Chapter XI

INTRODUCED CHEMICALS

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INTRODUCTION

Chemicals have played an important role in the success story of modern American agriculture. These same management tools — fertilizers, insecticides, herbicides, fungicides, rodenticides, avicides, piscicides, etc. — are equally important in meeting the rapidly growing demand for forest products. Their magnitude, intensity, and pattern of use is vastly different in forestry, and these chemicals provide an economically feasible means of controlling insects and disease and increasing timber production. However, their widespread use cannot proceed without adequate consideration of the potential impacts upon environmental quality. The forest land manager has a responsibility to protect the environment from contamination and thus must be aware of the potential hazards involved with each silvicultural practice that uses chemicals.

Chemicals introduced into a watershed as part of a silvicultural activity represent a potential non-point source of pollution for forest streams. Research findings and a long history of use have established that most forest chemicals offer minimum potential for degradation of the aquatic environment when they are used properly (Norris and Moore 1976). This chapter discusses the types of fertilizers and pesticides used, the magnitude and scope of chemical use, the behavior of chemicals in the forest environment, and the mechanisms by which chemicals may reach forest streams. This information forms the basis for understanding the non-point source pollution processes that result from chemicals used in silvicultural activities and for selecting effective controls. There is insufficient data to permit us to quantify control effectiveness.

DISCUSSION

MAGNITUDE AND SCOPE OF CHEMICAL USE

Newton and Norgren (1977) have categorized the chemicals used in forest management into three general groups based upon the broad objectives of their use. One group is herbicides which are used when forest productivity is to be focused on selected species. Herbicides do not influence the basic productivity of the forest ecosystem, but are used to channel that productivity into selected timber species that have special value. The second group of chemicals, including insecticides and rodenticides, is used to reduce losses of important tree species. The specific targets of these chemicals are insect and animal pests that are capable of damaging or destroying commercially desirable tree species. Fungicides used to control diseases in existing stands are also included in this group. The behavior of these two major groups is discussed together as "pesticides" in this publication. The third group of chemicals includes only fertilizers. Chemicals in this category are used to increase growth rates of commercial tree species by raising the overall productivity of forest ecosystems. Fertilizer chemicals also are used as fire retardants and will be included in this group rather than discussed separately. A wide variety of other chemicals are used in forestry for insect and disease control in nurseries, for soil stabilization, for dust control, for road surfacing, and various other purposes. However, these latter chemical uses are limited in scope and will not be discussed in this publication.

The potential impact of introduced chemicals upon forest water quality depends largely on the chemical and its pattern of use. In intensive agriculture, chemicals may be applied one or more times during a crop cycle. Crop cycles are short; thus, regular and repeated applications are a common practice. By contrast, most forest land will not be treated with chemicals at any time during a crop cycle. Lands that are treated seldom receive more than one treatment in a crop cycle. (Crop cycles range from 20 to more than 100 years.) A large number of chemical compounds are registered for use in agriculture, while in forestry less than 15 principal pesticides are used. Forestry practices account for only slightly more than 1 percent of the total pesticide use and less than 1 percent of the total fertilizer consumption in the United States.

Pesticides

Pesticide use on forest lands between July 1, 1975, and September 30, 1976, is summarized in table XI.1. The figures represent both pesticides used by the Forest Service and pesticides used on projects involving Federal assistance provided by the Forest Service (USDA 1977). In general, these figures underestimate the total use in forestry because they do not include pesticide use by other Federal land management agencies or by various State and private groups. In addition, data presented for insecticide use have been modified by deducting the figures for one large project conducted to control defoliation caused by the Eastern spruce budworm. This single insect control project accounted for 85 percent of the total figure for

Table XI.1.—Pesticide use in forests, July 1, 1975, to September 30, 1976[1]

Pesticide used	Acres treated	Percent	Pounds used[2]	Percent
Herbicide	235,551	38	563,517	62
Insecticide	326,148	53	[3]192,175	21
Fungicide	34,109	5	143,431	16
Rodenticide	22,599	4	6,053	1
Piscicide	481	0	833	0
Bird repellent	714	0	289	0

[1]Reporting period is 15 months, FY 1976 and Transition Quarter (USDA 1977).
[2]Reported as pounds of active ingredients.
[3]Data presented do not include 3,501,950 acres treated with 2,663,208 pounds of insecticide chemicals to control defoliation caused by the Eastern spruce budworm. These data were omitted in order to provide a closer approximation of the annual pesticide use pattern.

treated land area and 75 percent of the total figure for applied pesticide chemicals during the 15-month period covered by the report. Large control projects of this magnitude (3,501,950 ac) do not occur on an annual basis; therefore, the data were modified as described in order to provide a closer approximation of the annual pesticide use pattern.

Most insecticides applied to forests in the United States are applied to Forest Service and adjacent lands through Federal cooperative insect control projects for which the Forest Service has responsibility. Thus, the figures presented in table XI.1 provide a fairly close estimate of the total annual use of insecticides. Herbicide use projects are carried out independently by the various forest land management groups, and the figures presented reflect a considerable underestimate of total herbicide use. It is apparent, however, that herbicide use is considerably greater than insecticide use in terms of the amount of chemical applied, and probably exceeds insecticide use in terms of total area treated annually (with the exception of large insect control projects).

To further illustrate the scope of pesticide use in forests, a list of individual pesticide compounds or combinations is presented in table XI.2. The land area treated with each pesticide provides an indication of its importance in forest land management. Data presented were obtained from the Fiscal Year (FY)-1976 and Transition Quarter Pesticide-Use Report (USDA 1977) and essentially represent annual usage. The total number of pesticide chemicals or combinations is quite large, but the major applications employ only a few. Seven herbicide chemicals account for 95 percent of the total herbicide use.

These figures indicate that approximately 0.2 percent of the commercial forest land in the United States is treated with pesticides in any given year (0.8 percent in FY-1976 including the large Eastern spruce budworm spray program). Therefore, interaction between pesticides and water quality is not an extensive problem. In those areas treated with pesticides, however, the interaction, although localized, can be intense.

Table XI.2—Reported pesticides used for silviculture in the United States, July 1, 1975, to September 30, 1976.[1]

Herbicides	Acres treated	Herbicides	Acres treated	Insecticides	Acres treated
2,4-D	79,713	Cacodylic Acid	688	Carbaryl	[2]74,036
2,4,5-T	40,155	Methyl Bromide	605	Lindane	65,076
2,4-D & Picloram	36,662	Dacthal	473	Trichlorfon	[2]58,705
Picloram	29,891	Amitrol	412	Malathion	50,488
2,4-D & 2,4,5-T	12,797	Trichlorobenzoic Acid	354	DDT	[3]6,875
MSMA	7,624	Trifluralin	227	Acephate	[2]5,900
2,4-D & 2,4-DP	6,073	Ureabor	200	Dibrom	3,000
Simazine	5,424	Ammonium Sulfamote	194	Difluron	[2]1,800
Simazine and Atrazine	3,000	Pentachlorophenol	190	Mirex	1,674
Atrazine	2,440	Bromacil	166	Bacillus Thuringiensis	[2]950
Diphenamid	1,673	Prometryne	156	Crotoxyphos	900
Mineral Spirits	1,219	Glyphosate	146	Dimethoate	851
Dalapon	1,215			Azinphos Methyl	681
2,4,5-TP (Silvex)	1,198			Methomyl	450
Dicamba	981			Dursban	368
2,4-D & Dicamba	950			Pyrethrins	300

[1]Compiled from U.S. Forest Service Pesticide Use Reports, the amounts include chemicials used by the Forest Service and chemicals used on projects involving Federal assistance by the Forest Service (USDA 1977). Actual total amounts are considerably greater.

[2]Does not include amounts used to control Eastern spruce budworm.

[3]DDT and Carbaryl were used for plague control.

Fertilizers

Fertilizers are applied annually to only a small portion of commercial forest lands. Levels of management on most forest lands have not yet reached the intensity where fertility would severely limit economic yields; however, several major forest industrial corporations and public agencies have been using forest fertilization as a standard management practice for a little over 10 years. Fertilization operations are restricted to the Pacific Northwest, where nitrogen deficiencies are commonly encountered, and to the Southeast, where phosphorous deficiencies often limit tree growth and reduce survival of young stands.

Fertilization of forest stands in the Pacific Northwest was initiated in 1965 when one industrial corporation aerially fertilized 1,500 acres of Douglas-fir with urea. Between 1965 and 1975, approximately 750,000 acres of Douglas-fir were fertilized in western Oregon and Washington (Moore 1975b, Norris and Moore 1976). Annual fertilization increased rapidly up to 1973 when 160,000 acres were treated in 1 year. The practice then dropped drastically as the energy crisis caused a shortage of fertilizer and also raised the price of nitrogen use to nearly double the cost per acre. Fertilization practice is increasing again now in the Pacific Northwest, but has not yet reached the earlier peak of annual fertilizer application.

The first forest fertilization project in the Southeast was conducted in 1963 on 630 acres (Groman 1972). The scope of operations in the Southeast has not approached that of the Northwest, but by 1971 approximately 110,000 acres had received chemical fertilizers. When a moderate, but steady, increase in the practice was assumed, a gross estimate of total fertilized acreage through 1975 was 350,000 acres.

Investigations conducted in the hardwood stands of the Northeast indicate that nitrogen deficiencies appear to be limiting growth, and the application of potassium has effectively stimulated growth on old fields that are being reforested. However, additional field research is needed before forest fertilization will be used in that region (Beaton 1973, Mader 1973d, Weetman and Hill 1973).

Fertilizers, like pesticides, are applied to a very small proportion of the total commercial forest land each year, and applications to any given site occur infrequently. Through 1975, the total acreage fertilized was only 0.2 percent of the commercial forest land in the United States, and the forested area fertilized in any one year did not exceed 250,000 acres. However, a much larger total acreage of commercial timber stands is considered potentially amenable to fertilization. The use of this practice to increase the volume of wood fiber produced per unit area, and over a shorter period of time, can be expected to increase.

Patterns Of Chemical Use

Insecticides

At present, there are very few insecticides registered for use on forest lands. Insect damage problems in recent years have been handled as special projects, where approval for a particular chemical or formulation is usually granted by regulatory agencies on a case-by-case basis. An environmental impact statement must be prepared for each project and is used as the basis for approval or denial of the proposed chemical control program.

The chlorinated hydrocarbon insecticides are not usually selected for use in forestry when alternate chemicals are available. The application of DDT in Idaho, Oregon, and Washington for control of the Douglas-fir tussock moth in 1974 was an exception. Insecticides more likely to be used in forestry are various organophosphate and carbamate compounds. Nonresidual biological control agents are also being used. Recent research has developed suspensions of insect disease cultures that are quite specific for the target insects. Virus cultures have been used in several projects with considerable success and low impact on nontarget terrestrial and aquatic insects. This material is now registered for use in the control of Douglas-fir tussock moth.

Applications of insecticides to forest areas are almost exclusively made by aerial spraying. Large or contiguous areas may be treated in a single project to control an outbreak of defoliating insects on commercially valuable timber. Regional projects may include a large part of an entire river drainage basin. Thus, in any one year, a large percentage of the total amount of a given insecticide applied to forests in the United States may be applied in only one region. Several to many years will normally elapse before an application of any magnitude is made again in the same region. While the potential for impact of insecticides on water quality and the aquatic community may be relatively widespread on a regional basis, it is still infrequent in occurrence.

Herbicides

Herbicidal chemicals are used for a wide variety of purposes in silvicultural activities including fuel break management; vegetation control on powerline, road, and railroad rights-of-way; conversion of hardwood brush to conifers; release of established conifers from hardwood brush competition; thinning; cull tree removal in established stands; and control of noxious weeds. The most commonly used chemicals are the phenoxy herbicides (2,4-D,2,4,5-T, and Silvex), picloram, and triazines (atrazine and simazine), and the organic arsenicals (MSMA and Cacodylic acid).

Herbicides are applied by a variety of means — aerial (rotary or fixed-wing aircraft), low pressure-high volume ground spray equipment, mist blowers, stem injection devices — and in a variety of forms — pellets, granules, and undiluted concentrates. Treatment areas are typically small (5 to 200 ac) and widely scattered. Large contiguous blocks are seldom treated. The annual extent of herbicide use remains reasonably constant on a regional basis; therefore, the opportunity for interaction between herbicides and streams occurs regularly, but is of limited scope in any one drainage system. Use of herbicides on any given site is usually limited to one or, at most, two applications.

Fungicides

Fungicidal chemicals receive intensive use in forest nurseries, but are seldom used in silvicultural activities. Nursery use is more comparable to agricultural use than to forestry use and is not included in this discussion. Fungicide treatments to stumps and roots for control of root and butt rots affect only small and isolated areas and provide little, if any, opportunity for impact on water quality.

Rodenticides

Rodenticide use has decreased sharply in recent years. The small quantities used in forestry and the methods of applying them to the ground indicate that any effects on water quality are not likely to be detectable.

Fertilizers

Forest fertilization is carried out in the Pacific Northwest by aerial application. Present operations are conducted almost exclusively with helicopters (Moore 1975b). In the Southeast and on Southern pine lands, ground equipment is used to fertilize young stands and aerial equipment makes application on older stands. Soils in Florida, the Flatwoods, and Atlantic Coastal Plain subregions are deficient in phosphorus and fertilizer is applied to them at time of planting or soon thereafter. Older stands respond to nitrogen or to nitrogen plus phosphate, if the stand is on a phosphorous deficient site (Bengston 1970).

Fertilizers may be applied to relatively large contiguous areas, but a more typical practice is to fertilize smaller management units in a patchwork fashion. Treated areas are usually some distance from users of potable or irrigation waters. The infrequency of application coupled with application to undisturbed forest soils and vegetation tends to minimize the potential for impact on water quality. Buffer strips can be maintained along major streams, but it is not possible to avoid all of the smaller headwater streams. Thus, some forest streams in a fertilized watershed will normally contain detectable amounts of chemical immediately after application.

CONCEPTS OF HAZARD AND CHEMICAL ACTION

Pesticides used in forest management are selected because of their known effects on specific targets. The hazard involved in their use is the risk of adverse effects on nontarget organisms. Two factors determine the degree of hazard: (1) the toxicity of the chemical and (2) the likelihood that nontarget organisms will be exposed to a toxic dose. Toxicity alone does not make a chemical hazardous. The hazard comes from exposure to toxic doses of that chemical. Even the most toxic chemicals pose no hazard if organisms are not exposed to them. Therefore, an adequate assessment of the hazard involved in the use of any chemical requires that both the likelihood of exposure and the toxicity of the chemical be considered (Norris 1971).

Chemical action is the direct effect of a chemical on an organism. Chemical action on any organism requires exposure and, furthermore, requires sufficient quantity of chemical present at the site of action, in an active form and for a sufficient period of time, to produce a toxic effect. There are two kinds of toxicity: acute and chronic. Acute toxicity is the fairly rapid response of organisms to one, or a few, relatively large doses of chemical administered over a short period of time. Chronic toxicity is the slow or delayed response of organisms that occurs after repeated or continuous exposure to small doses of chemical extending over a relatively long period of time. There are various gradations between these two extremes. The kind of response (acute or chronic) observed in nontarget organisms depends on the magnitude of the dose, the duration of exposure, and the behavior of the chemical.

Toxicity. — A consideration of the principles of toxicity or a review of the toxicity characteristics of silvicultural chemicals is beyond the scope of this chapter. Newton and Norgren (1977) provide an excellent summary of this topic. Reference sources for the more frequently used silvicultural chemicals are given in appendix XI.C.

Potential for exposure. — The potential for exposure of nontarget organisms is determined by the initial distribution of the chemical and its subsequent movement, persistence, and disposition in the environment. When a chemical is applied to a forested watershed, there is an interaction between the properties of the chemical and the properties of the environment. These interactions follow the basic laws of physics, chemistry, and biology and define chemical behavior (fig. XI.1). The resulting quantities of a chemical found in different parts of the environment at varying times after application determine the duration and magnitude of exposure of different organisms to the chemical. The overall impact of chemicals on both target and nontarget organisms and the selective action of chemicals depend on this exposure.

CHEMICAL BEHAVIOR OF PESTICIDES

The behavior of a chemical consists of its movement, persistence, and disposition in the environment. Such behavior determines how much chemical is in what part of the environment for what period of time and in what form. The initial distribution of a silvicultural chemical and its subsequent behavior in the terrestrial environment determines its potential role as a non-point source pollutant. Its behavior in the aquatic environment and its inherent toxicity determine its importance.

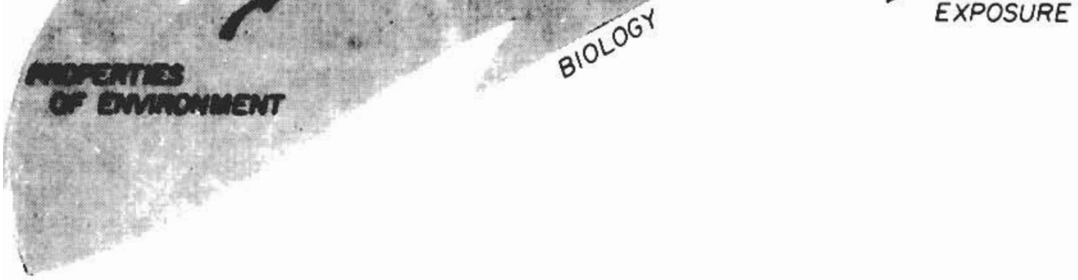


Figure XI.1.—The interaction of chemicals with the environment (Norris 1971).

