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(English translation)

**IMPACT OF HERBICIDES ON THE FOREST ECOSYSTEM,
AQUATIC ECOSYSTEMS AND WILDLIFE: THE AMERICAN
EXPERIENCE**

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The impact of forestry herbicides on ecosystems, terrestrial and aquatic, is a subject which has received much attention. That attention increased with the publication of Rachel Carson's book *Silent Spring* in 1962. At that time, several chlorinated hydrocarbons were in wide spread use in the US and around the world, the most well known of which is DDT. Although many of the claims in Carson's book are still highly controversial, it is clear that DDT posed a threat to constituents of some ecosystems. While DDT was banned for use in the US in 1972, its beneficial uses for protecting human health were recognized and it even has a place in the world arena today. According to the World Health Organization, its beneficial uses in controlling disease vectors far outweigh any human health risks (W.H.O., 1998) and the decision to use DDT until an acceptable replacement can be found has been made, even in the face of considerable opposition due to its known or suspected environmental impacts.

The DDT story has scared many people, but there are good lessons to be learned. The first is that science is not perfect. The second is that the vigilance of science as applied in the corporate, private, public, educational institution, and governmental sectors is essential in assuring safety of every technology that affects humans and the environment in which they live. It is not possible for any individual segment of the research society to test every factor or combination of factors to determine that any new technology is safe. Indeed it is not possible under tenets of science to 'prove' anything. In the face of this conundrum 'how can we be sure of the safety of technology when our science is unable to prove the safety,' we must find alternative methods of making decisions about new scientific developments and the role they will play in society. Currently the most widely accepted alternative is risk assessment.

Risk assessment recognizes and incorporates uncertainty into the process of evaluation and allows for the management of uncertainty in decision making. Risk assessment in the United States utilizes a tiered approach. Each tier may generate additional researchable questions and point to potential problems. At the conclusion of each tier more research may be required and more information provided.

Most of the information used in consideration of new pesticides comes from mandatory laboratory and field testing of prospective chemicals which begins prior to initial registration by the US Environmental Protection Agency (EPA). The EPA requires sufficient data to allow human health and ecological risk assessments prior to registration.

Human health risk assessments are based on extensive animal testing to determine acute, chronic, and subchronic toxicological endpoints for reproduction, teratogenicity, carcinogenicity and mutagenicity. Oral, skin absorption, eye contamination, and inhalation routes of exposure are considered as appropriate for these toxicological tests. Ecological risk assessments consider the environmental fate and impacts of forest herbicides on terrestrial and aquatic ecosystems and are based on tests on a host of terrestrial mammals, birds, insects, fish and aquatic invertebrates. The registration process requires laboratory and field tests conducted with pure or technical grade of the active ingredient (ai) and with the end-use product, i.e. the commercially formulated product. Field dissipation studies also require data from the end-use product for soils, aquatic and forest ecosystems. Aerobic and anaerobic soil and aquatic metabolism studies are required of all active ingredients and where metabolites can pose a problem, toxicity testing is required. Fate and impacts depend on many factors including patterns of use, chemical properties, persistence, mode of action, and toxicity of each herbicide. Modern forestry herbicides are characterized by chemical and biological properties which result in their relative safety in the environment compared to pesticides used in other suburban, urban and agricultural settings.

Pesticide Use Patterns and the US Land Base

Approximately 2.1 billion kg of pesticide active ingredient are used in the US annually. Of these, 442 million kg are conventional pesticides composed of 890 active ingredients registered by the EPA and formulated into approximately 20,700 registered and commercially available products. The remainder (1658 million kg) are classified as non-conventional pesticides and include such categories as wood preservatives (303 million kg), specialty biocides (119 million kg) used in swimming pools, spas, industrial water treatment, disinfectants, sanitizers and the largest group the chlorines/hypochlorites (1083 million kg) used mainly in the disinfection of potable and waste water (Aspelin and Grube 1999) and other chemicals used in a variety of ways. Conventional pesticides are used with varying intensities on most of the US land base (approximately 936 million hectares, mm ha), but the most intensive use is on household land which includes uses on and in dwellings and the lawns surrounding them (Figure 1). In addition, herbicides used in forest management are applied once or twice over a rotation (20-80 years depending on

the species and end product) while application to crops in agriculture and around homes occurs many times each year. In this paper, we will focus on a small portion of conventional pesticides, limited to the herbicides used in forest management.

