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Chapter 15

Effects of Forest Fertilization on Water Quality and Aquatic Resources in the Douglas-fir Region

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ABSTRACT. Increased concentrations of three species of dissolved nitrogen are found in surface waters after urea application: (1) urea-N (often reported as Kjeldahl-N or organic-N), which is present for only a few days; (2) ammonia-N, which is often elevated for several weeks to several months; and (3) nitrate-N, which may be elevated for up to a year or more. Peak urea-N levels immediately after fertilization usually range from 0.1 to 50 mg/L depending on the percentage of watershed fertilized, drainage density, urea application rate, precipitation, and length and width of buffer strips (unfertilized areas) along riparian corridors. Peak concentrations of total ammonia-N (ionized and un-ionized ammonia) after fertilization are typically 0.1 to 0.5 mg/L depending on the factors mentioned above, as well as temperature and soil chemistry. Relatively high concentrations of ammonia in surface waters have been associated with fertilizing at low temperatures, which inhibits nitrification. The highest nitrate-N concentrations reported from Pacific Northwest streams after fertilization have exceeded 3 mg/L, but peak concentrations ranging from 0.1 to 1.0 mg/L are more typical of the Douglas-fir region. Estimated percentages of fertilizer nitrogen exported from watersheds in streams during the first year range from less than 1% to more than 10% of the total nitrogen applied. Baseline nitrate levels may be increased in watersheds with histories of multiple fertilization. During normal operations, neither drinking water standards (10.0 mg/L nitrate-N, 0.5 mg/L ammonia-N) nor aquatic toxicity thresholds (about 1.2 mg/L total ammonia-N) are exceeded. Increased nitrogen in streams has the potential to promote the growth of periphyton. In some streams increased primary production can lead to enhanced production of aquatic invertebrates and fishes, although enhanced fish production after forest fertilization has not yet been clearly demonstrated in the Pacific Northwest. Transport of fertilizer-derived nitrogen downstream to hydraulic sinks in the drainage system (lakes, wetlands) may contribute to accelerated eutrophication if these water bodies are nitrogen limited. Considered as a whole, however, the data suggest that forest fertilization in the Douglas-fir region does not produce conditions that exceed water quality standards. Although increases in dissolved nitrogen relative to baseline levels may be considerable, adverse effects on beneficial uses have not been shown. Therefore, while forest fertilization may cause changes in the nitrogen dynamics of receiving waters, it does not result in water quality impairment, provided reasonable precautions are taken to minimize direct entry of urea to streams from aerial applications and to prevent surface runoff from urea storage and loading areas.

Addition of urea to promote the growth of various conifer species has been practiced on commercial forest land in the Pacific Northwest since the mid-1960s (Crown 1974; Fredriksen et al. 1975). Urea prill are applied by aerial spreading from a helicopter about 30 to 90 meters above tree tops, quite often over steep mountainous terrain. Watersheds in which fertilization takes place in the Douglas-fir region have many small streams, some of which are fish bearing, but most are not perennial and are too small to support fish populations. In any case, these small watersheds almost always drain into larger stream systems that contain valuable fishery resources or serve as domestic water supplies. Thus, environmental concerns related to urea fertilization and forested streams have centered on drinking water quality and the effects on fish populations of increased nitrogen concentrations (Norris et al. 1983).

Monitoring studies in connection with fertilizer applications began in the late 1960s and continued into the 1970s (Moore 1975*a*). Initial objectives were to determine whether drinking water standards would be exceeded or whether the increased nitrogen concentrations would be toxic to aquatic life. When it became apparent

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that routine urea applications to coniferous forests did not result in nitrate concentrations in excess of the recommended drinking water limit of 10 mg/L (Tiedemann 1973; Moore 1972, 1975*a*; Fredriksen et al. 1975) or higher than suggested limits for the protection of fish and other aquatic resources (Thut and Haydu 1971; Meehan et al. 1975; Stay et al. 1978), the number of monitoring projects declined. Nevertheless, some concerns still lingered over possible sublethal effects of elevated nitrogen concentrations on aquatic biota, especially that increased eutrophication of downstream waters would result (Thut and Haydu 1971; Groman 1972).