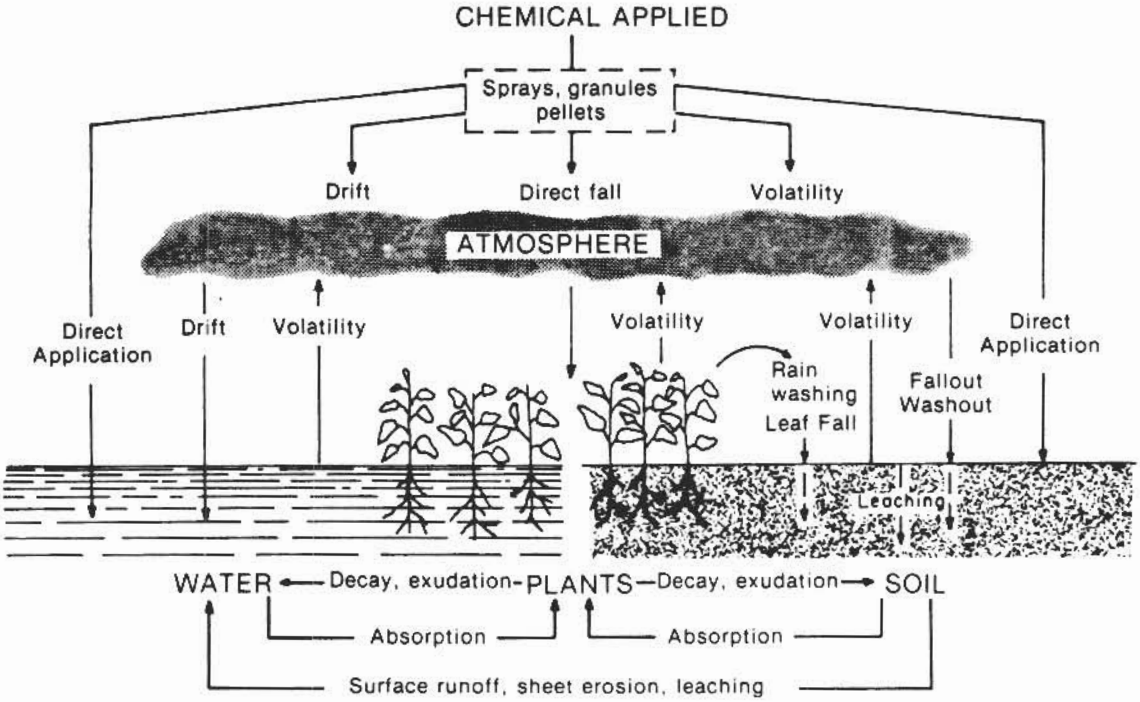


Figure XI.2.—The distribution and disposition of chemicals in the environment (Foy and Bingham 1969).

Initial Distribution Of Spray Materials

Aerially applied chemicals are distributed initially among four major components of the forest environment: air, vegetation, the forest floor, and surface waters (fig. XI.2). The amount of chemical entering each portion of the environment is determined by the chemical and equipment used and the environmental conditions that prevail at the time of spraying (Norris and Moore 1971).

Some spray material is dispersed by the wind as fine droplets called "drift." The degree of lateral movement of spray drift depends on droplet size, height of release, and wind velocity (fig. XI.3) (Reimer and others 1966). Additional amounts of chemical may remain in the air due to volatilization of spray materials while falling through the air. Most of the pesticide chemical not lost through drift or volatilization is intercepted by vegetation or the forest floor. Some small amount of pesticide may fall directly on surface waters.

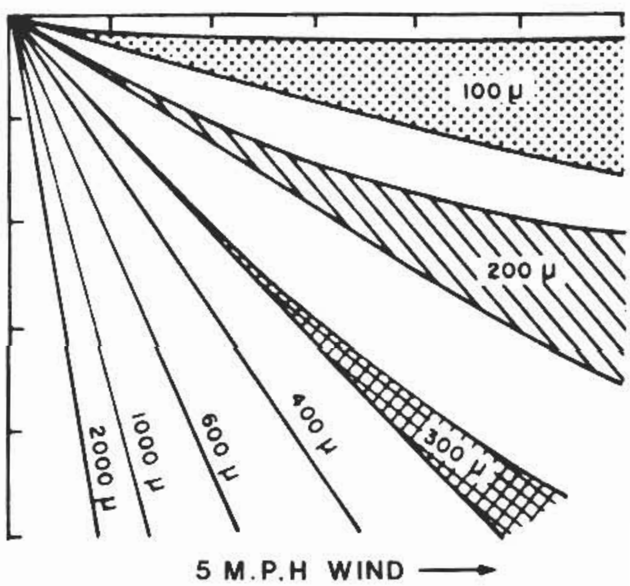


Figure XI.3.—Lateral movement of spray particles of various diameters falling at terminal velocity in an 8 km/hr crosswind (5 mph = 8 km/hr; 1 ft = 0.3048 m) (Reimer and others 1966).

Movement, Persistence, And Disposition Of Pesticides

The movement of pesticides includes movement within a given compartment of the environment (leaching in the soil profile) or movement from one compartment to another (washing pesticide residues from leaf surfaces to the forest floor by precipitation). Persistence is the tendency of pesticides to remain in an unaltered form. The disposition of pesticides concerns the various physical, chemical, and biological pathways taken by chemicals in becoming biologically harmless products. These aspects of chemical behavior will be discussed for each environmental compartment.

Distribution In Air

Losses of herbicides and insecticides to the air may be appreciable, but there is little quantitative data. During one test in western Oregon, for example, from 20 percent to 75 percent of a herbicide application did not reach the ground, but these results were confounded by the presence of nearby overstory vegetation[1]. Use of helicopters in place of fixed-wing aircraft and the introduction of improved drift control nozzles and spray additives have greatly reduced the amount of chemical reaching sites outside the target zone.

More recent work has used spray interception disks. Norris and others (1976b) reported 85 percent recovery of picloram and 70 percent recovery of 2,4-D when using the spray interception disks in a southern Oregon brush field that had been sprayed by helicopter. On four powerline rights-of-way in Oregon and Washington treated by helicopter with 2,4-D and picloram, interception disks recovered 71 percent of the 2,4-D and 90 percent of the picloram.

Several things can happen to that portion of chemical that becomes dispersed in the air. Fine droplets (drift) or vapors (volatiles) can be moved to other locations where they settle to the earth. Droplets and vapors can also be washed out with rain, absorbed or taken up by plants and other organisms, or adsorbed on various surfaces. Another possible fate for many pesticides is photodegradation (Moilanen and others 1975). With the exception of direct application or the deposition of spray drift, the air is not an important source of chemicals that later enter the aquatic environment.

[1]*Newton, M., L.A. Norris, and J. Zavitkovski. Unpublished data on file Sch. For., Oregon State Univ., Corvallis.*

Distribution In Vegetation

The amount of pesticide intercepted by vegetation depends on the rate of application, the nature and density of the vegetation, and the physical characteristics of the spray material. Chemicals intercepted by vegetation may be volatilized into the atmosphere, washed off by rain, or adsorbed on the leaf surface. There is limited absorption and very little translocation of many pesticides intercepted by foliage. Through the action of rain, much of the unabsorbed pesticide will be washed from the surface of the leaf. Pesticide remaining on the leaf surface and any pesticide not translocated to other plant parts will enter the environment of the forest floor during leaf fall.

Pesticides retained by the plant may be excreted back into the environment through the roots or they may end up in some plant storage tissue to be released at a later time. Through metabolic activity, plants may degrade a pesticide to non-biologically active substances.

Studies of herbicides show that the highest concentrations of residue occur in foliage shortly after application (see table XI.3) (Morton and others

Table XI.3.—Residues of herbicide[1] in forage grass

Time after treatment (Weeks)	Herbicide residue		
	2,4-D[2]	2,4,5-T[2]	Picloram[3]
	ppm		
0	100	100	135
1	60	60	---
2	50	30	32
4	30	15	---
8	6	6	24
16	1	2	16
52	---	---	3

[1]Rate of application equals 1.12 kg/ha.
[2]Data from figure 4 in Morton and others 1967.
[3]Data from table 5 in Getzendaner and others 1969.

1967, Getzendaner and others 1969). A combination of factors causes the residue concentrations to decrease rapidly with time. Growth, dilution, weather, and metabolism of the herbicide by the plant are particularly important.

Weathering is very important in reducing residue levels of carbaryl on foliage. Wells (1966) reported that rain in excess of 1.8 inches (45 mm) falling 12 to 24 hours after spraying reduced initial residue levels of carbaryl on oak foliage from 190 ppm to about 15 ppm 3 days later. Degradation of carbaryl residues on plants is less important, but plants absorb only small amounts (Union Carbide 1968). Formulation also influences persistence of residues on foliage (Fairchild 1970). Carbaryl applied in an 80 percent wettable powder formulation had a half-life (the time required by an organism to eliminate, by biological or chemical processes, half the quantity of a substance taken in) of 3 to 4 days, while carbaryl applied in a Sevin-4-oil formulation was found to have a half-life of 8 to 10 days on range grasses. Typical initial residue levels on forest foliage ranged from 30 to 100 ppm immediately after treatment. These residues decreased to 5 to 20 ppm after 2 or 3 weeks (Back 1971).

Dylox (trichlorfon) insecticide is relatively nonpersistent; only small amounts remain on treated foliage beyond 1 week after application. Residue levels of 0.33 to 3.3 ppm trichlorfon on leaves, 0.42 to 1.1 ppm on twigs, and 1.5 ppm on forest litter 26 days after application were reported by Wilcox (1971). Residues were still detectable after 106 days, even though residues declined most rapidly over the first 7 days following spraying (Devine and Wilcox 1972). Weiss and others (1973) reported that Dylox residues dropped sharply within a few days after spraying, and that after 60 days, 15 percent of the initial level remained on leaves, 5 percent on the forest floor, and less than 1 percent in the soil.

Orthene, also an organophosphate insecticide, is readily degraded by plants. It has an observed half-life of from 5 to 10 days (Chevron 1973). This insecticide adheres to or is absorbed by leaf surfaces and washing of field-treated vegetation will remove no more than 5 percent of the residue present. Translocation from treated leaves to other parts of the plant is only very slight. Orthene is not persistent on forest vegetation because of its short half-life (Devine 1975). Following field applications at ¼, ¾, and 1½-lb active ingredient/acre, residues on leaves and in forest floor material declined to nondetectable levels in 1 to 2 months.

Distribution On The Forest Floor And In Soil

The forest floor is a major receptor of aerially applied spray materials. Pesticides on the forest floor may be volatilized and reenter the air, adsorbed on soil mineral or organic matter, leached through the soil profile by water, absorbed by plants, or degraded by chemical or biological means. Volatilization of chemicals from the soil surface may be responsible for the redistribution of fairly large amounts of some pesticides such as DDT and perhaps some phenoxy ester herbicides.

The length of time chemicals persist in the forest floor and soil bears strongly on the probability they will contaminate the aquatic environment. Pesticide degradation is usually biological, but chemical degradation is important in the loss of amitrole and the organophosphate insecticides (Hance 1967, Kaufman and others 1968, Norris 1970).

The common brush control herbicides (2,4-D, amitrole, 2,4,5-T, and picloram) are all degraded in the forest floor although their rates of degradation vary considerably (fig. XI.4). In red alder (*Alnus rubra*) forest floor material, 80 percent of the amitrole and 94 percent of the 2,4-D were degraded in 35 days, but 120 days were required to degrade 87 percent of the 2,4,5-T. Picloram degradation was slow, 35 percent in 180 days (Norris 1970).

Adsorption and leaching are processes which work in opposition to one another. Adsorbed molecules are not available for leaching, but adsorption is not permanent. The amount of pesticide that is adsorbed is in equilibrium with the amount of pesticide in the soil solution. As the concentration of pesticide in the soil solution decreases, more pesticide will be released from adsorption sites (fig. XI.5). Thus, adsorption provides only temporary storage, and the soil is, in effect, a reservoir of the chemical that will eventually be released. Leaching is a slow process, capable of moving pesticides only short distances (Harris 1967, 1969). Herbicides are generally more mobile in soil than insecticides, but mobility is relative, and even the movement of herbicides is usually measured in terms only of inches or a few feet.

Most of the chemicals applied to the forest, regardless of method of application, eventually reach the forest floor and soil compartments. Chemical behavior in this part of a forest watershed is particularly important because it determines whether these introduced chemicals will be immobilized, degraded, or transported to

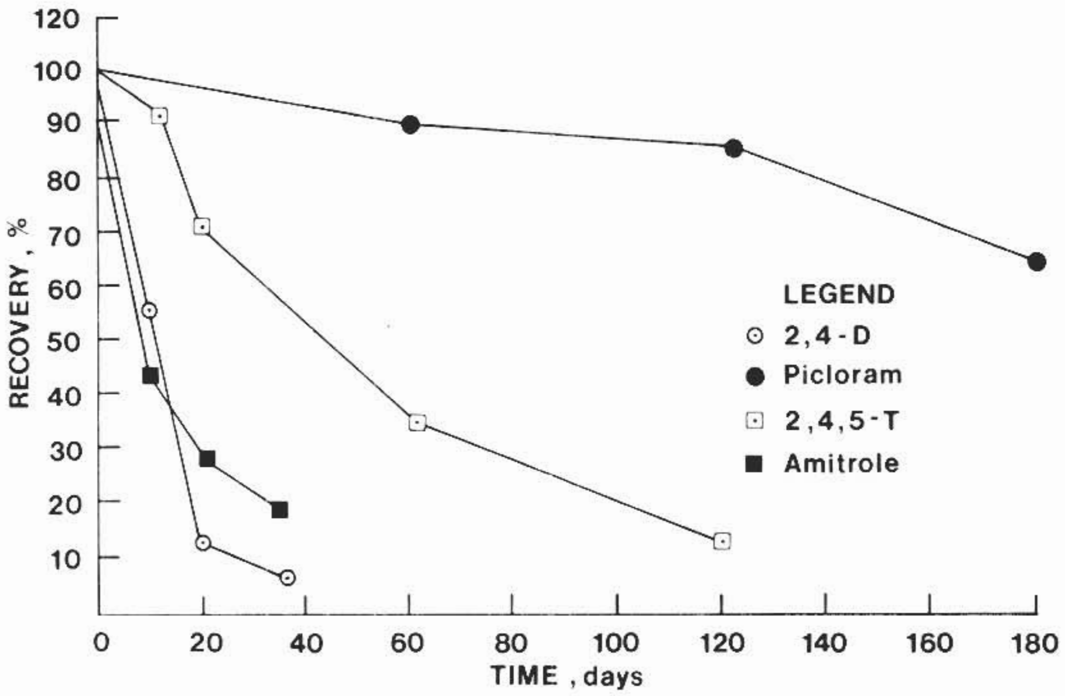


Figure XI.4.—Recovery of 2,4-D, amitrole, 2,4,5-T, and picloram from red alder forest floor material (Norris 1970).

$$\text{CHEMICAL} + \text{ADSORBENT} \underset{k_2}{\overset{k_1}{\rightleftharpoons}} \text{CHEMICAL : ADSORBENT}$$

Figure XI.5.—Chemical adsorption in soil is an equilibrium reaction.

the aquatic environment. The forest floor and soil make up a very active biological system that provides a number of processes by which pesticides can be destroyed, thus preventing their accumulation or redistribution. Each pesticide material, however, has its own chemical and physical properties that give it some degree of stability against degradation. Kearney and others (1969) have grouped the pesticides into major chemical classes and summarized their persistence in soil (fig. XI.6). Only the organochlorine insecticides have persistence times expressed in years. Persistence in the soil of all the other classes or groups of pesticides is measured in weeks or months. The length of each bar in figure XI.6. indicates the time required for 70 to 100 percent degradation of the particular pesticide when it was applied at normal rates. Data used to construct the graphs were obtained from studies conducted in agricultural soils, but the same pesticides used in forestry should have the same relative stability in forest soils. Some pesticides that are degraded by soil microbial activity persist for a shorter period of time in forest soils.

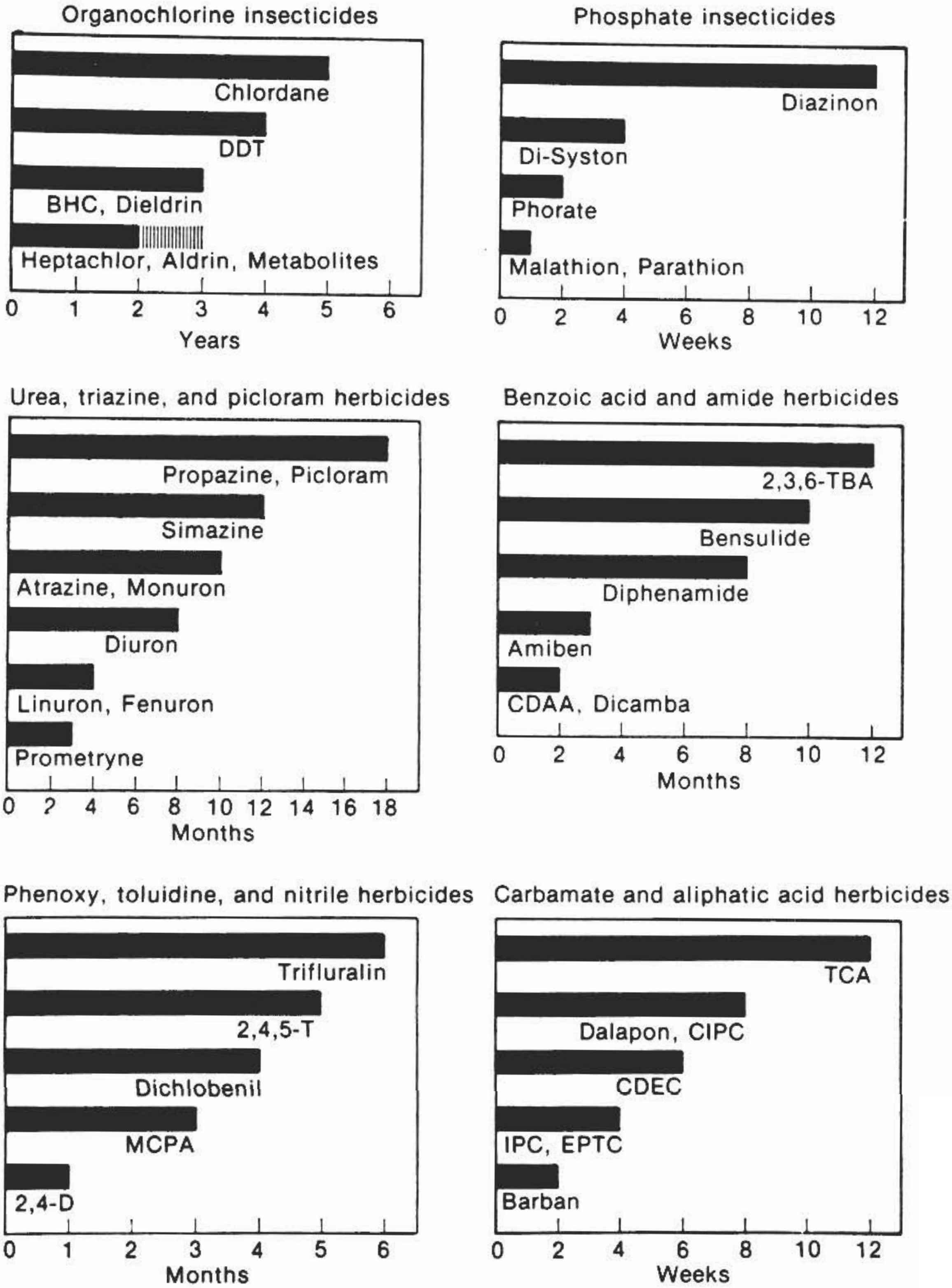


Figure XI.6.—Persistence of individual pesticides in soils (Kearney and others 1969).