Sixty-three percent of the 302 million ha of forest land in the US are under private ownership while 77 million ha are managed for the people in the National Forest System (NFS). The remaining land is managed by the National Park System, Bureau of Land Management, Bureau of Indian Affairs, and State and local governments. Of the 890 registered active ingredients, 20 account for more than 95% of the pesticide used in forest vegetation management. Forest vegetation management, in the broader context, includes such activities as plant protection from animal, insect, bacterial, and fungal damage. It also includes noxious weed control, conifer and hardwood culture, and improvement of recreational areas and wildlife habitat. It is difficult to determine exactly how much of each kind of pesticide is used in forest management in production forests because of the proprietary nature of that information. It is clear that pesticide use, especially herbicide

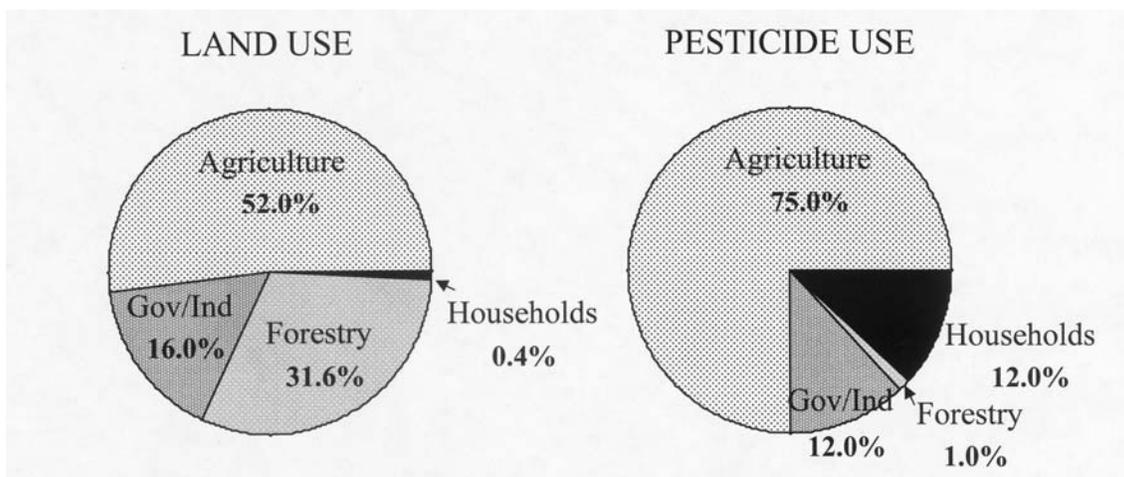


Figure 1. Land and pesticide use patterns in the United States of America. While most of the 442 million kg of conventional pesticide active ingredients used in the US annually are used on agricultural land, the most intensive use pattern is around households while the least intensive use is found in forestry (Aspelin and Grube, 1999; Pimentel and Levitan, 1986; Short and Colborn, 1999).

use, is more common on production forests than on NFS land. In the 12 southern states, herbicide use on private forests increased 53% from 1996 to 1998 and a total of 256,345 ha were reported treated with herbicides in 1998 (Dubois et. al., 1999). For that same time period only 48,169 ha of NFS land nation-wide were treated with herbicides (USFS, 1998).

Chemical Properties Affecting Herbicide Environmental Fate

The chemical properties of herbicides interact with environmental and site specific conditions to determine persistence, movement on- and off-site, and potential for bioaccumulation. Each of these aspects of environmental fate are components that

contribute to the probability of exposure for every organism. Among the most important properties are K_d , K_{ow} , water solubility, hydrolytic and photolytic degradation, and reduction-oxidation reactions.

K_d , the soil-water partition coefficient, is a measure of the potential for a herbicide to be preferentially sorbed (absorbed or adsorbed) from aqueous solution by soil particles. It is frequently, but not always, related to the amount of organic carbon in the soil (Wauchope et al. 1992). K_d is measured directly for each soil, but there have been many attempts to eliminate the need for direct measurement. Most commonly one finds the K_{oc} listed for pesticides. The K_{oc} is a number which derives from the K_d through the assumption that organic matter is solely responsible for pesticide adsorption to soil particles and is often 'predicted' solely on the basis of solubility. It is a useful number provided accuracy within an order of magnitude is acceptable. Because this assumption does not hold for polar, ionizable, highly water soluble herbicides we present the K_d instead as a more accurate estimate of each herbicide's affinity for soil. Therefore, because of the many different soil types around the world, K_d values in Table 1 are frequently expressed as a range of values. On others, data is limiting and so single values are presented.

Large K_d values indicate a strong tendency to sorb onto soil particles. Strongly sorbed herbicides are less available for movement off-site in storm runoff except where sorbed onto eroded soil particles. These chemicals are also generally less available for degradation and volatility losses than chemicals with small K_d values. The large K_d values for fluazifop-p-butyl, glyphosate and oxyflurofen (Table 1) indicate they are strongly held onto soil particles and are relatively immobile once they come in contact with soil.

Table 1. Chemical properties of some herbicides.