In this chapter we review studies on the effects of forest fertilization on surface waters in the Douglas-fir region. These studies have included a broader geographical range than early research in western Oregon and Washington, and have also included examinations of watersheds with histories of multiple urea applications. We discuss factors influencing baseline nitrogen concentrations in streams, case studies of intensively monitored small watersheds, the potential of fertilizer nitrogen to exceed drinking water standards and aquatic toxicity thresholds, the effects of fertilizer nitrogen in stream and lake ecosystems (including the potential to increase fish production), and the effectiveness of unfertilized buffer strips in ameliorating nitrogen runoff. Physical and biological processes controlling the storage and transport of nitrogen in streams are currently being investigated using a variety of new research tools (e.g., Munn and Meyer 1990), and there is still much to be learned about the response of stream ecosystems to nitrogen additions. However, we attempt to summarize what is known about short- and long-term patterns in stream-water nitrogen after urea fertilization in the Pacific Northwest, and to relate these changes to water quality and biological considerations of regional interest.

Early Water Quality Studies

Initial monitoring studies of stream-water quality during and after operational fertilization revealed that three species of dissolved nitrogen were present after fertilizer applications (Thut and Haydu 1971; Norris and Moore 1971; Groman 1972). Urea-N concentrations, sometimes analyzed as dissolved Kjeldahl-N (analytical test for dissolved organic nitrogen), rose immediately and were followed shortly by elevated ammonia-N concentrations (Moore 1975*a*). Nitrate-N concentrations began to rise at a more gradual rate as oxidation of ammonia occurred, but elevated nitrate persisted in streams long after urea-N and ammonia-N had returned to background levels (Burroughs and Froehlich 1972). It was not uncommon to observe a second peak in nitrate the following autumn during the first heavy rainfall (Malueg et al. 1972), although this was not always the case (Moore 1975b). Peak concentrations of urea-N, ammonia-N, and nitrate-N from selected forest fertilization monitoring studies in the Pacific Northwest prior to 1975 are given in Table 1. Although the magnitude of the peaks varied greatly among watersheds, these early studies demonstrated that the temporal sequence of change in stream-water nitrogen concentrations after forest fertilization was similar throughout the region (Fredriksen et al. 1975).

Table 1—Average stream-water peak concentrations of urea-N (most measurements using the Kjeldahl test), total ammonia-N, and nitrate-N after forest fertilization in Alaska, Idaho, Oregon, and Washington. From Norris et al. (1983) based on Moore (1975*a*).

	Peak Concentration (mg/L)	
Nitrogen Species	Mean	Range
Urea-N	7.87	0.09 - 44.4
Ammonia-N	0.27	0.01 - 1.4
Nitrate-N	0.78	0.04 - 4.0

Water Quality Standards

Water quality standards related to urea fertilization can be divided into different categories: (1) public health standards for drinking water, (2) thresholds of toxicity to aquatic life, and (3) narrative standards that pertain to antidegradation. Standards within the United States are generally taken to be those set by the U.S. Environmental Protection Agency (EPA 1986). The latest water quality standards in Canada are now contained in CCREM (Canadian Council of Resource and Environment Ministers), first published in March 1987 and last updated in May 1990 (CCREM 1990). Narrative criteria related to general water quality protection in the United States are listed in the federal Clean Water Act. Recommended limits for different species of dissolved nitrogen are similar for the two countries and are shown in Table 2.

Urea-N ($CO(NH_2)_2$) is relatively nontoxic, and extremely high concentrations would be required in order to be harmful to either humans or fish (Norris et al. 1983). These concentrations would occur only in the event of a direct spill of large quantities of urea into a stream, and even then rapid hydrolysis of urea to ammonia would pose a far greater environmental risk.

Table 2—Recommended concentration limits for different species of dissolved nitrogen in drinking water and in natural waters for the protection of cold-water biota. Based on water quality standards in the United States (EPA 1986) and Canada (McNeely et al. 1979).

	Recommended Limits (mg/L)	
Nitrogen Species	Drinking Water	Aquatic Toxicity
Urea-N	None	3,000 - 10,000 (acute)
Ammonia-N (total)	0.5 (chronic)	1.2 (acute)
Nitrite-N	1.0 (chronic)	0.24 (acute)
Nitrate-N	10.0 (chronic)	None

Ammonia is generally regarded as the most toxic species of dissolved inorganic nitrogen (Thurston 1980), but there is no single water quality standard for protection of aquatic life, because ammonia toxicity has been shown to be highly variable in laboratory bioassays (Thurston and Russo 1983; Meade 1985). McNeely et al. (1979) recommend a total ammonia-N limit of 0.5 mg/L for public drinking water. There are two forms of dissolved ammonia, ionized (NH_4^+) and un-ionized (NH_3), and the un-ionized form is the most toxic (Szumski et al. 1982). However, under low temperatures and circumneutral pH levels that are prevalent in Pacific Northwest streams, the un-ionized form is not favored and generally occurs in concentrations less than 0.01% of those of the ionized form (Emerson et al. 1975). In some cases, ionized ammonia can become a water quality problem, even though it is less toxic, simply because of its greater abundance (Meade 1985).