Carbamate and organophosphate pesticides are relatively nonpersistent in the forest floor and soil. When Sevin-4-oil was applied at 1 pound carbaryl/acre to control the gypsy moth, pesticide residues in the soil were still detectable 64 days later, but were below the level of detection (0.2 ppm) 128 days after spraying (Wilcox 1972).

Dylox (trichlorfon) breaks down rapidly in the soil. In studies carried out in New York (Judd and others 1972), trichlorfon was not detected in any forest soil or lake mud samples after 4 days. Wilcox (1971), in another New York study, reported that after 14 days no residues were detected in soil. Malathion applied to soil persisted for 2 days in one study and 8 days in another (Pimentel 1971). Devine (1975) found that residues of Orthene in soil dissipated in 3 days. Studies conducted by Chevron Chemical Company (1973) on the persistence of Orthene in nine soils types indicated a half-life of 0.5 to 6 days when treated at 1 ppm.

Distribution In Surface Waters

Degradation of environmental quality in the forest is often first recognized by changes in stream quality. Stream contamination is a most important expression of environmental contamination in the forest because water is not only the habitat for many biological communities, but also a critical commodity to downstream users. Pesticides may enter streams by several pathways and forest managers can greatly influence the amount of chemical which enters streams near treated areas.

Entry Of Pesticides Into The Aquatic Environment

Any amount of pesticide that has not been degraded, adsorbed, volatilized, or taken up by plants is available to move into the aquatic environment.

Movement To Streams From The Air

That portion of the introduced chemical which is not lost as drift or intercepted by vegetation or the forest floor will fall directly on surface waters. This route of entry offers the greatest potential for short-term, but high-level, contamination of streams by pesticides in the forest environment. Stream contamination by herbicide residues from forest spray operations in Oregon has been intensively studied (Norris 1967, Norris and Moore 1971, Norris and Moore 1976, Norris and others 1976a, Norris and others 1976b, Norris and others 1977). Herbicide residues were found for short periods in all streams that flow through or by treated areas.

Although stream monitoring has been carried out in conjunction with numerous field applications of herbicides over a period of more than 10 years, measured residues of the phenoxy herbicides have never exceeded 0.1 mg/l in western Oregon. Concentrations of amitrole to 0.4 mg/l were found in one stream immediately below a spray unit in the Coast Range of Oregon (Norris and others 1966). Examples illustrating several important points about minimizing residues in streams are presented in appendix XI.A.

For a given rate of application, the concentration of herbicides in streams depends on the surface area of the stream in relation to its volume. The total amount of herbicide entering a stream varies with the length of the stream which receives the spray materials and with the location of the spray unit boundaries with respect to the stream. The highest concentrations of herbicide are found in streams originating in or flowing directly through spray units. In contrast, lowest concentrations are found in streams which are totally excluded from the spray area.

Surface water contamination caused by direct application of DDT was measured during and after forest spraying in eastern Oregon. The maximum DDT concentration (0.28 μg/l) was a sample taken a few hours after spraying. Most samples contained less than 0.01 μg/l DDT (Tarrant and others 1972). Endrin has also been found in forest streams following direct aerial seeding with endrin-coated Douglas-fir seed. The maximum concentration of 0.070 μg/l occurred immediately after seeding and decreased rapidly to below detection level (0.001 μg/l) within 5 hours (Moore and others 1974). At a second site in the same study, the maximum concentration of endrin found in a slower moving stream was 0.013 μg/l. However, residue concentrations decreased slowly and did not reach the detection limit of 0.001 μg/l until 10 days after seeding.

During insecticide application, some spray does reach small inconspicuous streams and small bodies of water such as shallow ponds or puddles even though direct application to larger bodies of water is avoided. Triclorfon has been found in small amounts in water samples collected immediately after spraying, but the concentration dropped below detectable limits 4 days after spraying (Judd and others 1972). In an outdoor pond trichlorfon had a half-life of 0.3 days (Chemagro 1971).

The movement of spray drift from treatment areas to surface waters is also an important source of pesticides in the aquatic environment, especially when large contiguous areas are sprayed. The amount of spray drift which occurs is influenced by the carrier, the size of the droplets, and the height of release. Wind speed, temperature inversions, relative humidity, and temperature are environmental factors which influence the droplet's size, rate of evaporation, speed of vertical descent, and, therefore, the extent of its lateral movement (Hass and Bouse 1968).

Movement To Streams From Vegetation

Only small amounts of pesticides will enter the aquatic environment from the washing action of rain on the vegetation that overhangs stream courses and from leaves falling into the water. Residues on buffer strip vegetation will normally be restricted to small amounts of chemical moved laterally as spray drift during application and volatile material brought down by precipitation. Some pesticide chemicals are excreted from plant roots, but the quantities are very small and only the roots in the stream or hydrosoil would add chemicals to the water. How much chemical enters the stream in this way has not been studied.

Movement To Streams From The Forest Floor And Soil

Two competing reactions, leaching or infiltration and surface runoff, are the ways by which chemicals are moved from spray areas to streams. Factors favoring infiltration will decrease the amount of surface runoff and with it the overland flow of introduced chemicals. The amount of chemical actually entering a stream due to surface runoff will depend on:

1. Distance from treated area to the nearest stream,
2. Infiltration properties of the soil or surface organic layer,
3. Rate of surface flow, and
4. Adsorptive characteristics of surface materials.

Conditions that retard the rate of surface runoff will minimize the immediate level of stream contamination. The long-term stream load of pesticide will be reduced as well, since a longer residence time in the soil provides greater opportunity for adsorption and degradation.

Runoff from agricultural lands and discharge from manufacturing plants are the principal sources of water pollution by pesticides (Nicholson 1967). Barnett and others (1967) maximized the probability of runoff by applying artificial rain (2.5 in/hr) to recently tilled agricultural land and found 38 percent of the 2,4-D isooctyl ester in washoff (sediment plus water), but only 5 percent of the 2,4-D amine salt. In another study, only small amounts of 2,4,5-T and picloram moved from compacted sod or recently plowed fallow clay loam soil following artificial rainfall of 0.5 inch in 1 hour (Trichell and others 1968). Movement of contaminated water over untreated soils significantly reduced the concentration of herbicide in the runoff (table XI.4).

Table XI.4.—Effect of slope, rate of application, and movement over untreated sod on the concentration of picloram in runoff water[1,2]

Rate (lb/ac)	Slope (percent)	Portion of plot treated	Picloram in runoff water[3]	Applied picloram runoff
	Percent		ppm	Percent
2	8	Upper half	2.1	1.6
1	8	Entire	3.8	5.5
2	3	Upper half	1.3	0.9
1	3	Entire	2.0	2.8

[1]Data from Trichell, and others 1968.
[2]Picloram applied as potassium salt in water .88 lbs/ac (400 g/ac).
[3]Simulated rainfall was 0.5 in/hr, 24 hours after herbicide application.

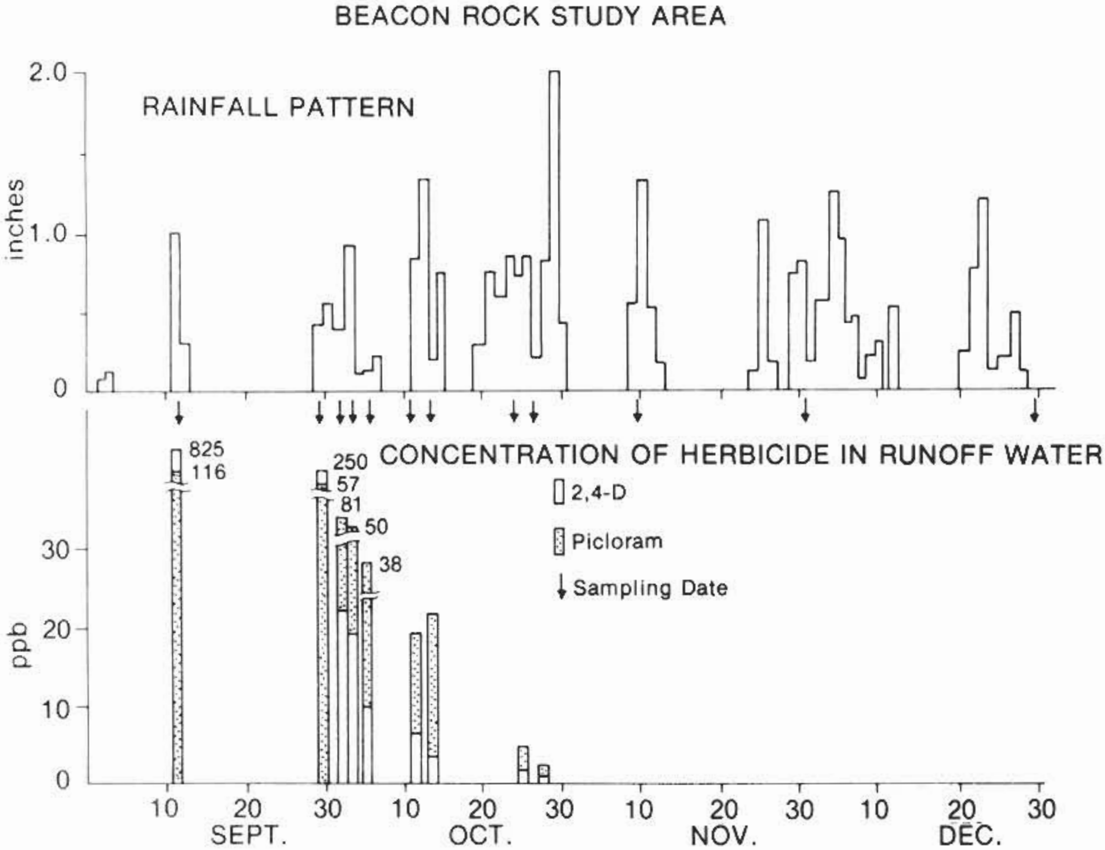


Figure XI.7.—Precipitation and herbicide runoff patterns at the Beacon Rock Study area. A total of 6 and 1.5 lbs/ac of 2,4-D and picloram, respectively, was applied in two treatments (July and August 1967). Herbicide residues were measured in ponded drainage water from the treated area (Norris 1969).

In areas where runoff is likely to occur, pesticide washoff will be greatest during the first storms after the pesticide is applied. The greatest potential for pesticide movement exists when significant amounts of precipitation occur shortly after application. On a powerline right-of-way in southwestern Washington, the highest concentrations of the herbicides 2,4-D and picloram in runoff water were associated with the first significant storm following the herbicides' application (fig. XI.7). The concentrations of herbicides declined with time despite subsequent storms of even greater intensity (Norris 1969). Mobilization of chemicals in transitory stream channels by the expanding stream system described by Hewlett and Hibbert (1967) is believed to account for the immediate flush of chemical observed with the first significant storms. Norris and others (1976a, 1976b) found the total discharge of picloram and trichlopyr from two watersheds was approximately equal to the amount of chemical applied to an ephemeral stream channel.

There is ample evidence to show that phenoxy and amitrole herbicides are not lost in runoff during intense fall precipitation from lands treated with herbicides in the spring in western Oregon (Norris 1968). Favorable conditions and ample time for degradation of the herbicides under these circumstances reduce the chance that they will be mobilized in ephemeral stream channels.

In order to determine to what extent trichlorfon might move with surface runoff, Chemagro (1971) sprayed this insecticide on sloping plots of three soil types at 20 pounds active ingredient/acre. Simulated rainfall was then applied once weekly for 5 weeks. After the 5-week period, total residue in runoff water from a silt loam soil was 2.86 percent of the total applied. Losses from a sandy loam were 0.65 percent, and from a high organic silt loam, 0.35 percent.

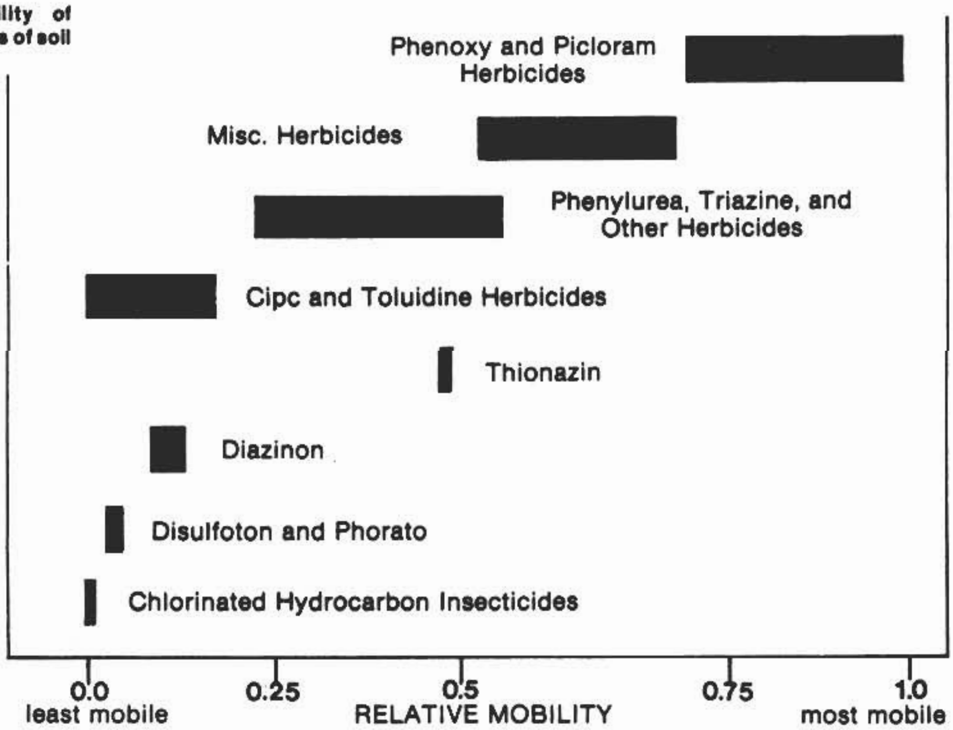
Pesticides leach into the soil profile and subsequently are transported to streams by subsurface drainage; this is another possible route to stream contamination. Leaching, however, is a relatively slow process in highly organic forest soils; only small amounts of chemical move through short distances. Harris (1967, 1969) has determined the relative mobility of pesticides in soil columns leached with water (fig.XI.8). Herbicides in general are more mobile in soil than pesticides, but this mobility is only relative. Even the herbicides move only short distances in the soil under normal conditions (Scifres and others 1969, Wiese and Davis 1964).

Orthene is not tightly bound by soil particles and is, therefore, susceptible to leaching. However, it does not persist long enough to allow any significant movement, either by leaching or surface runoff (Chevron 1973). This compound also degrades rapidly in water. In the laboratory, Orthene showed a half-life in water of 46 days, but, in the field, degradation is accelerated by breakdown in aquatic vegetation and soil microorganisms in bottom mud; measurable residues were gone in 1 to 9 days (Chevron 1973, 1975; Devine 1975).

Boschetti (1966) reported carbaryl residues of 1 to 3 parts per billion (ppb) in streams in or near areas treated for gypsy moth control in the Northeast. In a later study (Devine 1971), carbaryl residue in ponds and streams ranged from nondetectable to 50 ppb during an 8-day period following spraying. Residues in pond mud ranged from nondetectable to 620 ppb.

DDT is very low in water solubility (1.2 μg/l) and is extremely resistant to movement in soil (Bowman and others 1960, Guenzi and Beard 1967, Reikerk and Gessel 1968). Any appreciable movement of DDT through soils by leaching must, therefore, be the result of movement of colloidal particles of the free or adsorbed pesticide. The likelihood of large amounts of the chemical entering the aquatic system seems remote when movement of chemicals by leaching can be measured in inches and the distance between spray units and streams may be hundreds of feet.

Figure XI.8.—Relative mobility of pesticides leached in columns of soil (Harris 1967, 1969).



Summary Of Pesticide Entry Into The Aquatic Environment

To summarize, most chemicals enter the aquatic environment through either direct application or drift of spray materials to the water surface. The forest manager has considerable control over these. Research has demonstrated that direct application of spray materials to water surfaces can be minimized by excluding streams from treatment areas. Careful selection of spray equipment, chemical formulations, and conditions of application will minimize the potential for drift.

Mobilization of residues in ephemeral stream channels during the first significant storms following chemical application is the second most important source of chemical residues in forest streams.

Pesticide residues moving overland with surface runoff during intense precipitation is the third most important way by which chemicals may enter the aquatic system. The phenoxy herbicides, amitrole, and the carbamate and pyrethrum insecticides degrade rapidly so they are available for overland transport to streams for only short periods. Picloram may persist for more than one season, but its tendency to leach into the soil profile reduces its chances of moving by surface runoff into streams. DDT and similar compounds are resistant to degradation and leaching, therefore, they are exposed to overland transport for extended periods of time. However, the chlorinated hydrocarbon insecticides are no longer selected for use in forestry when alternate chemicals are available. Overland flow of water on forested watersheds is relatively uncommon, and pollution of streams from this source will be limited to areas where rates of infiltration are considerably less than normal rates of precipitation. The stream contamination that does occur will be reduced when the contaminated water moves over the untreated buffer strips. Leaching is not a significant process in the entry of forest chemicals into streams. Specific Controls are listed under "Aerial Drift and Application of Chemicals," and "All Resource Impacts" in Section B of Chapter II: Control Opportunities.