Herbicide	K_d	K_{ow}	Aqueous Solubility (mg/L)**	Photolysis	Hydrolysis
Asulam*	0.24-1.0	0.31-1.8	5000	Yes	Yes
2,4-D	0.14-3.38	2.81	620	Yes	Yes
Dalapon*	1	6	900000	Yes	Yes
Dichlobenil*	0.295-2.098	3.06	18	Yes	No
Dichlorprop*	2	1000	350	No	No
Fluazifop-p-butyl*	67	31622	2	No	Yes
Fosamine ammonium	0.095	0.00125	1790000	No	No
Glyphosate	62-175	0.0017	12000	No	No
Hexazinone	0.24-10.8	14.79	33000	Yes	No
Imazapyr	0.06-3.02	1.3	15000	Yes	No
Metsulfuron	1.4	0.01-1.0	9500	Yes	Yes
Oxyflurofen*	1160	29400	0.1	Yes	No
Propyzamide*	3.2-10.1	1568	15	Yes	No
Quizalofop ethyl*	6.2	15849	0.31	---	---
Sulfometuron	0.71	0.31	244	Yes	Yes
Triclopyr	0.165-0.975	<5 (TEA)	435	Yes	No

*Not registered for general forestry in US.

** Water solubility at 20 or 25 degrees C and pH 7 where pH is significant. Data are from Grover (1977); Hay (1990); Kidd and James (1991); Pesticide Information Profiles (PIPs, <http://ace.orst.edu/info/extoxnet/pips/searchindex.html>) a cooperative effort by University of California Davis, Oregon State University, Michigan State University, and Cornell University; SERA (1999); USDA-ARS Pesticide Properties Database online (<http://wizard.arsusda.gov/acsl/ppdb2.html>); and US Environmental Protection Agency's Reregistration Eligibility Decision (RED) documents.

K_{ow} , the octanol-water partition coefficient, is another measure of the hydrophobicity of a herbicide. Large values for K_{ow} indicate a potential for storage in fatty tissues, and therefore a greater potential for bioaccumulation. Bioaccumulation is the increase in concentration of a chemical in an organism resulting from tissue absorption levels exceeding the rate of metabolism and excretion. In general, persistent pesticides with K_{ow} values greater than 1000 have a potential for bioaccumulation. Oxyfluorfen has a low to moderate potential for bioaccumulation in bluegill sunfish (bioconcentration factor of 1300) and channel catfish (bioconcentration factor up to 5000) following a 30-40 day exposure in solutions containing 10 ppb oxyfluorfen (Exttoxnet, 1996c). Hexazinone, with a K_{ow} of 14.79 does not bioaccumulate. Fluazifop also has a very large K_{ow} , but with its low water solubility, rapid hydrolysis and half-life of approximately a week, bioaccumulation is not likely to be a problem.

Solubility may affect persistence and movement of herbicides on treated sites, but normally solubility is not a limiting factor when it exceeds 1 mg/L. Solubility and K_{ow} changes for some herbicides as a function of pH. The sulfonylureas, sulfometuron and metsulfuron, increase in solubility while K_{ow} decreases with increases in pH. These increases can be significant. For example sulfometuron solubility in water is 6.4 mg/L at pH 5 and 25° C, but solubility increases to 244 mg/L at pH 7. Thus at pHs common in forests, solubility of the herbicides in Table 1 is such that all but oxyfluorfen and quizalofop ethyl could be mobile in forest soils. Using the GLEAMS model, Michael et al. (1996) have postulated the off-site movement of imazapyr, hexazinone and triclopyr to be very similar if applied on the same soils. Soil partition coefficients, however, indicate that movement of fluazifop-p-butyl, glyphosate, and in some soils hexazinone and pronamide is limited by sorption to soil particles.

Persistence and movement off-site are further limited by hydrolysis and photolysis of many herbicides (Table 1). Rapid hydrolysis significantly limits the persistence of the butoxy ethyl ester and the triethyl amine formulations of triclopyr reducing them quickly to the triclopyr acid in a matter of hours to less than 2 days. Triclopyr acid is then further degraded by photolysis while hydrolysis does not affect the acid form. Photolysis and hydrolysis are also important considerations when water samples are being collected for residue monitoring. In most cases it is best to keep all samples cool and protected from light. In the case of the sulfonylureas the rate of hydrolysis decreases with increasing pH, so it is necessary on sites with water at acid pH to stabilize the sample solution by adjusting pH to neutral or above.