In natural waters, ammonia toxicity will be influenced by many factors (Figure 1), making it very difficult to establish a threshold against which to judge fertilizer monitoring results. Since some basis for comparison is needed, however, a conservative estimate of the maximum safe level of total ammonia-N (ionized plus un-ionized) was calculated from the EPA (1986) recommended acute concentration limits for protection of cold-water biota. The EPA water quality tables show that ammonia toxicity increases with decreasing pH and increasing temperature. Assuming a stream-water pH of 6.5 and a maximum temperature range of 20-25° C (the extreme conditions normally encountered in streams in the Douglas-fir region when urea is applied), the EPA recommends a maximum ammonia concentration of 1.5 mg/L. Conversion of this value to ammonia-N (the form usually reported in analytical tests) by multiplying 1.5 by 0.8235 yielded an estimated maximum safe ammonia-N concentration for Pacific Northwest streams of 1.2 mg/L (Table 2). This estimate included both ionized and un-ionized forms.

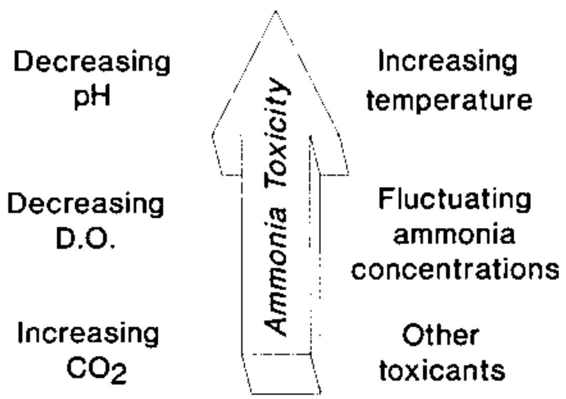

Figure 1. Some environmental factors contributing to increased toxicity of ammonia to fishes. D.O. = dissolved oxygen. Based on several papers by R.V. Thurston and colleagues.

Nitrite (NO_2^-) is an intermediate in the bio-oxidation of ammonia to nitrate. The acute toxicity of nitrite to fish is highly variable and has been reported to range from 0.24 to 11 mg/L for rainbow trout (Lewis and Morris 1986); therefore, 0.24 mg/L has been taken to be a conservative estimate of the concentration toxic to salmonids (Table 2). Nitrite normally occurs at extremely low levels in well-oxygenated water (less than 0.005 mg/L) and has not been implicated as a water quality problem associated with forest fertilization (Norris et al. 1983). Toxic concentrations in streams have been caused by industrial and sewage effluents, and have been associated with certain types of aquaculture (Lewis and Morris 1986). The EPA Gold Book indicates that "nitrite nitrogen levels kept at or below 0.06 mg/l should be protective of salmonid fishes. These levels are not known to occur or would be unlikely to occur in natural surface waters." The CCREM (1990) report suggests a drinking water nitrite-N limit of 1.0 mg/L.

There is no generally accepted aquatic toxicity threshold for nitrate-N (NO_3^-), but the eggs of some salmonids have demonstrated sensitivity to concentrations of approximately 10.0 mg/L (Kincheloe et al. 1979). Like humans, fishes suffer impaired respiratory ability from high nitrate concentrations. The drinking water standard of 10.0 mg/L appears to be sufficient to protect aquatic organisms (Norris et al. 1983).

Natural Processes Influencing Nitrogen Concentrations

Most streams in the Douglas-fir region have low dissolved nitrogen concentrations, sometimes as low as a few µg/L for each species (Table 3). However, there is considerable natural variation among watersheds and a number of factors can influence baseline levels. Nitrogen in streams is strongly influenced by precipitation

Table 3—Range of natural concentrations of dissolved organic nitrogen, ammonia, and nitrate in streams in the coastal Douglas-fir region, from baseline measurements in Oregon, Washington, and British Columbia.

Nitrogen Species	Range of Concentration (mg/L)
Dissolved organic-N	<0.005 - 0.4
Ammonia-N	<0.002 - 0.03
Nitrate-N	<0.005 - 0.7

(Feller 1977). Increases in the absolute amounts, but not necessarily concentrations, of dissolved organic-N, ammonia-N, and nitrate-N typically coincide with winter storms, with the greatest increases usually associated with the first fall freshets (Scrivener 1982). Dissolved organic and inorganic nitrogen also increases in fall and winter in rainfall-dominated climates, due to the release of nitrogen stored in forest soils (Feller 1977; Vitousek et al. 1982), leaf decomposition (Triska and Buckley 1978), entrainment of organic matter from the floodplain (Naiman and Sedell 1979), and, possibly, decomposing salmon carcasses (Richey et al. 1975; Kline et al. 1990).