Behavior In The Aquatic Environment

How an aquatic organism responds to a chemical will depend on the duration and magnitude of the exposure and the interaction of the organism with other stresses in its environment. How a chemical behaves in the aquatic environment will determine both duration and magnitude of the exposure.

Chemicals may be lost from the aquatic environment through volatilization; adsorption in stream sediments; absorption by aquatic biota; degradation by chemical, biological, or photochemical means; or dilution with downstream movement (fig. XI.2).

Volatilization

The amount of pesticide lost from water by volatilization varies with both the properties of the chemical and the environmental conditions. The chlorinated hydrocarbon insecticides (like DDT) are of very low solubility in water and tend to collect at water surfaces in films where they may be subject to co-distillation. Water suspensions containing 5 μg/l DDT have been reported to lose 30 percent of the insecticide in 20 hours at 79° F (26° C) (Bowman and others 1964). Fuel oil carriers may concentrate oil soluble pesticides at water surfaces (Cope 1966).

Adsorption

In turbulent streams chemicals will be quickly dispersed throughout the water allowing maximum interaction with various adsorbing surfaces (Cope and Park 1957). Reductions in pesticide concentrations in water by adsorption depend on the rate, extent, and strength of adsorption, and the mixing characteristics of the stream (which will govern the opportunity for interaction within the stream bottom). Researchers have given these factors only limited attention. Clay and fine silt are effective in adsorbing and reducing the activity of DDT and other chlorinated hydrocarbon insecticides in river water (Ferguson and others 1966, Fredeen and others 1953). Bottom sediments from bodies of water treated with various phenoxy herbicides frequently contain residues which may indicate adsorption (Bailey and others 1970, Smith and Ison 1967). Aly and Faust (1965) reported that the amounts of 2,4-D adsorbed on suspended clays in water were small. Considerable research is needed to clarify the importance of adsorption in reducing pesticide concentrations in water.

Degradation

There are conflicting reports on the persistence of pesticides in streams. In one study, 2,4-D esters were hydrolyzed to free acid in 9 days in lake water, but 2,4-D acid persisted up to 120 days (Aly and Faust 1964). In another study, only 40 percent degradation of 2,4-D in water was observed in 6 months, during which excellent conditions for biological activity were present (Schwartz 1967). A considerable decrease in degradation of 2,4-D was observed in bacterially active natural river waters that had reduced levels of dissolved oxygen (fig XI.9).

Robson (1968) reported that the persistence of 2,4-D in fresh water was decreased from 9 weeks to 1 week when small quantities of soil previously treated with phenoxy herbicides were added. Rapid degradation of 2,4-D occurred in water samples collected from areas with a history of repeated 2,4-D applications (Goerlitz and Lamar 1967). Many surface waters may lack suitable conditions for biological degradation of herbicides or they may not contain populations of microbes adapted to use of the phenoxy herbicides as substrates (Hemmet and Faust 1969).

Degradation of certain chemicals is pH dependent. Amitrole resists degradation in activated sludge cultures, distilled water, or sewage held at room temperatures for various periods of time (Ludzak and Mandia 1967). Carbaryl rapidly degrades in sea water, but it will persist for longer periods in the more acid conditions found in forest streams (Aly and El-Dib 1971, Karinen and others 1967). The rapid hydrolysis of malathion in water is also pH dependent (Guerrant and others 1970), 50 percent decomposition occurred in 26 days at pH 6.0 and in 2.5 hours at pH 10.0.

In studies conducted as a part of gypsy moth suppression in the Northeast, carbaryl persistence in the aquatic environment was found to be brief. Romine and Bussian (1971) suggest that an initial level of 1 mg/l will be completely gone in 1 to 2 days. In an earlier study, water residues of 30 $\mu g/l$ dropped to 1-5 $\mu g/l$ in 1 day (USDA 1964).

Carbaryl, the phenoxy herbicides, amitrole, and picloram are all susceptible to photodegradation (Crosby and Li 1969, Karinen and others 1967). The importance of this reaction in the natural environment is questionable, however, because most streams are shaded and there is limited penetration of the water by ultraviolet radiation.

Downstream Movement

Downstream movement of chemicals and the resulting dilution due to natural stream mixing and the addition of uncontaminated water from side streams is one of the most important mechanisms by which the concentration of pesticides in streams is reduced near treatment areas. Although the hazard of exposure is not eliminated until the residues are completely degraded to nontoxic compounds, dilution as the result of downstream movement can reduce the concentrations of pesticides in streams to levels that do not represent a hazard to nontarget organisms. DDT residues were carried downstream in well defined blocks and did not persist for long periods at sampling stations located

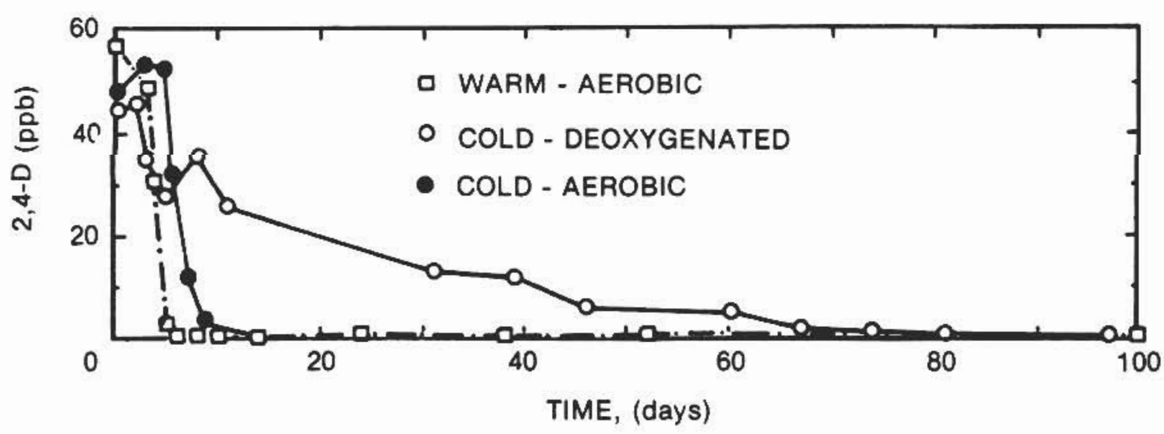


Figure XI.9.—The degradation of 2,4-D in a bacterially active water culture (DeMarco and others 1967).

along an 85-mile stretch of the Yellowstone River following spray operations in Montana (Cope 1961). Marked reductions in concentrations of amitrole and the phenoxy herbicides were observed in water due to downstream movement (Marston and others 1968, Norris and others 1966).

CHEMICAL BEHAVIOR OF FERTILIZERS

Initial Distribution In Air, Vegetation, And Forest Floor

Many concepts concerning the initial distribution of pesticides apply also to fertilizers, but there are some important exceptions. The rate at which nitrogen fertilizer is applied varies with site and timber type but is usually 150 or 200 pounds of urea nitrogen/acre. Phosphorus is applied at rates between 80 and 100 pounds P_2O_5/acre in the southeast. In contrast with pesticides, where significant quantities may remain in the atmosphere, essentially all of the fertilizer applied reaches the intended target. However, because of the higher rates of application, it is necessary to make at least two flights over the unit and a uniform rate of application over an entire unit is difficult to obtain (Strand 1970).

The introduction of large, specially coated urea granules (forest grade) has eliminated the drift problems that were experienced when standard agricultural urea was used. Drift problems still exist, however, when standard agricultural urea (45% N) is used, or when experimental liquid formulations of nitrogen are substituted for the forest granules. Should liquid fertilizer formulations come into commercial use, their initial distribution in the environment will be subject to the same factors controlling distribution of aerially applied pesticides.

Because very little granular fertilizer is intercepted by a dry forest canopy, the forest floor is the major receptor. The initial distribution of aerially applied fertilizers is thus restricted to the forest floor and to exposed surface waters within the treated areas.

Urea fertilizer is highly water soluble and readily moved into the forest floor and soil by any appreciable amount of precipitation. Under normal conditions, urea is rapidly hydrolyzed (4-7 days) to the ammonium ion by the enzyme urease. When moisture is limited, however, urea granules may be slowly hydrolyzed on the forest floor, resulting in a marked increase in surface pH and a loss of ammonia nitrogen by volatilization. In a laboratory study, Watkins and others (1972) measured losses of ammonia nitrogen ranging from 6 percent to 46 percent of the urea nitrogen applied to forest floor and soil depending on the nature of the surface, surface pH, and rate of airflow across the surface. Although some applied nitrogen is undoubtedly lost by volatilization in the field, it is generally conceded that such losses are small. Time of application is important, and forest fertilization projects are usually conducted during the spring or fall months to take advantage of precipitation. Urea nitrogen is quickly distributed throughout the living complex, becomes a part of the nutrient budget, and is cycled within the ecosystem.

$$CO(NH_2)_2\,[solid] \xrightarrow{H_2O} CO\,(NH_2)_2\,[solution]$$

$$CO\ (NH_2)_2 + 2H_2O \xrightarrow{urease} (NH_4)_2\,CO_3$$

$$(NH_4)_2\,CO_3 \xrightarrow{H_2O,\ CO_2} 2\,NH_4HCO_3$$

Entry Of Fertilizers Into The Aquatic Environment

Fertilizer chemicals may enter the aquatic environment by one of several routes. Direct application of chemicals to exposed surface water is the most important way. This can be minimized by carefully marking and avoiding larger streams during applications, but it is usually impractical to avoid small headwater streams, which frequently are intermittent and difficult to see from the air. Exposed surface water may absorb ammonia nitrogen that has volatized from the forest floor into the air. It is doubtful, however, that this source adds significant amounts to the streams.

Overland flow, or surface runoff, is a major source of nutrients in streams draining nonforested areas, but it is not an important route for fertilizers from treated forest watersheds to enter streams since surface runoff rarely occurs. Subsurface drainage is another possible way soluble forms of nitrogen enter into streams. Forest soils are excellent filters for most plant nutrients because of their high exchange capacities and dense root systems which can absorb and recycle nutrients (Moore 1970). However, measurable levels of ammonium-, nitrate-, urea-, and organic-nitrogen have been found in several streams that were monitored for water quality in western Oregon and Washington.

There is an enormous amount of literature concerning the effects of farm fertilization on water quality, but only a few papers concerning the effects of forest fertilization. Soileau's (1969) extensive bibliography (701 entries) on effects of fertilizers on water quality contains no references on effects of forest fertilization.

Several forest fertilization projects have been monitored recently and examples of the data obtained are presented in appendix XI.B. Data from one study conducted in the Pacific Northwest are discussed below to illustrate the magnitude and pattern of nutrient loss to streams. Measures that may be used to minimize the potential for stream contamination are also indicated.

Moore (1971) measured the amounts and forms of nitrogen entering streams during and following aerial application of 200 lbs/ac of nitrogen (as urea) to an experimental watershed in southwestern Oregon in March 1970 (fig XI.10). Data obtained during the first 15 weeks after application are summarized in table XI.5. Urea concentrations increased slowly and reached a maximum of 1.39 mg/l urea-N 48 hours after application started. Ammonium-N increased slightly above pretreatment level, but never reached 0.10 mg/l. Nitrate-N began to increase slowly the second day, reached 0.168 mg/l in 72 hours, and was 0.140 mg/l at the end of 2 weeks. Nitrite-N was not detected and wouldn't be expected to occur in well aerated streams.

All urea losses of applied nitrógen occurred during the first 3 weeks. Losses in the form of ammonium-N, even though small, continued for 6 weeks. During the first 9 weeks after application, net loss of applied nitrogen amounted to only 1.81 kilograms from watershed 2 (table XI.6).

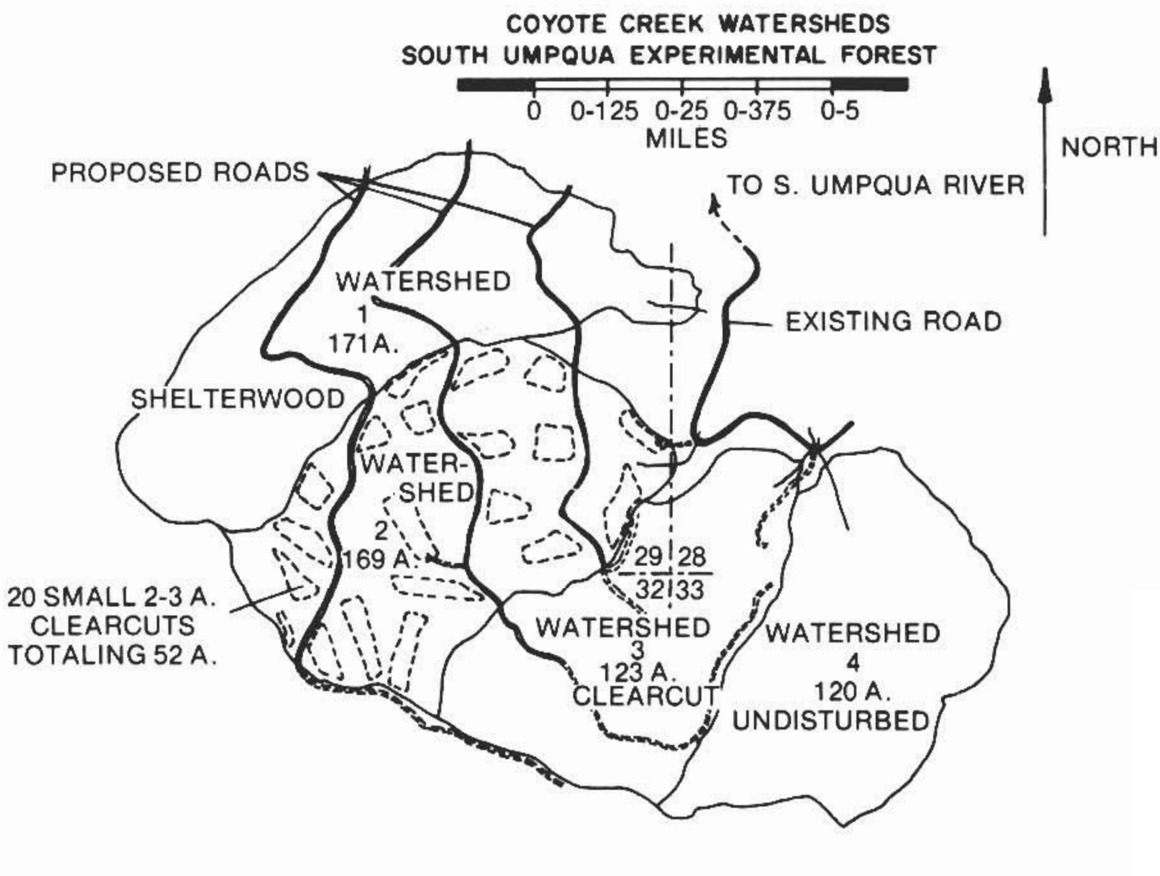


Figure XI.10.—Coyote Creek watersheds, South Umpqua Experimental Forest, Umpqua National Forest, Oreg. (Moore 1971).

Table XI.5.—Concentration of fertilizer nitrogen in selected water samples collected at watershed 2, South Umpqua Experimental Forest, following application of 200 pounds urea-N/ac (Moore 1971)

Date	Time	Urea-N	NH_3N[1]	NO_3-N	Total
		mg/l			
3/25	0800	.007	.001	.002	.010
3/26	0815	.437	.016	.040	.493
	1230	.237	.012	.069	.318
	2025	.171	.034	.067	.272
3/27	0805	1.389	.048	.107	1.544
	1640	.606	.036	.150	.792
	2005	.488	.029	.168	.685
3/28	0805	.075	.036	.117	.228
4/1	---	.007	.016	.091	.185
4/8	---	.028	.015	.140	.183
4/15	---	0	.010	.030	.040
4/22	---	0	.010	.021	.031
5/6	---	0	.013	.022	.035
5/27	---	---	0	.004	.004
6/17	---	---	0	.002	.002
7/8	---	---	0	.006	.006

[1]Includes both ionized (NH_4^+) and un-ionized (NH_3) ammonia-nitrogen

Table XI.6.—Nitrogen lost from treated watershed 2 and untreated watershed 4, South Umpqua Experimental Forest, during the first 9 weeks after application of 224 kilograms urea-N/ha (Moore 1971)

Unit	Urea-N	NH_3-N	NO_3-N	Total
	Kilograms N			
Watershed 2	0.65	0.28	1.01	1.94
Watershed 4	0.02	0.06	0.05	0.13
Net loss	0.63	0.22	0.96	1.81
Percent of total loss	34.75	12.25	53.00	100.00

Low streamflow caused by limited precipitation throughout the summer and fall months resulted in essentially no loss of applied nitrogen during the next 24 weeks. Storm activity in November brought the soil moisture level back to maximum storage capacity. In December the nitrate-N concentration in samples for the fertilized watershed reached a second peak of 0.177 mg/l (fig. XI.11). Both streamflow and nitrate-N levels remained high throughout December and January, resulting in the loss of an additional 23.8 kg applied nitrogen. This second peak accounted for 92 percent of the total amount of fertilizer nitrogen which was lost during the first year.

Total net loss of applied nitrogen from the fertilized watershed (68 ha) during the first year amounted to 25.85 kg, or 0.38 kg of nitrogen/ha (table XI.7). Over the same period the total amount of soluble inorganic nitrogen lost from the control watershed (49 ha) was 2.15 kg, or 0.04 kg nitrogen/ha. Data for soluble organic nitrogen, total phosphorus, silica, and exchangeable cation content of the stream samples, including sodium, potassium, calcium, magnesium, iron, manganese, and aluminum, indicate that there was no apparent effect of nitrogen fertilization on loss of native soil nitrogen or other plant nutrients. Movement may have occurred in the soil profile, but there was no measurable change in stream water quality.