All of these chemical properties play a significant role in the environmental fate of the herbicides listed in Table 1. Coupled with biotic factors including microbial activity, the persistence of these chemicals is significantly reduced below that of the older chlorinated hydrocarbons so vilanized by the public following the publication of Carson's book. In fact soil half-lives for these compounds is relatively short under field conditions. I have summarized the available half-life data from around the world for the most popular forestry herbicides in the US. The average half-lives in soil are glyphosate, 29 days; hexazinone, 88; imazapyr, 46; metsulfuron, 42; sulfometuron. 26; and triclopyr; 99. These averages are for data from a very large variety of soil and site conditions around the world.

Mode of Action

Pesticide toxicity, especially herbicide toxicity, is an issue of concern for most people. However, pesticides are not the only toxic chemicals with which we come in contact on a daily basis. All chemicals, natural or man-made, are toxic at some level of exposure. The difference between acute and chronic toxicity versus the no observed effect level (NOEL) is primarily a function of the amount of exposure in a unit of time and the mode of action of the chemical. For example, vitamin D is essential to good health and mammals consume it on a daily basis. However it can be very toxic, in fact more toxic than most of the herbicides used in forest management. The human acute oral LD₅₀ (the amount required to kill half the test animals) for vitamin D is 10 mg⁻¹ kg⁻¹ day⁻¹ (Ottoboni 1984) while the acute oral LD₅₀ for 2,4-D is 300 mg⁻¹ kg⁻¹ day⁻¹. Nicotine, found in most tobacco products and also used as an insecticide, is listed as highly toxic with an acute oral LD₅₀ of less than 5 mg⁻¹ kg⁻¹ day⁻¹.

Most herbicides are effective and useful because of their mode of action. The mode of action makes it possible to kill susceptible plants at relatively low rates of application while much higher rates would be required to kill animals. Mode of action also permits selectivity among plant species. The principal reason most herbicides are relatively nontoxic to mammals, birds, fish, insects, and aquatic non-plant species is the mode of action.

There are several metabolic systems in plants which do not exist in animals and it is these systems which are most often the target for individual herbicide mode of action. A large number of forestry herbicides are grouped together as plant growth regulators (PGRs). Included in this group are asulam; 2,4-D; dalapon; dichlobenil; fosamine ammonium; pronamide; and triclopyr. PGRs can act at multiple sites in a plant to interrupt hormone balance especially by mimicking or inhibiting the plant growth hormone indoleacetic acid (IAA), inhibiting protein synthesis in order to inhibit leaf bud initiation, shoot and root growth, and as in the case of triclopyr causing uncontrolled cell growth in plants.

Fluazifop-p-butyl and quizalofop ethyl inhibit lipid synthesis by preventing the formation of fatty acids essential for production of lipids. Lipids are essential to the formation and function of cell membranes and new cell growth. These inhibitors of fatty acid synthesis inhibit a single enzyme, acetyl-CoA carboxylase which is also found in animals.

Glyphosate inhibits the shikimic acid pathway by inhibiting the enzyme 5-enolpyruvylshikimate-3-phosphatesynthase and stopping the production of chorismate. Chorismate is necessary for the production of many essential plant metabolic products like the essential acid tryptophan and IAA needed by plants for growth. Up to 35% of plant dry mass is derived from products of the shikimic acid pathway.

Hexazinone inhibits photosynthesis. Because photosynthesis is essential to carbon fixation and energy production in green plants, inhibition of this process for significant periods of time leads to plant death. Most plants tolerant to hexazinone are able to either detoxify the herbicide by breaking it down into nontoxic molecules or store the herbicide at inactive sites in the plant.

Imazapyr and the sulfonyl ureas, metsulfuron and sulfometuron, inhibit acetolactate synthase (ALS). ALS is an enzyme necessary for the synthesis of the essential branched-chain amino acids isoleucine, leucine, and valine. These branched-chain amino acids are called essential because mammals do not have the ALS pathway, cannot synthesize them and rely on plants as their source.

Toxicity

There are several terms useful in any discussion of toxicity. The term LD₅₀ is in popular use and denotes the dose which administered kills half the test organisms. While toxicity testing is not generally conducted on humans, several surrogate mammalian species are used for tests. However, LD₅₀ is not a very comforting concept since it means that half of the animals die and the other half are probably very sick. A more acceptable and useful term is the no observable effect level (NOEL). The NOEL is the highest exposure at which no effect on the test animal is observed. A similar term often used is the no observable adverse effect level (NOAEL). The oral reference dose (RfD) is the total amount of a pesticide that can be consumed daily over a lifetime without adverse health effects. Calculated from the lowest animal NOEL, EPA applies an additional safety factor of 100 (10 for interspecific and 10 for intraspecific variation) to 1000 depending on their estimate of the reliability of available data. Another measure of relative safety to mammals is the health advisory level (HAL), the maximum concentration allowable for a herbicide in drinking water, which if consumed daily over a lifetime would not be expected to cause adverse health effects. The HALs, RfDs, and NOELs from Table 2 can be used to evaluate potential toxicity from herbicide values observed in water.