Other watershed characteristics can influence baseline nitrogen levels. Beaver ponds and riparian wetlands provide water and organic matter storage sites where anaerobic conditions in sediments facilitate denitrification and nutrient release (Dahm et al. 1987). These areas can deliver nitrogen-rich water to streams and are heavily used by wildlife.

Composition of riparian and upland vegetation also influences baseline nitrogen levels. Nitrogen-fixing species such as red alder (*Alnus rubra*) and low growing plants such as *Ceanothus* and *Lupinus* species add substantial amounts of nitrogen to forest soils where these species are abundant (Tarrant and Miller 1963; Berg and Doerksen 1975; Van Miegroet et al. 1990). In addition, the rate of nitrification in soils is influenced by the relative abundance of available nitrogen and the C:N ratio (Van Miegroet et al. 1990). Timber harvest in portions of a drainage basin can accelerate nitrogen runoff (Vitousek and Melillo 1979), but the extent of nitrogen losses after clearcutting in the Pacific Northwest can vary greatly (Feller 1977).

Under certain circumstances, riparian forests and wet sites can serve as nutrient filters. Where bacterial denitrification occurs, nitrate inputs to streams can be reduced. Schipper et al. (1991) studied nitrate dynamics in saturated soil riparian areas and found reductions in nitrate concentrations as great as 80% over a distance of 2 meters. Concentrations of nitrate in groundwater were reduced by as much as 98% as the water passed through the riparian zone. Lowrance et al. (1984) found that riparian forests could act as nutrient filters in agricultural watersheds. These authors reported nitrogen reductions in groundwater due to plant uptake of about 25%.

Storage and recycling of nutrients within the stream itself also affect the concentration of dissolved nitrogen (Triska et al. 1982). Munn and Meyer (1990) found that the frequency of organic debris dams strongly influenced the distance required for nitrogen to be recycled by stream biota; the more frequent the dams, the tighter the nutrient "spirals." Uptake by periphyton can remove significant amounts of dissolved nitrogen from stream water before it is transported downstream, and convert it to particulate form (Newbold et al. 1982). Nitrification and denitrification can take place in stream sediments (Cooke and White 1987; Wissmar et al. 1987) and in mats of periphyton (Duff et al. 1984), and these processes can add or remove available nitrogen from the water column. In addition, stream-water nitrogen can be entrained in or released from deep hyporheic sediments adjacent to the stream channel (Triska et al. 1989; Duff and Triska 1990).

The processes involved in nitrogen storage and transport in forested watersheds are interconnected and extremely complex (Triska et al. 1984). Regional geoclimatic differences, vegetative conditions, soil structure, groundwater transport characteristics, and stream channel morphology all influence baseline nitrogen levels. Because extrapolation from one watershed to another can lead to significant error, extensive prefertilization monitoring is often necessary to establish baseline concentrations prior to fertilization.

Nitrogen Increases Following Fertilization

Short-term and Long-term Patterns

Each operational fertilizer application produces a unique change in the pattern and magnitude of dissolved nitrogen runoff in streams. Nevertheless, the general sequence of nitrogen export is similar throughout the Douglas-fir region, although the timing and amount of increase are variable. This variability is illustrated in a comparison of two urea applications in western Washington (Figure 2). Louse Creek is a second-order stream that drains a small midelevation watershed on the west slope of the Cascade Range. The watershed is steep and soils are well drained and of volcanic origin. Ludwig Creek is a small third-order stream that drains a slightly larger, low elevation watershed in the Coast Range. The gradient is more gentle, the watershed is highly dis-