Initial losses of applied nitrogen were largely caused by direct application of urea fertilizer to the drainage channel. These losses were measured first as an increase in urea-nitrogen and then as a small increase in ammonium-nitrogen, the latter as a result of hydrolysis of urea applied to open water. The nitrate-nitrogen entering the stream shortly after application was probably leached from the soil immediately adjacent to the stream channel.

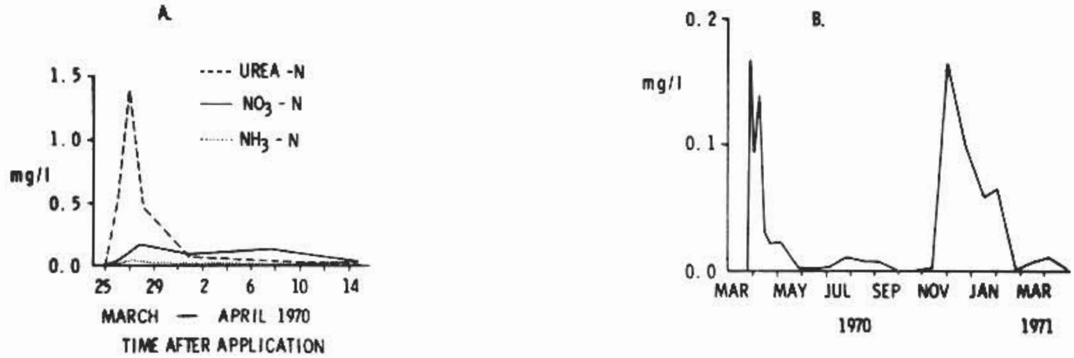


Figure XI.11.—Fertilization of a 68-ha watershed with 224 kg urea-nitrogen/ha in March 1970. A. Immediate effect on water quality; B. Effect on nitrate-nitrogen concentration in streamflow for 1 year following fertilization (Fredriksen and others 1975).

During the first 9 weeks after application, approximately half of the applied nitrogen was lost through direct application and half entered the stream as nitrate-nitrogen. However, all of the applied nitrogen lost during this 9-week period amounted to only 7 percent of the total loss that occurred over the first year.

High streamflow coupled with the second peak in nitrate-nitrogen levels during the winter storm period accounted for 92 percent of the total loss. In February and March 1971, streamflow remained high, but most of the mobile nitrogen had already been lost, and nitrate-nitrogen concentrations had returned to near normal.

Similar data have been obtained in each of the monitoring studies that have been conducted in the Douglas-fir region and elsewhere. The length of the monitoring period has varied from a few weeks following treatment to 6 or 7 months, and in a few studies monitoring continued for at least a full year. Sampling usually continued until the forms of nitrogen being measured decreased to near pre-treatment levels. Increases in the concentration of urea-N are almost entirely caused by direct application to surface waters, and the peak concentration reached is directly proportional to the amount of open surface water in the treated unit. Peak concentrations above 5.0 mg/l are in every case associated with projects where no buffer strips were left along the main streams; or where fertilizer application was carried out early in the spring, when the drainage system was greatly expanded by spring runoff of snowmelt. Even when buffer strips of 30 to 90 m are left along main streams and tributaries, some direct application to water surfaces still will occur because of a relatively dense network of small feeder tributaries that are only a foot or two wide and cannot be identified from the air.

Peak concentrations of urea-N do not persist for more than a few hours. Concentrations characteristically reach a peak each day that fertilizer is being applied and then drop rapidly back toward pre-treatment levels. Within 3 to 5 days after application is completed, levels of urea-N in the stream have returned to pre-treatment concentrations.

Table XI.7.—Nitrogen lost from treated watershed 2 and untreated watershed 4, South Umpqua Experimental Forest, during the first year after application of 224 kilograms urea-N/ha (Moore 1971)

Unit	Urea-N	NH_3-N	NO_3-N	Total
	kilograms N			
Watershed 2	0.65	0.28	27.09	28.03
Watershed 4	0.02	0.06	2.07	2.15
Net loss	0.63	0.22	25.02	25.88
Percent of total loss	2.44	0.86	96.70	100.00

Ammonium-N levels also increase as a result of direct application of urea-N to open water. Urea is readily hydrolyzed to ammonium-N in the aquatic system. Urea applied to the forest floor and soil will not reach the stream since it hydrolyzes rapidly to ammonium carbonate and is held on cation exchange sites in the soil and forest floor like any other salt. Concentrations of ammonium-N in the stream are rapidly reduced through uptake by aquatic organisms and by adsorption on stream sediments. Levels in the streams sampled exceeded 1.00 mg/l only when direct application of urea to the stream was noted. Peak concentrations are normally 0.10 mg/l or less and do not persist for more than a few hours, but levels of ammonium-N remain slightly above pre-treatment level for up to 3 and 4 weeks.

The peak concentration of nitrate-N in stream samples usually occurs from 2 to 4 days after fertilization. Magnitude of the peak concentration depends on whether buffer strips are left along the main stream channels, the width of the waterside area, and the density of small feeder and tributary streams in the drainage system of the fertilized area. Peak concentrations of nitrate-N are generally below 1.0 mg/l, but higher levels have been measured in a few studies. Concentrations usually decrease rapidly after the peak is reached, but remain above pre-treatment level for 6 to 8 weeks. In monitoring studies where sampling has continued through the first winter following fertilization, additional peaks in the concentration of nitrate-N have been measured. These peaks usually coincide with the more intense winter storms, and the concentration drops sharply between storms. Maximum concentrations measured are still low and tend to decrease with each successive storm.

Losses of applied nitrogen are usually very small because the maximum concentrations are generally low, and streamflow decreases rapidly with the onset of the growing season. Following spring application, about half of the applied nitrogen entering the stream during the first 30 days is from direct application and is measured as urea-N and ammonium-N; the other half enters as nitrate-N. All subsequent losses of applied nitrogen to the stream enter as nitrate-N. During early fertilization projects, where buffer strips were either inadequate or not used, estimated total loss was between 2 and 3 percent of the applied nitrogen. In later projects, where direct application to the open surface waters has been avoided or minimized by buffer strips along the main streams and tributaries, measured amounts of applied nitrogen entering the stream are less than 0.5 percent.

Increased phosphorous concentrations following application of phosphate fertilizers have not been reported. Phosphorus added to forest soils is readily utilized by forest organisms or is rapidly converted to nonsoluble forms. Powers and others (1975) have stated that most forest soils have the capacity to tie up, in nonmobile form, many times the quantity of phosphate that foresters are likely to apply. There have been no reports of significant increases in phosphorous concentration in streams following fertilizer application.

Summary Of Fertilizer Entry Into The Aquatic Environment

The most important mechanism of fertilizer entry into the aquatic environment is direct application to open surface waters. Numerous studies (appendix B) have shown that the amount of applied nutrients entering streams has resulted in minimal increases in the instream concentrations of nitrogen and phosphorus. When direct application of fertilizer to streams can be reduced or prevented by use of adequate buffer strips and marking of water courses, the potential impact on stream quality can be minimized.

Transport of mobile forms of nitrogen (nitrate-N) to streams by subsurface drainage from the riparian zone during dormant season storms is the second most important mechanism by which fertilizer nitrogen may enter the aquatic system. Again, the use of adequate buffer strips will reduce the potential impact on water quality. Nitrogen that does enter the stream is rapidly decreased through utilization by biological communities in the stream. Concentrations are further reduced by dilution with downstream movement. Studies conducted to date indicate that forest fertilization will not result in degradation of water quality to the detriment of other resources. With only one exception, none of the studies have recorded nitrogen concentrations that approach the Public Health Service maximum permissible levels for drinking water (Moore 1971, Hornbeck and Pierce 1973, Moore 1975b, Sopper 1975, Norris and Moore 1976, Newton and Norgren 1977).

Behavior In The Aquatic Environment

Forest fertilizers properly applied to an entire watershed undoubtedly will change the nutrient balance among soil, vegetation, animal life, and water in the forest ecosystem, but should pose little or no threat to water quality (Cole and Gessel 1965). Fertilizers applied directly into streams, however, do represent a potential problem, and the total impact of the introduced chemicals will depend on their behavior in the aquatic environment.

When urea nitrogen is introduced into small streams of forested watersheds, either from wildlife activity or through aerial application of fertilizers, it disappears rapidly and only traces can normally be detected in undisturbed ecosystems. Urea is hydrolyzed to ammonium nitrogen by urease enzyme adsorbed on suspended solids and bottom sediments. Ammonium nitrogen may remain in solution or be adsorbed by suspended organic and inorganic colloids and bottom sediments. All forms of nitrogen are diluted by downstream movement caused by natural stream mixing and increased flow volume from side streams and ground water. Dissolved inorganic and organic nitrogen may also be removed by aquatic organisms to such an extent that they are undetectable at a downstream sampling point (Thut and Haydu 1971).

Phosphorus is not considered a mobile element in the soil system. Even those forms of phosphorus that are readily available for plant uptake are not subject to leaching to any significant extent. Phosphate fertilizer applied to a forest watershed would not be expected to enter the stream system except by direct application. Since most headwater streams in relatively undisturbed forest watersheds contain only low concentrations of phosphorus, the small amounts of phosphorus added during a normal fertilization program would be rapidly utilized by the biological community in the stream. Many of the streams in forested areas of the Douglas-fir region are nutrient deficient, and it has been suggested that forest fertilization may have a beneficial effect on forest stream productivity (Thut and Haydu 1971).

The fate of nitrogen applied to cultivated crops has been studied extensively (Allison 1966), but only limited data are available on the nitrogen cycle in temperate forests (Cole and others 1967, Weetman 1961). The output of nitrogen in drainage from actively growing forest stands appears to nearly balance inputs in precipitation (Cooper 1969). Since stream enrichment resulting from forest fertilization is apparently small and of short duration, it can be assumed that any deleterious effects that do occur will not persist. However, the effect of small additions at upstream sites on accumulation of nutrients in downstream impoundments must be considered.

CONCLUSIONS

The amount of a particular chemical that enters a stream will vary depending on many of the factors discussed in this chapter. Each of the components of the forest environment indicated in figure XI.2 can be designated as a compartment in a systems diagram or conceptual model and the various processes responsible for transformation or movement of chemicals within or between compartments identified. With an adequate data base for any given site and a thorough knowledge of the controlling processes, one could then predict the extent of non-point source pollution that would be expected as the result of using a silvicultural practice that includes the application of a pesticide or fertilizer chemical. Although much is known about the behavior of chemicals in the environment, we still lack a precise mathematical model that will meet this objective. Therefore, the major routes of entry of chemicals into forest streams have been identified, and the processes which are involved within each environmental compartment are identified and discussed primarily from a conceptual and qualitative basis. This framework should provide a logical basis for understanding the mechanisms and processes which may result in non-point source contamination of stream water in a qualitative way even though quantitative estimates are not yet possible.

Based on research experience, history of use, consideration of the manner in which most chemical application operations are conducted, and an analysis of the chemical and physical properties which influence the behavior of chemicals in the environment, it is estimated that the following concentrations of various chemicals may be encountered in the aquatic environment near treatment areas.

Herbicides. — A strong background of research experience permits prediction with confidence that concentrations of 2,4-D, picloram, 2,4,5-T, and amitrole exceeding 0.05 mg/l will seldom be encountered in streams adjacent to carefully controlled forest spray operations. Concentrations exceeding 1 mg/l have never been observed and are not expected to occur. The chronic entry of these herbicides into streams for long periods after application does not occur.

Insecticides. — Concentrations of carbamate insecticides exceeding 0.1 mg/l will rarely be found in forest streams. Carbamate and pyrethrum insecticides do not persist in the environment and they offer little opportunity for movement to streams. The organophosphorous insecticide, malathion, is rapidly degraded in soil and water and enters water only by stream channel interception and limited streamside surface runoff. Ultra-low-volume aerial applications will rarely produce more than 0.5 mg/l malathion in streams.

Fertilizers. — There is still only a limited history of field use and research experience concerning the behavior and fate of fertilizer nitrogen introduced into the aquatic environment as a result of forest fertilization. Available data suggest, however, that concentrations of the various forms of nitrogen found in streams adjacent to treated units are well below accepted standards for public water supplies. The impact of these introduced chemicals on various elements of the ecosystem must be investigated.

Direct application to surface waters is the major source of aerially applied forest chemicals in the aquatic environment. Drift is another important pollution source with pesticides, but not with fertilizer. Careful selection of chemicals, carriers, and equipment and control of the manner in which the project is conducted can materially reduce both the direct application and the drift of chemicals to streams. Specific control opportunities were described in Chapter II. Volatilization, adsorption, degradation, and downstream movement of residues will minimize the exposure time of aquatic organisms to chemicals which do enter the aquatic environment.

The forest manager has no control over the inherent toxicity of a selected chemical, but the hazards of chemical use to nontarget organisms can be minimized by limiting their exposure to biologically insignificant doses. Research experience and history of use have established that important forest chemicals offer minimum potential for pollution of the aquatic environment when they are used properly. The key to proper use is an understanding of the ways which chemicals can enter streams and an appreciation of the factors which influence the degree to which these mechanisms operate.

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APPENDIX XI.A
WATER QUALITY DATA — PESTICIDE CHEMICALS

Table XI.A.1.—Cascade Creek Unit, Alsea Basin, western Oregon (Norris 1967)

Sample point 3[1]		Sample point 4		Sample point 5	
Hours after spraying	2,4,5-T	Hours after spraying	2,4,5-T	Hours after spraying	2,4,5-T
	µg/l		µg/l		µg/l
0.05	0	0.17	1	0.27	lost
0.62	16	1.33	2	1.40	3
1.28	7	2.2	1	2.0	3
2.0	4	3.9	1	3.9	0
4.0	4	5.4	0		
5.2	4				
9.8	4				
24.7	2				
48.2	1				
[2]74.8	1				

[1]Entire watershed feeding the sampled stream was sprayed.
[2]Herbicide was detected for 16 weeks at sample point 3.

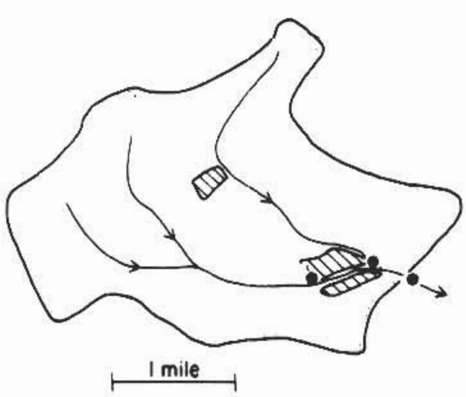

Figure XI.A.1.—Cascade Creek Treatment Unit. (26 ha (2%) of a 1400-ha watershed was treated with 2.24 kg/ha 2,4,5-T. Large streams not included in treatment area.) (Norris 1967).

Table XI.A.2.—Eddyville Unit, Yaquina Basin, western Oregon[1] (Norris 1967)

Sample point 12		Sample point 13		Sample point 14	
Hours after spraying	2,4-D	Hours after spraying	2,4-D	Hours after spraying	2,4-D
	μg/l		μg/l		μg/l
0.83	33	1.33	62	1.38	30
1.83	13	2.3	71	2.3	44
2.8	13	3.3	58	3.3	25
[2]53.5	9	4.3	44	4.3	23
		[2]53.6	25	[2]53.6	11

[1]Rate of application was 2.5 to 3.36 kg/ha.
[2]No further residues detected although sampling continued for 10 months.

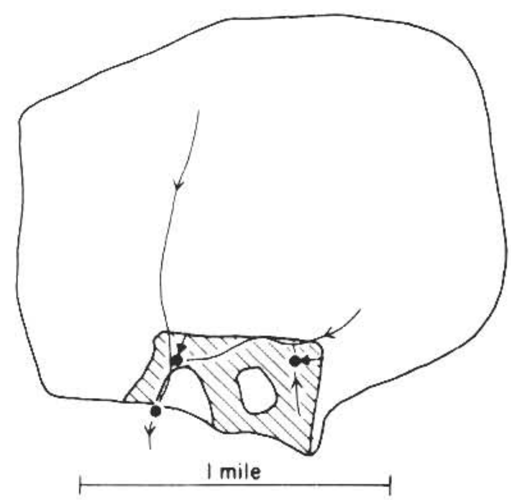

Figure XI.A.2—Eddyville Treatment Unit. (20 ha (10%) of a 287 ha watershed was treated with 2,4-D (LVE) at rates ranging from 2.5 to 3.36 kg/ha. Sampled streams flowed from or through treatment area.) (Norris 1967).

Table XI.A.3.—Concentration of 2,4-D in West Myrtle Creek, Malheur National Forest, eastern Oregon[1] (Norris 1967)

Sample point 1		Sample point 2[2]	
Hours after spraying	2,4-D	Hours after spraying	2,4-D
	μg/l		μg/l
1.7	132	2.0	0
3.7	61	3.9	0
4.7	85	5.0	0
6.0	10	6.2	2
7.0	26	7.2	7
8.0	75	8.2	8
9.0	59	9.2	13
13.9	51	14.1	14
26.9	3	17.0	7
37.9	9	38.0	6
78.0	8	77.8	9
80.8	1	81.0	9
168.0	0	104.8	3
		168.0	1

[1]Rate of application was 2.24 kg/ha.
[2]Sampling point 2 is 1.6 km downstream from point 1.

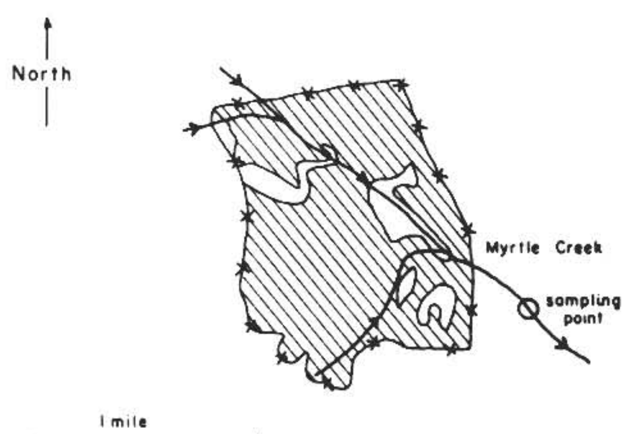

Figure XI.A.3.—West Myrtle Treatment Unit. (240 ha treated in one block. Live streams included in the treatment area.) (Norris 1967).