Mammals. Mammalian toxicity of forestry herbicides is very low. NOEL values for animals in Table 2 show that most forestry herbicides are far less toxic than vitamin D. Most of the herbicides in Table 2 have NOELs in the range of the LD₅₀ for vitamin D (10 mg⁻¹ kg⁻¹ day⁻¹) and nicotine (less than 5 mg⁻¹ kg⁻¹ day⁻¹). Of even greater public concern is the potential for forestry herbicides to cause cancer. Technically, any chemical which causes the development of a tumor is classified as a carcinogen whether that tumor is malignant or benign. EPA classifies pesticides into groups according to their carcinogenicity: A, Human Carcinogen; B, Probable Human Carcinogen; C, Possible

Human Carcinogen; D, Not classifiable as to human carcinogenicity; E, Evidence of Noncarcinogenicity (Table 2). In general, regulatory protection of human health is initiated when the lifetime cancer risks are greater than one in a hundred thousand (10^{-5}), or one in ten thousand (10^{-4}) unless severe technical or economic constraints are present (EPA 1991). While these limits are certainly arbitrary, they are reasonable in light of the risks of cancer humans face in the course of a lifetime. For example the approximate risk of developing skin cancer over a lifetime from exposure to the sun is 1 in 3, from smoking 1 or more packs of cigarettes per day is 8 in 100, from second-hand smoke exposure for nonsmokers is 7 in 10 000, and from eating 2 oz per week of peanut butter containing naturally occurring aflatoxin B1 is 8 in 100 000 (EPA 1991). All herbicides registered for forestry use in the US are classified in Cancer Groups D or E, either not classifiable as to human carcinogenicity (D) or evidence of noncarcinogenicity (E) and the risk of developing cancer from exposure to these chemicals is currently assessed as less than 1 in 100 000.

There is a large body of literature available on the effects of forest herbicides on other mammalian wildlife species, but it is beyond the scope of this paper to discuss that literature base. Suffice it to say that herbicides are used frequently to improve wildlife habitat and to maintain plant species composition favorable to wildlife. Most of the toxicity testing reported in this paper was conducted on mice, rats, rabbits, dogs, and goats and therefore represent toxicity to be expected on mammalian wildlife species.

Table 2. Toxicity of some herbicides. Values are from U.S. Environmental Protection Agency sources, principally Office of Drinking Water.

Herbicide	HAL (mg/L)	Oral RfD (mg/kg/day)	Mammalian NOEL (mg/kg/day)	Aquatic NOEC (mg/L)	Cancer Group
Asulam*	NA	0.36	36	NA	C
2,4-D	0.07	0.01	1-37	2-15	D
Dalapon*	0.2	0.03	8.45	NA	D
Dichlobenil*	NA	0.013	1.25-60	0.006-10.0	C
Dichlorprop*	NA	NA	4-50	NA	D
Fluazifop-p-butyl*	NA	NA	NA	NA	NA
Fosamine ammonium	NA	0.01	10	NA	D
Glyphosate	0.7	0.1	10-500	25-50	D
Hexazinone	0.4	0.05	10-250	10->80	D
Imazapyr	NA	2.5	300-10000	.024->100	E
Metsulfuron	NA	0.25	25-700	>150	D
Oxyfluorfen*	NA	0.003	0.3-1000	NA	C
Propyzamide*	0.05	0.075	5-30	NA	C
Quizalofop ethyl*	NA	0.013	1.25	NA	D
Sulfometuron	NA	0.02	50	>1.2	D
Triclopyr	NA	0.05	2.5-240	NA	D

NA, Not Available

* Not registered for general forestry in US.

Avian, Insect, and Fish and Other Aquatic Organisms. The minimum requirements for pesticide registration by the EPA in the US, includes both human health and environmental risk assessments. For the environmental risk assessment, toxicological data must be presented for representative species of at-risk populations. In addition to the mammals listed above, these include honeybees, waterfleas, mallard ducks, bobwhite quail, rainbow trout, bluegill sunfish, Coho and several other species of salmon, fathead minnows, oysters, fiddler crabs, grass shrimp, pink shrimp, green algae including *Chlorella* sp and *Selenastrum* sp, bluegreen algae (typically *Anabaena* sp), diatoms including *Naviculla* sp (freshwater) and *Skeletonema* sp (marine), and at least one susceptible aquatic macrophyte (typically some species of *Lemna*). Chemical companies requesting pesticide registration have some flexibility in the species on which they conduct their toxicology tests so data for all herbicides is not available for all species and several additional or substitute species may be tested depending on the chemical company's justification. Typically, the non-mammal tests are reported as LD₅₀ values and NOECs are not consistently reported. The toxicological data for birds, fish, and other species are summarized as the LD₅₀ in Tables 3-5.

