canals have identified herbicide up to 4 miles downstream of application areas. Advective movement in lakes also has been observed. Concentration of 2,4-D in the receiving water body depends on application rate, mixing capacity, and time allowed for diffusion.

Leaching of 2,4-D to groundwater as a result of normal surface application was not found; instances of groundwater contamination have been associated with spills or unusual circumstances (e.g., 2,4-D BEE pellets inappropriately applied to frozen soil washing off a hillside into an irrigation canal that intercepted a groundwater recharge area.

In general, broadleaf plants are susceptible to 2,4-D, while grasses and grains are resistant. Adverse effects on nontarget species depend on rate of application, number of applications over a given period, and relative susceptibility of individual species. Application timing relative to plant life cycle stages also plays an important role in determining herbicide effectiveness. Because the presence and numbers of resistant species will likely vary from site to site, the rate of revegetation after 2,4-D application is also expected to be highly variable.

Studies indicate that application of 2,4-D at low concentrations stimulates growth of phytoplankton, that no adverse affects are common at concentrations resulting from normal application rates, and that temporary loss of species occurs at high concentrations. Typical field application rates of 2,4-D can also stimulate algal

growth and could result in algal blooms. Of particular concern, is the potential to cause blooms of toxic algae such as *Anabaena*.

The impact of 2,4-D on animals appears to depend on formulation. The ester forms exhibit moderate to high toxicity toward aquatic invertebrates and some species of fish. The toxic thresholds for many invertebrates are lower than the concentrations that would be expected from typical field application rates, which indicates that use of 2,4-D to control aquatic vegetation may cause invertebrate mortality. The acid and salt formulations are practically nontoxic to moderately toxic toward the same animal groups. Limited available information on toxicity of 2,4-D to birds and mammals indicates moderate to practically nontoxic effects. The concentrations required to produce mortality in terrestrial animals are higher than those typically resulting from overspray in aquatic applications.

Available information indicates that the bioaccumulation potential for 2,4-D in mammals and fish is low. The acid and salt formulations (e.g., sodium and dimethylamine) of 2,4-D are soluble in water (300-3,000,000 mg/L; Reinert and Rodgers 1987) and exhibit relatively low octanol-water partition coefficients (479-645 for acid, and reported as low for DMA; Reinert and Rodgers 1987) which indicates that these compounds would not tend to accumulate in tissue. The bioconcentration factor (BCF) estimated based on the chemical characteristics of the free acid form is fairly low (51.2) and for DMA is expected to be extremely low (Reinert and Rodgers 1987). Furthermore, studies indicate that animals do not metabolize 2,4-D and excrete almost the entire applied dose in urine within 24 hours of exposure (USDA 1984).

Studies have shown that 2,4-D BEE is rapidly hydrolyzed to the free acid form and is also rapidly excreted in urine. Measured BCF values for BEE in fish range from about 2 to 21 (Reinert and Rodgers 1987). Although estimated values are somewhat higher (162-408) (Chiou et al. 1977; Veith et al. 1979), studies with rainbow trout and bluegill indicate that muscle tissue concentrations decrease to below detection limits within about 48-120 hours of exposure (Rodgers and Stalling 1972).

Data on the uptake of 2,4-D in other species is limited. However, available data for 2,4-D BEE indicate that 2,4-D can accumulate in zooplankton and benthic organisms, with BCF values of 1 to 603 and 8,267 to 10,825 reported for zooplankton and benthic organisms, respectively (Reinert and Rodgers 1987). Bioaccumulation of 2,4-D in algae appears to be low, with a reported BCF value of 6.8 in *Chlorilla fusca* (Reinert and Rodgers 1987).

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SECTION 3

PUBLIC PERCEPTION OF PESTICIDES

1.0 PUBLIC PERCEPTION OF PESTICIDES

Public perception of pesticides varies widely depending on the particular group or constituency. People in general are cognizant of the contributions of pesticides to the control of insect borne diseases and to the widespread year-round availability of vegetables and fruits. However, people are also aware of reports of pesticides in groundwater and surface water and of the controversies regarding the safety of pesticide residues in food.