Table XI.A.4.—Camp Creek Spray Unit, Malheur National Forest, eastern Oregon[1] (Norris 1967)

Hours after spraying	2,4-D
	µg/l
0.1	0
2.0	25
5.4	1
8.8	1
84.5	3
168.0	0

[1]Rate of application was 2.24 kg/ha.

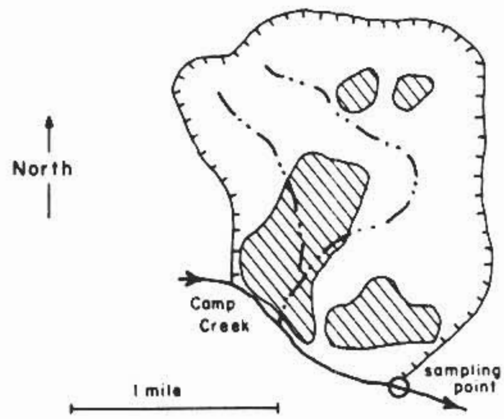

Figure XI.A.4.—Camp Creek Spray Unit. (121 ha treated with 2.24 kg/ha 2,4-D (low volatile esters). Spray boundaries adjacent to, but did not include, live streams.) (Norris 1967).

Table XI.A.5.—Concentration of 2,4-D in streams in Keeney-Clark Meadow eastern Oregon[1] (Norris 1967)

Hours after spraying	2,4-D	Hours after spraying	2,4-D
	µg/l		µg/l
0.7	840	14.3	113
2.5	48	37.8	91
3.1	128	56.4	76
3.6	106	100.1	115
4.1	106	103.6	95
6.1	121	289.9	5
8.1	176	297.0	7
9.6	138		

[1]Rate of application was 2.24 kg/ha.

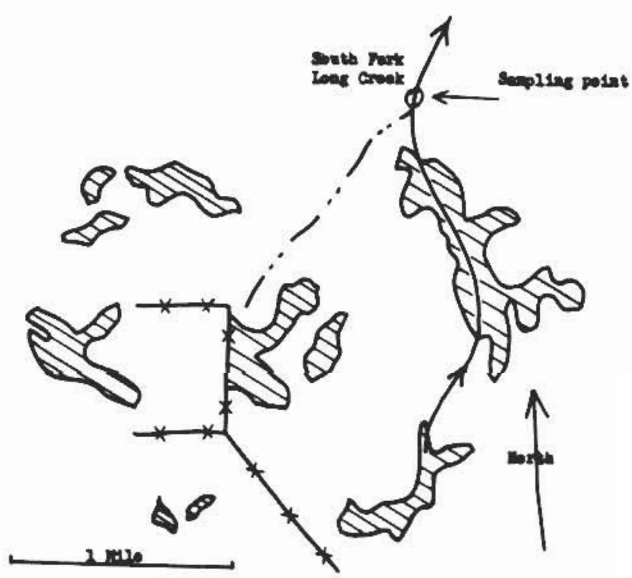

Figure XI.A.5.—Keeney-Clark Meadow Spray Units. (89 ha treated with 2.24 kg/ha 2,4-D. Flat, marshy area with many small live streams and other sites with standing water.) (Norris 1967).

Table XI.A.6.—Concentration of Amitrole-T in Wildcat Creek, Coast Range, western Oregon[1] (Norris and others 1966)

Sample point 2		Sample point 3	
Hours after spraying	**Amitrole-T**	**Hours after spraying**	**Amitrole-T**
	μg/l		μg/l
0.05	1	0.05	0
0.39	30	0.33	0
0.74	35	0.67	9
1.13	37	1.07	90
1.43	17	1.38	110
1.73	16	1.60	40
2.1	19	2.0	35
3.3	21	2.8	24
4.8	12	4.2	14
5.8	8	5.2	7
7.1	5	6.9	5
8.1	4	8.0	5
9.5	3	10.3	3
10.4	2	15.2	2
15.3	1	20.5	25
26.1	7	26.0	8
30.1	4	45.7	3
46.1	2	69.4	0
71.5	0		

[1]Rate of application was 2.24 kg/ha.

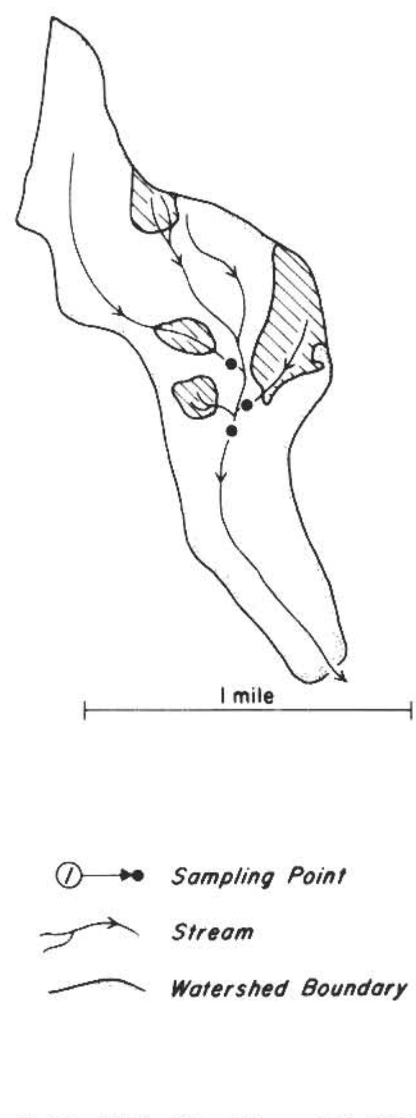

Figure XI.A.6.—Wildcat Creek Spray Unit. (28 ha treated with 2.24 kg/ha amitrole-T. Spray units include live streams.) (Norris and others 1966).

Table XI.A.7.—Concentration of amitrole in stream water, loss or dilution with downstream movement. Amitrole-T applied to 105 ha at 2.24 kg/ha[1] (Norris and others 1967)

Hours after spraying	Amitrole concentration on sampling point			
	1	2	3	4
hours	μg/l			
0.1	1	0	0	0
0.5	5	0	0	0
1	7	2	0	0
2	45	42	0	0
3	24	15	0	0
4	8	18	4	0
5	10	5	6	0
6	9	5	6	0
8	3	3	12	0
10	2	2	2	0
12	1	1	2	0
14	1	1	2	0
24	1	2	1	0
35	1	0	1	0
48	0	0	0	0
72	0	0	0	0

[1]Study was conducted in Coast Range of Oregon. Sampling point 1 was located just below boundary of sprayed unit; point 2 was 3.2 km downstream from point 1; point 3 was 0.48 km below point 2; and point 4 was 1.49 km below point 2. No detectable quantity of amitrole was found between 3 and 150 days after treatment.

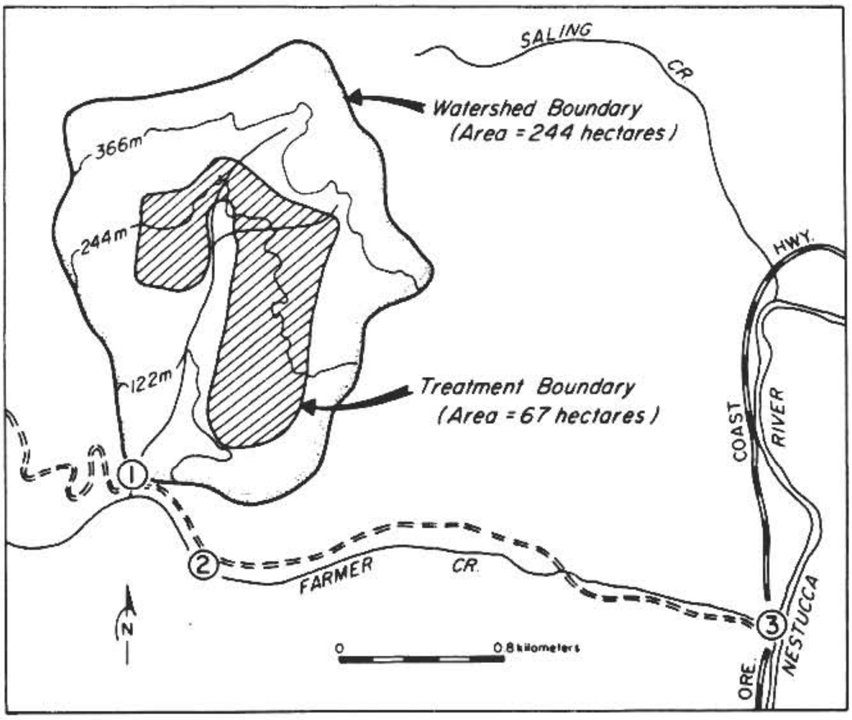

Figure XI.A.7.—Farmer Creek Treatment Watershed. (67 ha of a 244 ha watershed sprayed by helicopter with 1.12 kg dicamba and 2.24 kg 2,4-D per ha. Sampling point 1 is about 1.3 km from edge of treated unit) (see table XI.A.8) (Norris and Montgomery 1975)

Table XI.A.8.—Concentration of dicamba in Farmer Creek[1] (Norris and Montgomery 1975)

Sampling date	Hours after application	Dicamba	Sampling date	Dicamba
	hours	µg/l		µg/l
6/05/71	(prespray)	0	6/10/71	2
6/07/71	0.3	0	6/11/71	4
	0.6	0	6/13/71	9
	1.0	0	6/16/71	0
	1.2	0	6/18/1	2
	1.7	0	6/21/71	0
	2.1	1	6/30/71	0
	2.5	0	7/08/71	0
	2.7	0	7/09/71	0
	3.3	3	8/11/71	0
	3.8	12	8/20/71	0
	4.3	16	8/25/71	0
	4.8	28	9/01/71	0
	5.2	37	9/02/71	0
	6.2	33	9/07/71	0
	6.8	30	9/29/71	0
	7.8	27	10/19/71	0
	8.8	24	11/17/71	0
	10.2	16	11/29/71	0
	13.1	11	12/22/71	0
	22.8	6	5/18/72	0
6/08/71	30.1	2	6/08/72	0
	37.5	0	6/30/72	0
6/09/71	50.2	0	7/28/72	0

[1]Coastal Oregon; 67 ha treated with 1.12 kg/ha dicamba and 2.24 kg/ha 2,4-D.

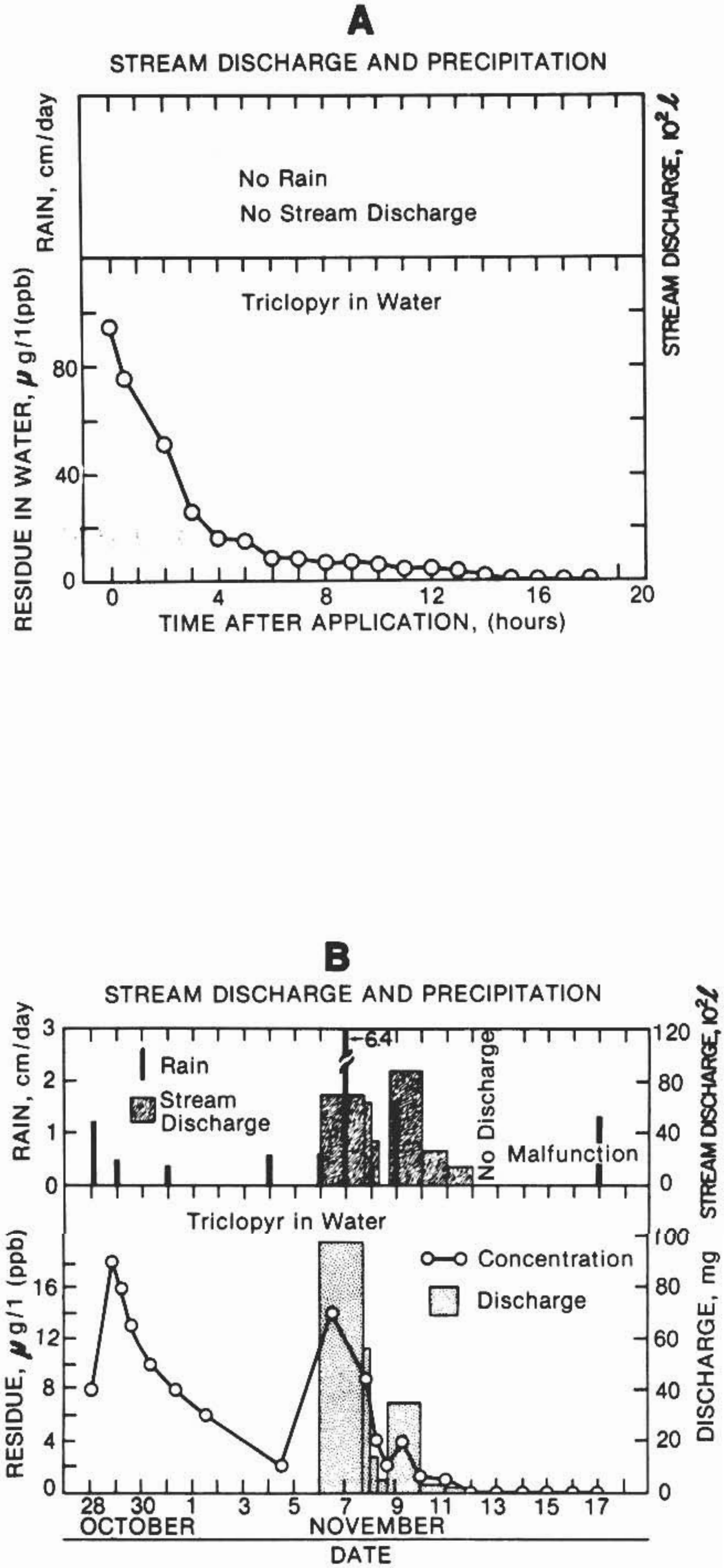

Figure XI.A.8—Precipitation, stream discharge, and concentrations of tryclopyr in stream water following application of 3.36 kg/ha by helicopter to a small watershed in southwest Oregon in May 1974 (Norris and others 1976b).

A. First 20 hours after application.

B. First significant storm activity, channel flushing.

Table XI.A.9.—Concentrations of 2,4-D and picloram in drainage waters from a 7-ha hill-pasture watershed in southwest Oregon[1] (Norris and others 1976a)

Date	Rain	2,4-D	Picloram
	cm.	µg/l	
9/18/69		0	110
10/09/69	7.9	22	43
10/13/69		0	64
10/21/69	3.0	3	39
11/14/69	5.0	0	0
11/24/69	---	0	0
12/01/69	0.1	0	0
12/09/69	2.0	0	0
12/19/69	6.8	0	0
12/24/69	9.9	0	12
1/01/70	4.6	0	1
1/24/70	18.6	0	0

[1]Rate of application—2.3 kg picloram and 4.6 kg 2,4-D in 93.5 l/ha applied as Tordon 212 by helicopter.

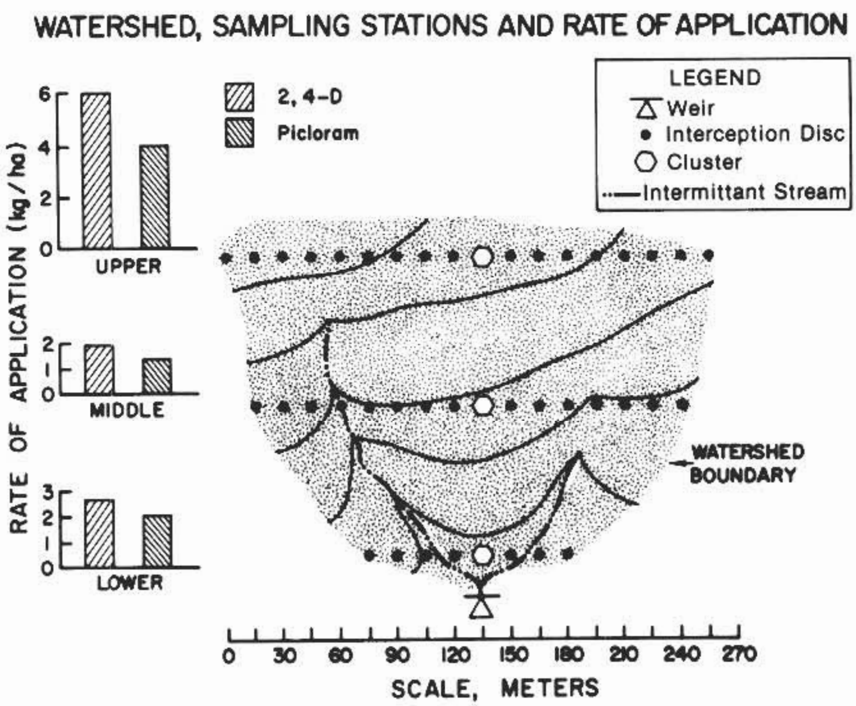

Figure XI.A.9.—Boyer Ranch, southwest Oregon. Small 7-ha hill-pasture spray unit treated with Tordon 212 (Norris and others 1976a).