Most forestry herbicides are practically nontoxic to birds as evidenced by the data in Table 3. Toxicity testing has included both acute (gavage) and chronic (dietary administration) testing for these species. Only 2,4-D and triclopyr are considered slightly toxic. Large data-gaps exist for dalapon and dichlorprop which are not used in forestry in the US, but may be used in the European Community.

Herbicide toxicity to fish is shown in Table 4. Most are classified as practically nontoxic or slightly toxic to fish. A few, however are classified as moderately to highly toxic.

Table 3. Avian toxicity of some herbicide products.

Herbicide	Mallard Duck Acute LD ₅₀ mg/kg (Chronic LC ₅₀ , ppm)	Bobwhite Quail Acute LD ₅₀ mg/kg (Chronic LC ₅₀ , ppm)	USEPA Avian Toxicity Class ^A
Asulam*	>4000 (>75000)	>4000	PN
2,4-D	1000	668	ST
Dalapon*	---	---	---
Dichlobenil*	>2000(>5200)	683(5200)	PN
Dichlorprop*	---	---	---
Fluazifop-p-butyl*	>3528(>4321)	(>4659)	PN
Fosamine ammonium	>5000(10000)	>5000(5620)	PN
Glyphosate	(>4640)	>2000(>4640)	PN
Hexazinone	2251(>5000)	2251(>5000)	PN
Imazapyr	>2150(>5000)	>2150(>5000)	PN
Metsulfuron	>2510(>5620)	(>5620)	PN
Oxyfluorfen*	(>5000)	>2150 (>5000)	PN
Propyzamide*	20000(>10000)	8770(>4000)	PN
Quizalofop ethyl*	(>5000)	(>5000)	PN
Sulfometuron	>5000(>5000)	(>5620)	PN

Triclopyr	1698(5620)	(2934)	ST
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NA, Not Available

* Not registered for general forestry in US.

^A PN, Practically Nontoxic; ST, Slightly Toxic

These include 2,4-D, dichlobenil, dichlorprop, fluazifop, glyphosate and oxyfluorfen. Fluazifop, glyphosate and oxyfluorfen are nearly immobile in soil so it is unlikely they will reach streams in sufficient concentrations to cause a problem except in the case of a missapplication or a spill. During aerial treatment, missapplications do occur, however use of adequate streamside management zones and/or ground application should preclude the possibility of missapplication. The toxicological data in Table 4 were determined at static concentrations for 48 or 96 hours. Therefore for stream concentrations to approach LC₅₀ levels, storm runoff must contain from 1 to 6 mg L⁻¹ for 2 to 4 days. Assuming complete mixing, no hydrolysis, no photolysis, no microbial degradation and no sorption onto soil or sediment, it would be necessary to spray an application rate of 1.5 kg ai/ha directly onto the entire surface of a body of water 15 cm in depth, 3 kg ai/ha for water 30 cm in depth, and 4.5 kg ai/ha for water 45 cm in depth to achieve 1 mg L⁻¹ concentration. As we shall see later, these concentrations do not normally appear in field operations.

Table 4. Freshwater fish toxicity (48-96 hr exposures) of some formulated herbicide products.

Herbicide	Rainbow trout (<u>Oncorhynchus mykiss</u>) Acute LC ₅₀ , mg/L	Bluegill sunfish (<u>Lepomis macrochirus</u>) Acute LC ₅₀ , mg/L	USEPA Fish Toxicity Class ^A
Asulam*	>5000	>5000	PN
2,4-D	1-100	1-100	HT
Dalapon*	105	105	PN
Dichlobenil*	6.26	8.31	MT
Dichlorprop*	---	1.1	HT
Fluazifop-p-butyl*	1.37	.53	HT
Fosamine ammonium	377	590	PN
Glyphosate (IPA)	8.2	5.8	MT-PN
Hexazinone	>585.6	>1000	PN
Imazapyr	>100	>100	PN
Metsulfuron	>150	>150	PN
Oxyfluorfen*	0.41	0.2	HT
Propyzamide*	72	>100	ST
Quizalofop ethyl*	---	---	---
Sulfometuron	>12.5	>12.5	ST
Triclopyr(tech acid)	117	148	PN

NA, Not Available

* Not registered for general forestry in US.

^A HT, Highly Toxic; MT, Moderately Toxic; ST, Slightly Toxic; PN, Practically Nontoxic.