A survey conducted by Environment U.S.A. (Environment U.S.A. 1992) reveals that two-thirds of Americans believe that the amount of pesticide used in agriculture today is certain to be harmful to the environment. In addition, 62% of Americans see the agricultural industry as environmentally careless. More than half the population is so concerned with the environmental damage resulting from agricultural chemicals that they are willing to support an outright ban on home and garden use. Water pollution in general was considered to be one of the highest environmental priorities by those surveyed, surpassed only by concern over air pollution.

2.0 PUBLIC REACTION TO USE OF HERBICIDES IN AQUATIC SYSTEMS

Education of the public may be an effective tool in reducing negative perception and in increasing the acceptance of herbicide use in aquatic systems, particularly for noxious plant management. In Florida, the use of Rodeo® to control aquatic weeds initially caused concern for several interest groups, specifically fishermen and homeowners with properties near the areas targeted for spraying. However, when the interest groups were informed about the problems associated with non-native plants and their negative impacts on diverse habitat, and were provided the technical data on glyphosate environmental fate and toxicity, many of their concerns were alleviated (Lisa Drake, personal communication). The key to a successful aquatic weed management program is to specifically inform people prior to initiation of any plan incorporating the use of herbicides to control noxious weeds and to effectively communicate the invasive plant control issue.

Another area in which education of the public can influence public perception is in regulatory testing and registration of pesticides. A better understanding of the regulatory system can help dispel the myth that the chemical industry is able to manufacture and sell pesticides at will. Most citizens do not have knowledge about the regulatory process for pesticide registration and the extensive toxicological, chemical, and environmental testing requirements. On average, only one in 20,000 chemicals makes it from the laboratory to the marketplace (Miller, personal communication) and the length of time required to register a new pesticide in the U.S. can be as long as ten years. More than 100 different studies must be performed to demonstrate to the EPA that no adverse risk to human health or the environment will result from application of a new pesticide. This rigorous testing and federal registration process is supplemented by the individual state systems of pesticide regulation. All these regulatory agencies work to ensure compliance with existing laws by pesticide producers and users.

Explaining to the public the concept of Integrated Pest Management (IPM) and the value of chemical treatment as a component of that approach is also important. The Chemical Manufacturers Association and all major pesticide producers support IPM and the concept of using pesticides only when necessary to address specific problems.

3.0 RESPONSE OF PUBLIC TO HERBICIDES IN DRINKING WATER

From time to time the media report that traces of pesticides have been detected in waters that provide the drinking water supply to municipalities. Public reaction to this information is typically one of concern: Is there a risk to human health from trace levels of pesticides in drinking water? While the answer to this question may be "no" based upon compliance with scientifically derived guidelines such as the Maximum Contaminant Level (MCL) established by EPA, communicating the level of risk to the public can be difficult. The MCL is the level that is considered protective of a person who might drink water at that level of contamination every day for a lifetime.

Education of the public regarding the research which has been conducted in the last several years to evaluate the nation's surface water and groundwater is an important component of the risk communication process. EPA and several of the major pesticide producers have independently sponsored large studies of the nation's groundwater. These studies have shown that fewer than 0.2 percent of the nation's wells had pesticide traces above the MCL (Monsanto 1991). Many of the problem wells were traced to specific incidents at the mixing or loading site, or to poorly constructed wells.

Surveys of surface water do show seasonal peaks, due to the run-off of pesticides into surface water during spring planting seasons. The pesticides typically appear in peak concentrations and quickly dissipate. Health impacts due to these seasonal peaks have been addressed by EPA in establishing a health advisory level at which water is considered safe to drink for up to 10 days. For long term consumption, a water source is considered safe to drink if the average daily contaminant concentrations for a year do not exceed the MCL or lifetime health advisory (Monsanto 1991).

Since July 1992, providers of public drinking water are required to monitor for organic synthetic chemicals, including pesticides such as glyphosate and 2,4-D. Even though glyphosate has a low toxicity to humans, as discussed in the glyphosate risk assessment, because it is one of the few herbicides approved by EPA for controlling weeds in delicate aquatic environments, the EPA has established an MCL of 700 parts per billion for glyphosate (Monsanto 1991). An MCL of 70 ppb has also been established by EPA for

2,4-D (USEPA 1992). The potential risk to human health from exposure to 2,4-D is discussed in detail in the 2,4-D risk assessment.

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