Table XI.A.10.—Total DDT content of stream water flowing from sprayed area — before treatment and for 3 years after treatment[1] (Tarrant and others 1972)

Date	Days after spraying	Total DDT residues in Rattlesnake Creek	
		East Fork	West Fork
		------------- µg/l -------------	
5/24/65	-30	[2]	
6/19/65	- 4	ND	ND
6/23/65	1	.104	.277
7/14/65	21	.031	.022
8/26/65	64	.028	.015
11/17/65	147	.014	ND
6/07/66	349		ND
7/19/66	391	.010	
11/09/66	505	ND	
7/04/67	742	ND	ND
11/07/67	869	.032	.010
7/16/68	1,131		
11/12/68	1,251	.010	

[1]Area sprayed with DDT at rate of 0.84 kg/ha.
[2]Blank = levels of DDT isomers and metabolites less than 0.01 mg/l but greater than 0.002 mg/l.
ND = not detected

HERBICIDE DISCHARGE

	Total Applied	Discharged Total	Discharged %	Area in Stream Channel % of Total
2,4-D	29800 g	6 g	0.02	0.21
Picloram	14900 g	43g	0.29	0.21

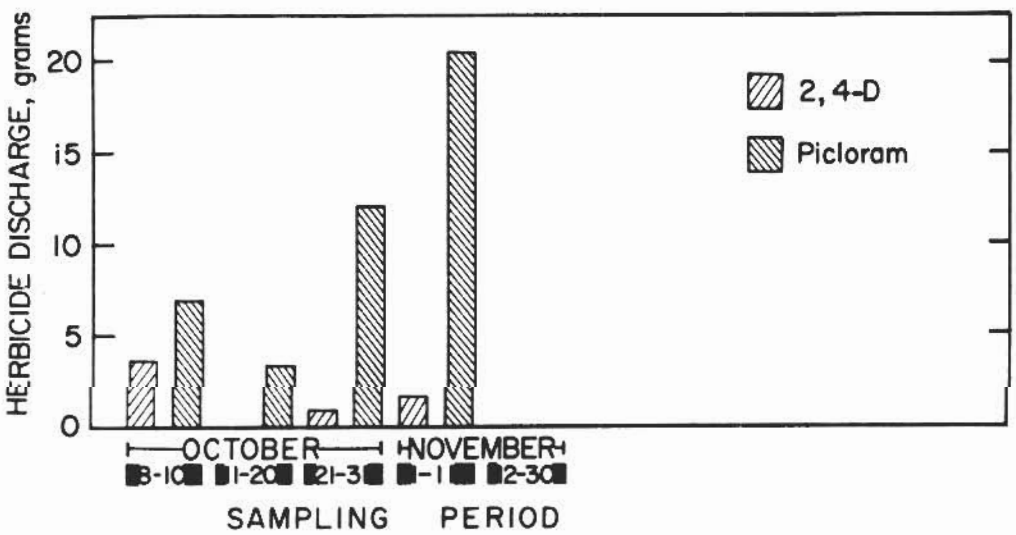

Figure XI.A.10.—Discharge of herbicide in streamflow from small 7-ha hill-pasture watershed, Boyer Ranch, southwest Oregon. Treatment was with Tordon 212 at 2.3 kg picloram and 4.6 kg 2,4-D per hectare (Norris and others 1976a).

Note: All of the herbicide discharged with streamflow is accounted for by the quantity applied to the stream channel and adjacent banks. (The question mark for the period December 21 through 31 reflects equipment malfunction resulting in no measure of stream discharge.)

Table XI.A.11.—Concentration of herbicides in water samples, as determined by odor tests[1] (Reigner and others 1968)

Herbicide and time of sample	Pennsylvania streams	New Jersey streams
	μg/l	μg/l
2,4,5-T butory ethanol ester:		
Immediately after spraying	40	40
4 hours later	20	20
Next 9 samples[2]	ND[3]	ND
After first large storm	10	ND
2,4,5-T emulsifiable acid:		
Immediately after spraying	40	20
4 hours later	10	ND
Next 9 samples[2]	ND	ND
After first large storm	20	ND
All downstream samples (both herbicides)	ND	ND

[1]Test panel used procedure approved by American Society for Testing and Materials.
[2]Samples taken daily for first week; twice a week for next 2 weeks.
[3]ND = no detectable odor.

Figure XI.A.11.—Concentration of endrin in streamflow after aerial seeding with endrin-coated Douglas-fir seed. Needle Branch Watershed—seed treated with 1.0% endrin and sown at 0.84 kg/ha; Watershed 1, H.J. Andrews Experimental Forest—seed treated at 0.5% endrin and sown at 0.56 kg/ha (Moore and others 1974).

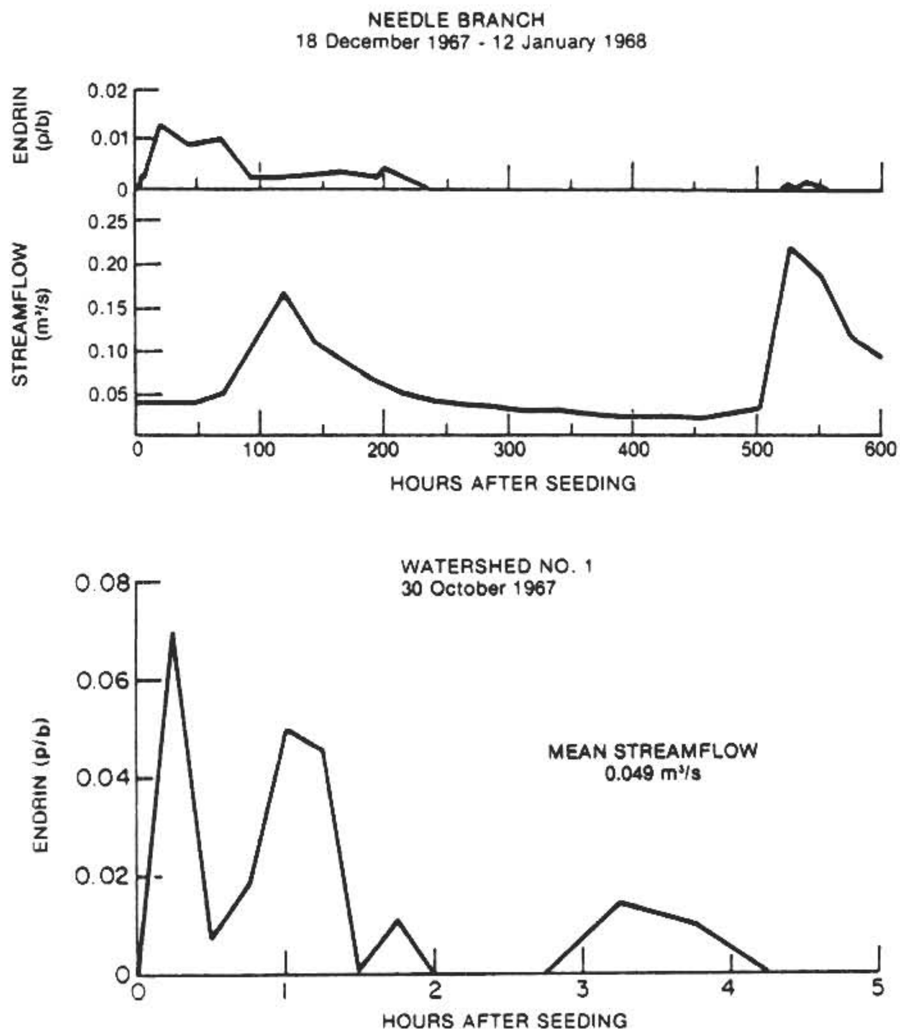

Table XI.A.12.—Concentrations of 2,4-D and 2,4,5-T herbicide in water samples from Monroe Canyon, San Dimas Experimental Forest, northeast of Glendora, California (Krammes and Willets 1964)[1]

Date	Site			
	Weir	Surface	Well 1	Well 2
	ppm			
May 10/61	0.00	---	---	---
May 22/61	.00	---	---	---
June 5/61	.05	0.09	0.01	0.01
July 24/61	.05	.03	.00	.00
July 31/61	.00	.00	.00	.00
Aug. 28/61	.00	.00	.00	.01
Sept. 25/61	.00	.00	.04	.00
Oct. 30/61	.00	.00	.00	.00
Jan. 29/62	.00	---	---	---
Feb. 26/62	.00	---	---	---
June 20/63	.00	---	---	---

[1]The riparian zone and intermediate slopes of a 354-ha watershed were hand sprayed several times with a mixture of equal parts of 2,4-D and 2,4,5-T in diesel oil. Care was taken to avoid any direct contamination of the stream. A total of 170 l of herbicide was applied on May 10, 1961, but actual rates of application are not known. Maintenance spraying was carried out again in June, 1963, also followed by hand spraying at later dates. Stream contamination was below the safe limit of 1 ppm. No traces of diesel oil were found. Riparian zone vegetation was handsprayed during the week following the May 22, 1961 sampling and just before the June 20, 1963 sampling.

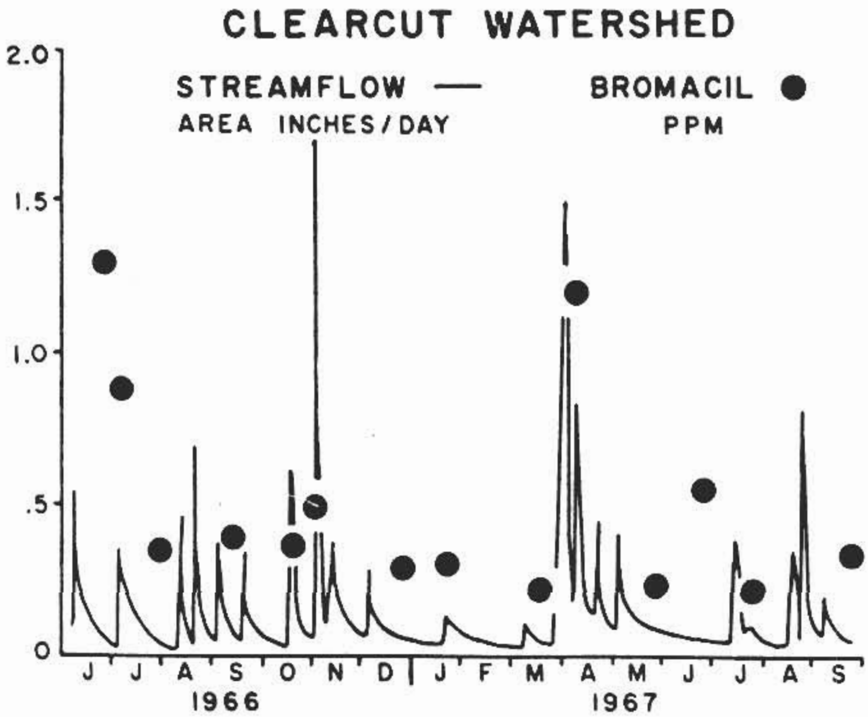

Figure XI.A.12.—Water yield and bromacil release from watershed 2, Hubbard Brook Experimental Forest, West Thornton, New Hampshire (Pierce 1969).

Note: Watershed 2 (15.8 ha) was clearcut of all timber and woody vegetation in late fall and early winter of 1965. In June 1966, bromacil was broadcast sprayed by helicopter at a rate of 28 kg/ha. Persistent sprouts were sprayed with 2,4,5-T in the summer of 1967. About 20 percent of the bromacil left the watershed through the stream in 1½ years. The concentration of 2,4,5-T in the stream was less than 1 mg/l for the entire period following application.

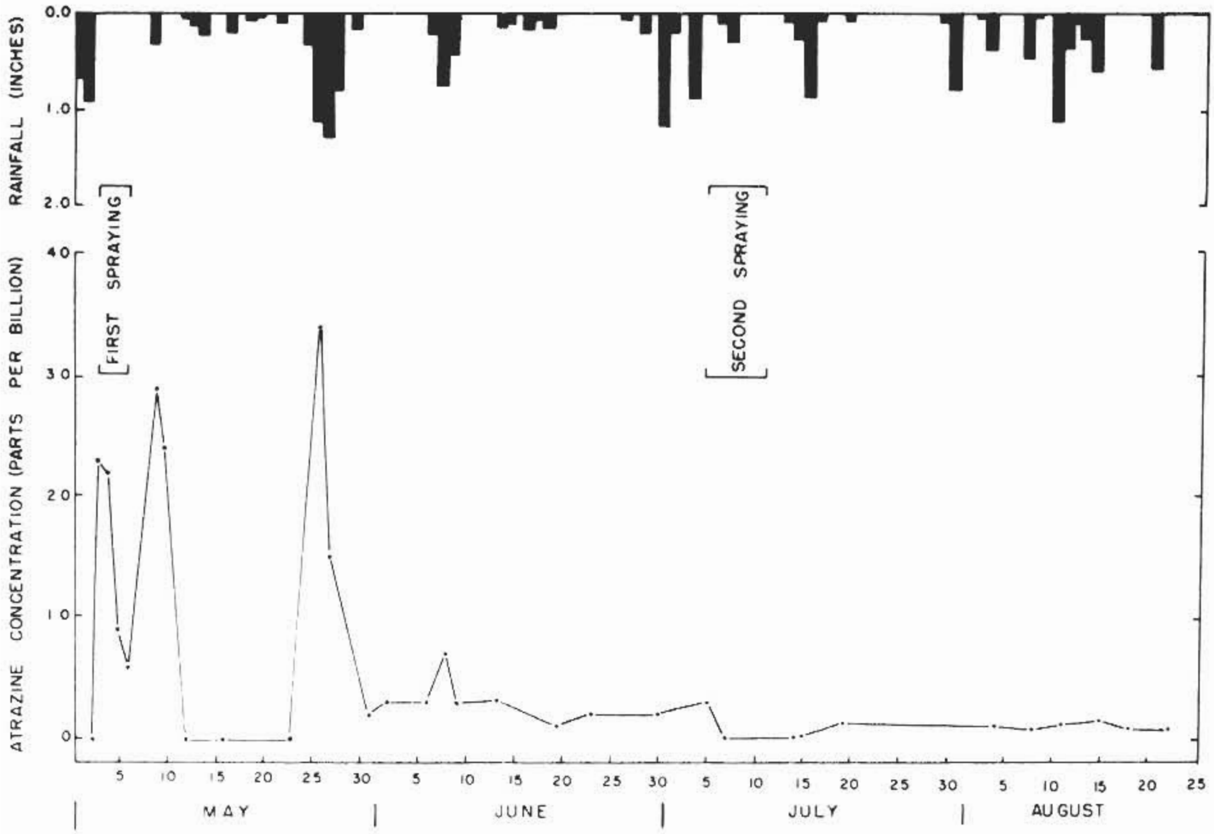

Figure XI.A.13.—Atrazine concentration in streamflow during and for 3½ months after herbicide treatment (Douglas and others 1969).

Note: A 9-ha watershed was treated May 3-6, 1966, with 3.9 kg atrazine and 0.95 l technical paraquat per hectare, including the water course. Surviving vegetation was sprayed again on July 5-11 with a mixture of 3.36 kg 2,4-D (isobutyl esters) and 5 kg atrazine per hectare, but a 3-m buffer strip was left unsprayed on both sides of the stream. Atrazine content in water samples from the stream is graphed above. Paraquat was detected in only 5 of more than 35 samples, and maximum concentration measured was 19 $\mu g/l$. After the second spraying, 2,4-D was never detected in the stream and the concentration of atrazine did not increase, even during storms.

APPENDIX XI. B
WATER QUALITY DATA — FERTILIZER CHEMICALS

Table XI.B.1.—Stream water quality following forest fertilization, fall 1975: Hoodsport-Quileene Ranger Districts, Olympic National Forest, Washington (Stephens 1975b)

Treatment: Urea pellets were applied to several thousand acres of second growth Douglas-fir. As a general rule, stream buffer strips of 100 ft (30 m) were left along tributary streams which were flowing greater than 0.5 ft^3/sec (14 l/sec). 300 ft (91 m) wide buffer strips were left along main streams.

Site	Rate of application lb-N/ac	kg-N/ha	Date of application	Treatment area		Range concentrations Urea-N	NH_3-N	NO_3-N
				ac	ha	mg/l		
McDonald Creek	200	224	Oct.-Nov. 75	316	128			
Pre-treatment						0.01-0.02	0	0.03-0.05
Post-treatment						0.32-0.01	0-0.18	0.03-2.85
Jimmycomelately	200	224	Oct.-Nov. 75	48	20			
Pre-treatment						---	---	---
Post-treatment						0-0.05	0-0.07	0.03-0.13
Gold Creek	200	224	Oct.-Nov. 75	229	93			
Pre-treatment						0	0	0.02-0.05
Post-treatment						0-0.31	0-0.22	0.02-0.18
Elbo Creek	200	224	Oct.-Nov. 75	33	13			
Pre-treatment						0	0	0.01-0.02
Post-treatment						0-0.28	0-0.10	0-0.07
Mile & ½ Creek	200	224	Oct.-Nov. 75	169	68			
Pre-treatment						0-0.02	0	0.06-0.07
Post-treatment						0-0.22	0-0.02	0-0.92
Fulton Creek	200	224	Oct.-Nov.75	592	240			
Pre-treatment						0	0	0.01-0.02
Post-treatment						0-0.13	0-0.10	0.01-0.09
Waketickeh Creek	200	224	Oct.-Nov. 75	1432	580			
Pre-treatment						0-0.01	0	0-0.02
Post-treatment						0-0.84	0-0.55	0-0.40

Table XI.B.2.—Stream water quality following forest fertilization, spring 1975: Hoodsport-Quileene Ranger Districts, Olympic National Forest, Washington (Stephens 1975a)

Treatment: Urea pellets were applied by helicopter to several thousand acres of second growth Douglas-fir. As a general rule, stream buffer strips 200 ft (60 m) wide were left along streams which were flowing greater than 0.5 ft³/sec (14 l/sec).

Site	Rate of application		Date of application	Treatment area		Range concentration
	lb-N/ac	kg-N/ha				NO_3-N
				ac	ha	---mg/l---
Mile & ½ Creek	200	224	Apr. 75	292	118	
Pre-treatment						0.01-0.03
Post-treatment						0-0.18
Trapper Creek	200	224	Apr. 75	200	81	
Pre-treatment						-0.03
Post-treatment						0.01-0.54
Salmon Creek	200	224	Apr. 75	112	45	
Pre-treatment						0
Post-treatment						0.03-0.65
Eddy Creek	200	224	Apr. 75	240	97	
Pre-treatment						0
Post-treatment						0-0.72
Jackson-Marple	200	224	Apr. 75	460	186	
Pre-treatment						0-0.01
Post-treatment						0-0.50
Turner Creek	200	224	Apr. 75	286	116	
Pre-treatment						0-0.04
Post-treatment						0-0.25

Table XI.B.3.—Stream water quality following a wildfire and fertilization with reseeding for erosion control, 1971: Entiat Experimental Forest, central Washington (Klock 1971; Tiedemann and Klock 1973; and Helvey and others 1974)

Treatment: Following a wildfire in August 1971, three watersheds were monitored for water quality. Fox Creek was used as a control, Burns Creek was fertilized with ammonium sulfate and McCree Creek was fertilized with urea. An unburned watershed, Lake Creek was also monitored as an undisturbed control.