Many other groups of organisms are tested in addition to mammals, birds, and fish, but it is not possible to summarize all for this paper. The final table for toxicity provides some insights into the toxicity of forestry herbicides (commercial product) to some other organisms (Table 5). Data gaps exist for several compounds in Table 5. Only quizalofop and 2,4-D have clearly displayed toxicity to honeybees. Most of the herbicides in Table 5 are listed as slightly or toxic or toxic to duckweed. Duckweed, an aquatic macrophyte, predictably should be more susceptible to herbicides than insects, birds, fish and mammals. The data for triclopyr is for the technical grade acid instead of the formulated product except for duckweed where TEA represents the triethyl amine formulation and BEE the butoxyethyl ester.

Table 5. Toxicity of some formulated herbicide products on insects, aquatic invertebrates and aquatic macrophytes.

Herbicide	Honey Bee (<i>Apis mellifera</i>) Acute LD ₅₀ , µg/bee	Waterflea (<i>Daphnia magna</i>) Acute LC ₅₀ , mg/L	Duckweed (<i>Lemna gibba</i>) Acute LC ₅₀ , mg/L
Asulam*	>36(PN)	27(ST)	0.14(T)
2,4-D	11.5(ST)	>5(MT)	NA
Dalapon*	NA	NA	NA
Dichlobenil*	>120(PN)	6.2(MT)	0.03(T)
Dichlorprop*	NA	NA	NA
Fluazifop-p-butyl*	>200(PN)	>10(ST)	NA
Fosamine ammonium	>200(PN)	1524(PN)	>21(NOEC, PN)
Glyphosate (IPA)	>100(PN)	780(PN)	21.5(ST)
Hexazinone	>100(PN)	339.9(PN)	37.4(ST)
Imazapyr	>100(PN)	>100(PN)	0.024(T)
Metsulfuron	>25(PN)	>150(PN)	NA
Oxyfluorfen*	NA	NA	NA
Propyzamide*	NA	>5.6(MT)	NA
Quizalofop ethyl*	0.1(T)	NA	NA
Sulfometuron	>100(PN)	>1000(PN)	0.0005(T)
Triclopyr(tech acid)	>100(PN)	133(PN)	[11,TEA; 0.9, BEE]

NA, Not Available

* Not registered for general forestry in US.

^A HT, Highly Toxic; MT, Moderately Toxic; ST, Slightly Toxic; PN, Practically Nontoxic.

Forestry Herbicides in Surface Water

The results of several studies which monitored the environmental fate of forestry herbicides used in the southern United States have been reported (Michael and Neary 1993). Another report (Michael 2000) summarized surface and groundwater contamination data from reports for North America published between 1974 and 2000. The relevant data have been extracted from those two reports and are summarized in Table 6. Comparing the data in Table 6 with that in Tables 2-5, it is clear that while

surface water contamination occurs the levels of contamination are far below any recognized toxic level except for the aquatic macrophyte Lemna which is in itself a target of the herbicides. Most reported levels of contamination are even within EPA's drinking water standards so that water consumed on the site after direct spraying of streams would not contain toxic levels. Michael et al. (1999) reported on dissipation of hexazinone in forest ecosystems after application at three times the normal rate. They found hexazinone contamination of streamflow on the day of application exceeded drinking water standards for less than 30 minutes and decreased rapidly to near detection levels for the remainder of the day. The maximum concentrations found in streams subsequent to application lasted 15 to 30 minutes, came at or near peak storm discharge following precipitation events, and did not exceed 0.230 mg/L, well below all toxicity data in Tables 2-5.

Table 6. Summary of occurrence of forestry herbicides in surface water in North America.

Herbicide	Range of Concentrations in Surface Water (mg/L)
2,4-D	0.0000007-2.0
Glyphosate	0.0032-3.08*
Hexazinone	0.00007-2.4*
Imazapyr	0.13-0.680*
Metsulfuron	0.002-0.008
Sulfometuron	0.005-0.044*
Triclopyr	0.002-0.35

Data from Michael and Neary 1993; Michael 2000.

*Maximum values are result of intentional direct application to at least a portion of the stream draining the treated watershed.

Forest Herbicide Environmental Fate and Cumulative Impacts Studies in North America. There have been several environmental fate studies conducted in North America, but three serve to provide some idea of the magnitude of the effort to establish the environmental consequences of herbicide use in forest management. The studies reported below were conducted with the commercially formulated product containing the active ingredients described below. They are the Carnation Creek Glyphosate Study, the Coosa County Hexazinone Study and the Florida Wetlands Imazapyr study.

Carnation Creek Glyphosate Study. The study, conducted from 1984-1986 on a portion of the Carnation Creek Experimental Watershed Project which covers 10 km² in British Columbia, Canada, aurally applied 2 kg ai glyphosate/ha as the commercially available product Roundup. It assessed the short-term impacts of glyphosate on stream water, vegetation, soils, and stream biota. Long-term indirect impacts on water quality, erosion processes, and stream biota were also assessed. Approximately 10-15 scientists were involved in the study which began with the aerial spraying in September 1984 of 41.7 ha of the watershed. Application observed a 10m buffer strip on each side of the streams except that two tributaries and their associated wetlands were intentionally sprayed as a part of the study. Maximum observed glyphosate concentration in water occurred 2 hours post-application (0.162 mg/L) and decreased to near the detection limit of 0.0001 mg/L within 96-hours of treatment. Stream sediment glyphosate concentration peaked at

0.0068 mg/kg 24 days after treatment. Concentrations of glyphosate in leaf tissue peaked at 0.261 mg/kg in red alder and 0.448 mg/kg in salmonberry on day of treatment decreasing to less than 5% of the peak within 15 days. Caged coho salmon fingerlings suffered 2.6% mortality in the tributary that was intentionally sprayed, but mortality was 8.8% in the unsprayed control tributary. The researchers found nothing to suggest the survival of coho salmon or cutthroat trout was affected by glyphosate. They determined the LC₅₀ to be 15 to 55 mg/L for Coho Salmon fingerlings (compare with rainbow trout in Table 4). There were no statistically significant impacts on periphyton or on aquatic invertebrates, but some qualitative changes in drift patterns may have occurred (Reynolds 1989).

Coosa County Hexazinone Study. This study used by the USEPA for hexazinone reregistration was conducted by the US Forest Service in Coosa County, Alabama, USA. The study conducted from 1990-1992 utilized a 290 ha portion of the Hatchet Creek drainage and utilized 3 watersheds, two treated and one untreated control. The study assessed the direct and indirect impacts of hexazinone applied as the commercially available products Velpar L (one watershed) and Velpar ULW (one watershed) on stream water, vegetation, litter, soil, and stream biota. The study began with the aerial application in April 1990 of two watersheds with 6.72 kg/ha (3 times normal rate) of hexazinone active ingredient. Application observed a 10m buffer strip on each side of the streams except that the upper reaches of each channel were left unprotected by buffer strips. Maximum observed hexazinone concentration in water occurred during application (0.473 mg/L) and decreased to near detection limit of 0.001 mg/L within 24-hours. All subsequent inputs (peak with first storm was 0.230 mg/L) were of continuously decreasing concentrations and driven by precipitation events. Stream sediment hexazinone concentration peaked at 1.7 mg/kg. Concentrations of hexazinone were highest in leaf tissue on the day of treatment with the liquid formulation of hexazinone (Velpar L) for *Vaccinium* sp (526 mg/kg), *Cornus* sp (702 mg/kg), fern (384 mg/kg), and grass species (626 mg/kg). Dissipation from leaf tissue was rapid and nearly complete in 180 days. Native fish were not affected by hexazinone in this study. Benthic macroinvertebrate diversity, population numbers, and trophic level function were unaffected by hexazinone treatment. The six principal metabolites of hexazinone were also monitored, but were not of significant concentrations in any samples analyzed (Michael et al. 1999).

Florida Wetlands Imazapyr Study. Conducted jointly by the US Forest Service and the University of Florida, Center for Wetlands, this study investigated the impacts of imazapyr applied as the commercially available product Arsenal Applicator's Concentrate at extremely high rates on benthic macroinvertebrates. The study site, located 40 km north of Gainesville, Florida on the National Council for Air and Stream Improvement's Wetlands Study Site consisted of a 30 ha block of pine flatwoods with intermittent pond cypress domes. The microcosm experiment to determine toxicity of imazapyr to benthic macroinvertebrates, especially *Chironomus* sp was conducted in a seasonally inundated, logged pond cypress dome. Microcosms were treated with 0, 0.56, 5.6, and 56 kg imazapyr active ingredient/ha, equivalent to 0, 1, 10, and 100 times the normal prescription rate for this location. Imazapyr treatment had no statistically

significant impact on taxa richness or total abundance for all taxa, dipterans, and chironomids, even at the highest treatment rate of 18.7 mg/L. (publication submitted to J. Environmental Toxicology and Chemistry, in review).

The results of these very large studies have consistently showed no adverse effects attributable to the respective herbicides used in forest management, even when experimental conditions utilized treatment conditions at rates up to 100 times the rates normally used in forestry.

Conclusions.

Toxicological data, human health and environmental risk assessments validate the approach the US Environmental Protection Agency has used over the last 40 years in registering pesticides for use under the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA). Modern forest herbicides have not been shown to cause cancer and have been demonstrated to dissipate rapidly from the forest and aquatic environments. They do not contaminate surface or ground water in concentrations greater than drinking water standards allow and have not been shown to adversely impact aquatic ecosystems when used according to label directions.

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