Site	Rate of application		Dates of appli-cation	Percent of total applied	Treatment area		Peak concentrations		
	lb-N/ac	kg-N/ha					Urea-N	NO_3-N	NH_4-N
					ac	ha	mg/l		
Fox Creek	Control		no application		1,169	473			
Pre-treatment	1970						[1]0.035	N.D.[2]	N.D.
Post-treatment	1971						N.D.	N.D.	N.D.
McCree Creek	48 urea	54	10/30/70	7.5	1,270	513			
			11/05/70	24.3					
			11/08/70	68.2					
Pre-treatment	1970						N.D.	N.D.	N.D.
Post-treatment	1971						0.616	0.210	<0.02
Burns Creek	51 $(NH_4)_2SO_4$	57	10/30/70	13.6	1,394	564			
			11/09/70	86.4					
Pre-treatment	1970						N.D.	N.D.	N.D.
Post-treatment	1971						0	0.068	0
Lake Creek	Control		no application						
	1972							0.065	

[1]Attributed to wildlife activity

[2]N.D.—Not detected, concentration below detection limit of equipment.

Table XI.B.4.—Stream water quality following forest fertilization, 1970: Mitkof Island, southeast Alaska (Meehan and others 1975)

Treatment: Two areas of cutover land were fertilized in May 1970 by helicopter with urea pellets.

Site	Rate of application		Date of application	Treatment area		Urea-N	NO_3-N	NH_3-N
	lb N/ac	kg N/ha						
				ac	ha	mg/l		
Falls Creek								
Control	---	---	---	---	---			
1970						N.D.	0.23	0.23
1971						N.D.	0.24	0.11
Treated	190	210	May 70	---	---			
1970						N.D.	1.26	1.28
1971						N.D.	1.66	0.11
Three Lakes								
Control	---	---	---	---	---			
1970						N.D.	0.20	0.10
1971						N.D.	0.18	0.12
Treated	190	210	May 70	---	---			
1970						N.D.	2.36	0.14
1971						N.D.	0.30	0.08

N.D. = Not Detected

Table XI.B.5.—Stream water quality following forest fertilization of two small watersheds, 1970 and 1971: Siuslaw River Basin, western Oregon (Burrough and Froehlich 1972)

Treatment: Two watersheds, Nelson Creek and Dollar Creek, were fertilized by helicopter with urea pellets. There were no buffer strips established along watercourses within the treated area. Untreated adjacent watersheds were also monitored as a control.

Site:	Rate of application		Date of appli-cation	Treatment area		Peak Concentration		
	lb-N/ac	kg-N/ha				Urea-N	NH_3-N	NO_3-N
				ac	ha	mg/l		
Nelson Creek	200	224	Apr. 70	94	---			
treated						8.6	0.32	7.6
untreated						0.20	0.33	4.3
Dollar Creek	200	224	Apr. 71	85	---			
treated						44.4	0.49	0.13
untreated						<0.02	0.15	0.16

Table XI.B.6.—Stream water quality following fertilization of forested watershed on the Olympic Peninsula, spring 1970: Quileene Ranger District, Olympic National Forest, Washington (Moore 1975b)

Treatment: Two watersheds, Jimmycomelately and Trapper Creek, were fertilized by helicopter with urea. Pelletized or large granule forest grade urea was unavailable so agricultural grade was used. Drift of the fertilizer was noted. The stream was flagged and fertilizer was not applied within 200 ft (60 m) of the stream.

Site:	Rate of application		Date of appli-cation	Treatment area		Peak Concentration		
	lb-N/ac	kg-N/ha				Urea-N	NH_4-N	NO_3-N
				ac	ha	mg/l		
Jimmycomelately	200	224	Apr. 70	120	49			
Pre-treatment						0	<0.004	0.002
Post-treatment						0.71	0.04	0.042
Trapper	200	224	Apr. 70	158	64			
Pre-treatment						0.013	<0.004	0.055
Post-treatment						0.71	0.01	0.121

Table XI.B.7.—Stream water quality after fertilization of a small forested watershed on the west slopes of the Cascade Mountains, 1970: Oregon (Malueg and others 1972)

Treatment: A watershed was fertilized by helicopter with urea pellets. No effort was made to prevent the direct application of urea into the water courses.

Site:	Rate of application		Date of appli-cation	Treatment area		Concentrations		
	lb-N/ac	kg-N/ha				NH_4-N	NO_2-N	NO_3-N
				ac	ha	mg/l		
Crabtree Creek	200	224	May 70	569	230			
Pre-treatment						<0.01	<0.01	<0.01
Post-treatment						<0.08	<0.01	<0.25

Table XI.B.8.—Stream water quality after fertilization following wildfire in north-central Washington, 1970: Chelan, Washington (Tiedemann 1973)

Treatment: Urea fertilization following wildfire. Falls Creek was fertilized, Camas Creek was not fertilized, and Grade Creek was unburned and unfertilized.

Site:	Rate of application		Date of appli-cation	Treatment area		Peak Concentrations		
	lb-N/ac	kg-N/ha				Urea-N	NH_3-N	NO_3-N
				ac	ha	mg/l		
Falls Creek	70	78	Oct. 70	6,180	2,500			
Pre-treatment						0.330	0.011	0.016
Post-treatment						0.029	0.011	0.310
Camas Creek	---	---	---	1,680	680	0.006	0.001	0.042
Grade Creek	---	---	---	6,920	2,800	[1]0.450	0.011	0.016

[1]Attributed to animal activity.

Table XI.B.9.—Stream water quality following forest fertilization, spring 1976: Quileene Ranger District, Olympic National Forest, Wash. (Stephens 1976)

Treatment: Urea pellets were applied to 800 ac of second-growth Douglas-fir. As a general rule, stream buffer strips 100 ft (30 m) wide were left along tributary streams which were flowing greater than 0.5 ft^3/sec (14 l/sec); 300 ft (91 m) wide buffer strips were left along main streams.

Site:	Rate of application		Date of appli-cation	Treatment area		Range Concentrations		
	lb-N/ac	kg-N/ha				NH_3	NO_3	Urea
				ac	ha	mg/l		
Townsend Creek	200	224	Apr. 76	102	41			
Pre-treatment						0	0-0.05	0-0.02
Post-treatment						0-0.11	0-0.008	0-0.75
Big Quilcene River	200	224	Apr. 76	800	324			
Pre-treatment						0-0.03	0-0.06	0-0.01
Post-treatment						0-0.05	0-0.09	0-0.04

Table XI.B.10.—Stream water quality and quantity of flow following fertilization of a forested watershed, 1971: Fernow Experimental Forest, W.Va. (Aubertin and others 1973)

Treatment: Hardwood sprouts and seedlings were fertilized by helicopter with urea. No attempt was made to avoid a small perennial stream.

Site:	**Rate of application**		**Date of appli-cation**	**Treatment area**		**Concentration**			
						NH_4-N		**NO_3-N**	
	lb-N/ac	**kg-N/ha**				**max**	**ave**	**max**	**ave**
				ac	ha	-------------mg/l-------------			
Treated	230	258	May 71	74	30				
1970-1971						0.8	0.23	19.8	0.76
1971-1972							0.19		0.10
Control	---	---	---	---	---				
1970-1971							0.19		0.10
1971-1972							0.20		0.21

Table XI.8.11.—Stream water quality following fertilization of a gaged experimental watershed, spring 1970: South Umpqua Experimental Forest, Oreg. (Moore 1971)

Treatment: Watershed 2 was fertilized in March 1970 by helicopter. Urea, prill formulation, was applied and there was no attempt made to leave an untreated buffer zone along the stream. Watershed 4 was untreated and served as a control.

Site:	**Rate of application**		**Date of appli-cation**	**Treatment area**		**Concentrations**		
	lb-N/ac	**kg-N/ha**				**Urea-N**	**NH_3-N**	**NO_3-N**
				ac	ha	-------------mg/l-------------		
Watershed 2	200	224	Mar. 70	169	68	1.39	0.048	0.177
Watershed 4	---	---	---	120	49	0.006	0.005	0.002

Table XI.B.12.—The impact of forest fertilization on stream water quality in the Douglas-fir region—a summary of monitoring studies in Alaska, Idaho, Oregon, and Washington (Moore 1975a, 1977)

Treatment: Aerial application of urea.

Site:	Rate of application lb-N/ac	Rate of application kg-N/ha	Date of appli-cation	Treatment area		Peak Concentration Urea-N Pre-treatment	Urea-N Post-treatment	NH_3-N Pre-treatment	NH_3-N Post-treatment	NO_3-N Pre-treatment	NO_3-N Post treatment
				ac	ha	mg/l					
Burns Creek[1]	50	56	Nov 1970	1390	562	0	0	0	0	0	0.068
Canyon Creek	200	224	Nov 1969	3325	1346	0.005	15.20	nd	nd	0.005	0.80
Coyote Creek	200	224	Mar 1970	170	68	0.006	1.39	0.005	0.048	0.002	0.177
Crabtree Creek	200	224	May 1969	570	230	---	24.00	0	0.080	0	0.25
Dollar Creek	200	224	Apr 1971	85	34	0.016	44.40	0.030	0.490	0.060	0.13
Elochoman Creek	200	224	Nov 1969	735	297	0.073	19.10	nd	nd	nd	4.00
Fairchilds Creek	200	224	Apr 1972	475	192	0.008	23.40	0.009	0.280	0.030	0.828
Falls Creek	190	213	May 1970	650	263	nd	nd	0.020	1.28	0.015	1.67
Jackson Creek	150	168	May 1969	235	95	0.007	0.09	0.004	0.044	0.065	0.116
Jimmycomelately Creek	200	224	Apr 1970	120	49	0.002	0.71	0	0.040	0.005	0.042
McCree Creek	50	56	Oct 1970	1265	513	0	0.62	0	0	0	0.210
Mica Creek	200	224	Sep 1972	115	47	0	0.30	0	0	0.15	0.28
Mill Creek	200	224	Dec 1969	565	228	0.02	0.68	0	0.12	0.02	1.32
Nelson Creek	200	224	Apr 1970	95	38	0.016	8.60	0.010	0.32	0.290	2.10
Newaukum Creek	150	168	Sep 1971	6085	2463	0.009	0.26	0	0.008	0.011	0.438
Pat Creek	200	224	Apr 1972	600	243	0.003	3.26	0.007	0.079	0.061	0.388
Quartz Creek	200	224	May 1972	125	51	0.004	1.75	0	trace	0.120	0.70
Roaring Creek	200	224	Mar 1972	660	267	0.007	0.76	0.004	0.040	0.017	0.210
Row Creek	150	168	Oct 1972	6500	2630	0.006	0.13	0.005	0.022	0.004	0.044
Skookumchuck Creek	150	168	Sep 1969	470	191	0	2.63	0.004	0.026	0.005	0.085
Spenser Creek	200	224	Nov 1972	7680	3108	0.019	0.37	0.041	0.123	0.005	0.005
Tahuya Creek	200	224	Oct 1972	4005	1620	0.01	27.20	0	1.40	0.01	1.83
Thrash Creek[2]	200	224	May 1974	300	121	---	---	nd	0.06	nd	1.88
Three Lakes Creek	190	213	May 1970	170	69	nd	nd	0.015	0.13	0.003	2.36
Trapper Creek	200	224	Apr 1970	160	64	0.008	0.70	0	0.010	0.034	0.121
Trout Creek	200	224	Mar 1968	1600	648	0.10	14.00	0.12	0.700	0.03	0.160
Turner Creek	200	224	Mar 1972	870	352	0.004	4.36	0	0.046	0.032	0.243
Waddel Creek	200	224	Dec 1969	1480	600	0.01	2.48	0	0.340	0.02	0.99
Wishbone Creek	200	224	May 1972	115	46	0	0.30	0	0	0.12	0.28

[1] $(NH_4)_2SO_4$ applied [2] NH_4NO_3 applied nd = no data available or not determined

APPENDIX XI.C:

REFERENCE SOURCES FOR PESTICIDE CHEMICALS

Common name:	2,4-D
Chemical name:	2,4-dichlorophenoxyacetic acid
Other names:	Stauffer, Esteron, Amine, Dacamine
Registered use:	Control method for herbaceous and woody plants on cropland, forest, and rangeland, in orchards, on fallow land, and in pastures.

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Common name:	Dichlorprop, 2,4-DP
Chemical name:	2-(2,4-dichlorophenoxy) propionic acid
Other Names:	Weedone 2,4-DP, Weedone 170, Envert 170
Registered Use:	Brush control on non-agricultural lands

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Weed Science Society of America. 1974. Herbicide handbook of the Weed Sci. Soc. Am., 3rd ed. 430 p. Champaign, Ill.

Common name:	2,4,5-T
Chemical name:	(2,4,5-Trichlorophenoxy) acetic acid
Other names:	Esteron 245—PGBE ester; Ded-weed—Isooctylester; Brush/killer Lo Vol 4T—Isooctylester; Dinoxol—Butoxyethanol ester.
Registered use:	2,4,5-T is registered for control of woody and herbaceous plants; especially for brush control, selective conifer release, and control of woody plants in rangeland and pastures.

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Common name:	Atrazine
Chemical name:	2-chloro-4-ethylamino-6-isopropylamino-s-triazine
Other names:	AAtrex 80 W
Registered use:	Selective control of broadleaf and grassy weeds in conifer reforestation where it serves to increase seedling survival appreciably; also used in forest and Christmas tree plantations of Douglas-fir, grand fir, noble fir, white fir, lodgepole pine, ponderosa pine, and Scotch pine.

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Common name:	Carbaryl
Chemical name:	1-Naphthyl N-methyl carbamate
Other names:	Sevin, Sevin 4-Oil
Registered use:	Suppression of various insect outbreaks including the gypsy moth, cankerworm, saddled prominent and tent caterpillar, and the spruce budworm (eastern and western).

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Common name:	Chlorpyrifos
Chemical name:	0,0-diethyl-0-(3,5,6-trichloro-2-pyridyl) phosphorothioate
Other names:	Dursban, DOWCO 179, LORSBAN
Registered use:	Insect control.

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Common name:	Dalapon
Chemical name:	2,2-dichloropropionic acid
Other names:	Dowpon, Dowpon C, Dowpon M
Registered use:	A moderately specific grass herbicide commonly used as a pre-plant treatment on conifer planting sites.

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Weed Science Society of America. 1974. Herbicide handbook of the Weed Science Society of America. 3rd ed. 430 p. Champaign, Ill.

Common name:	Dicamba
Chemical name:	3,6-dichloro-o-anisic acid; also 2-methoxy-3,6-dichlorobenzoic acid
Other names:	Banvel, Banvel Brush Killer, Banvel 5G Granules
Registered use:	Brush control on non-croplands, including forest lands.

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Common name: Diflubenzuron

Chemical name: N(((4-Chorophenyl) amino)carbonyl)-2,6-di fluorobenzamide

Other names: Dimilin, Difluron, TH-6040

Registered use: Control of the gypsy moth; also used in aquatic ecosystems.

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Common name: Ethylene Dibromide

Chemical name: 1-2 dibromoethane

Other names: EDP, Fumo-gas, E-D-Bee, Bromo-fume, Soil-Fume, Dow-fume, Urifume

Registered use: Forest insecticide against Douglas-fir beetle, Jeffrey pine beetle, mountain pine beetle, roundheaded pin beetle, spruce beetle, California flatheaded bores, Monterey pine ips, fir engraver beetle, and western pine beetle.

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White, V.L. 1972. Letter to Stanley I. Undi, U.S. Dep. Agric., For. Serv. from Great Lakes Chem. Corp. [Feb. 21, 1972.]

Common name: Fenitrothion

Chemical name: 0,0-dimethyl-0-(3 methyl-4-nitrophenyl) phosphorthioate; also 0,0-dimethyl 0-(4-nitro-m-tolyl) phosphorothioate (1)

Other names: Sumithion, Sumitomo

Registered use: Control of hepidoptera, diptera, orthoptera, hemiptera, and coleoptera in field crops and on fruits and vegetables; forest protection through control of Japanese pine sawyer, pine caterpillar, hemlocklooper, spruce budworm, bark beetle, and weevil; control of insects affecting public health such as mosquitos, flies, bedbugs, and cockroaches; and control of locust and grasshopper.

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Common name:	Malathion
Chemical name:	(0,0-dimethyl dithiophospate of diethylmercaptosuccinate)
Registered use:	Control of a number of forest insects including defoliators and sucking insects of conifers and hardwoods.

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Common name:	MSMA
Chemical name:	Monosodium methane arsonate or Monosodium acid methan arsonate
Other names:	Silvisar 550 Tree Killer, Vichem 120 Arsonate Silvicide, Glowon Tree Killer
Registered use:	For post-emergent weed control and as a silvicide for control of undersirable conifers and big leaf maple.

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Common name:	Orthene (acephate)
Chemical name:	(O,S,Dimethyl acetylphosphoramidothioate)
Registered use:	Control of gypsy moth.

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Common name:	Picloram
Chemical name:	4-amino-3,5,6-trichloro-picolinic acid
Other names:	Tordon, ATCP
Registered use:	Control of annual and deep rooted perennial weeds in non-cropland.

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Common name: Silvex-fenoprop
Chemical name: 2-(2,4,5-trichlorophenoxy) propionic acid
Other names: Kuron, Weedone
Registered use: Control of woody plants, trees, and shrubs; specific brush control in forest site preparation and release; aquatic herbicide.

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Common name: Simazine
Chemical name: (2-chloro-4,6 bis(ethylcunino)-s-triazine)
Other names: Princep 80W
Registered use: Weed control in Christmas tree plantations.

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Common name:	Trichlorfon
Chemical name:	Dimethyl-(2,2,2-trichloro-1-hydroxy-ethyl) phosphorate
Other names:	Dylox
Registered use:	Control of the gypsy moth larvae on forest land shade trees.

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