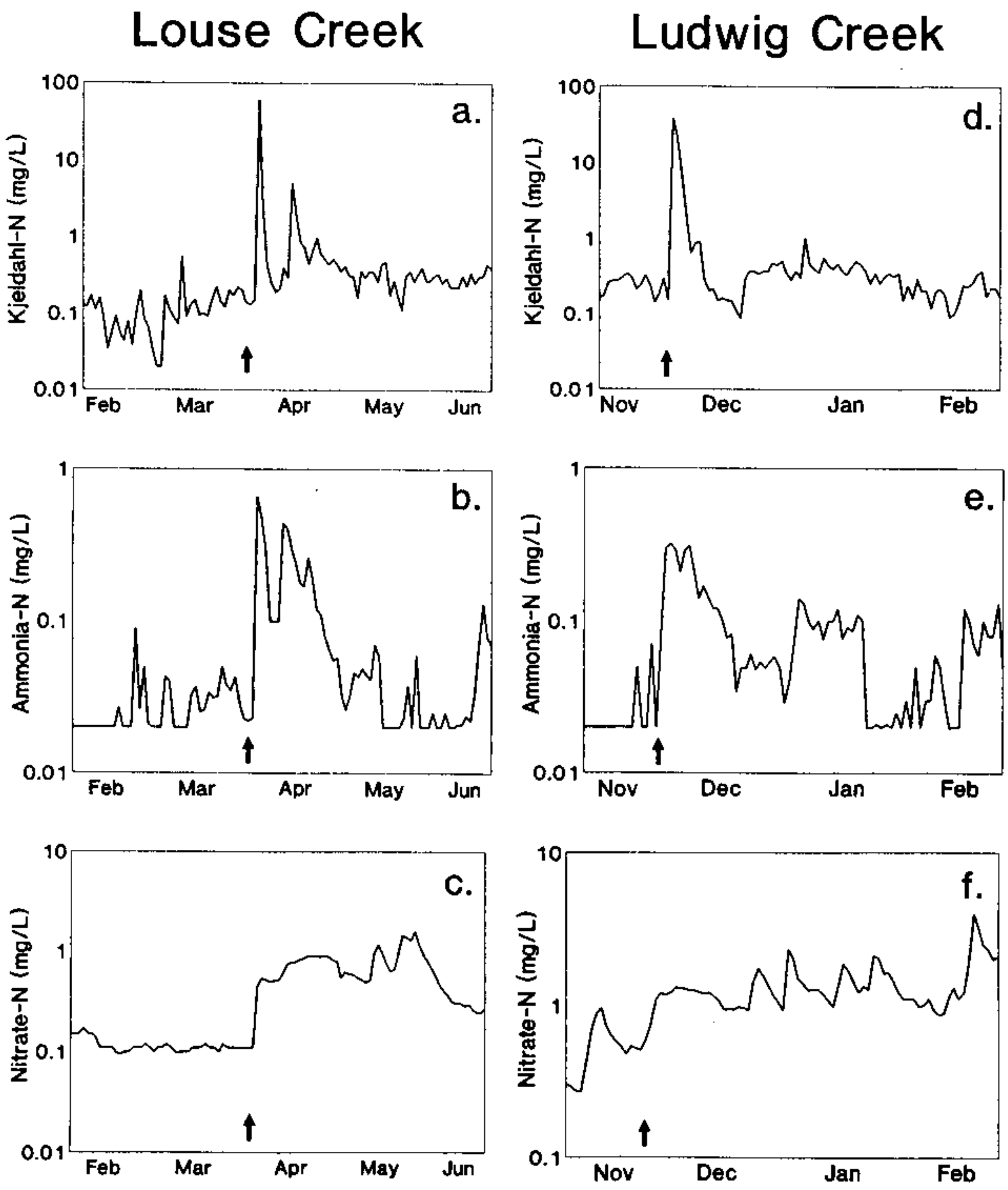

Figure 2. Changes in dissolved nitrogen species after urea application (224 kg N/ha) in Louse Creek, a midelevation stream in the Cascade Range, and Ludwig Creek, a low elevation stream in the Coast Range. Arrows indicate time of fertilization. From P. Bisson and B. Fransen, unpublished.

sected by approximately 25 first-order tributaries, and the deep soils are of both volcanic and sedimentary origin. In addition, the Ludwig Creek watershed contains several small wetlands and beaver ponds.

Virtually the entire areas of both drainages were fertilized at an application rate of 224 kg N/ha, and both watersheds had been fertilized at least once previously. The Louse Creek watershed was fertilized in early spring

(April 13, 1989) while the Ludwig Creek watershed was fertilized in late fall (December 1, 1988). No unfertilized buffer strips were left along either stream during applications by helicopter, and urea prill were observed in both streams for up to an hour after fertilization. Water samples were drawn from each stream with automatic pump samplers.

In Louse Creek, organic nitrogen, ammonia, and nitrate responded immediately to fertilization, but Kjeldahl-N returned to a level very near prefertilization concentrations within a few days (Figure 2a-c). Ammonia-N remained elevated for slightly more than one month, and nitrate-N remained elevated throughout the postfertilization monitoring period of approximately 90 days. The latter observation was not surprising since increased nitrate-N concentrations for up to a year or more have been found in some long-term fertilization monitoring studies (Stay et al. 1979; Hetherington 1985).

Nitrogen increases in Ludwig Creek were more protracted (Figure 2d-f). Organic nitrogen was elevated for more than a week before returning to the prefertilization concentration range. Ammonia underwent a gradual decline but then increased sharply with heavy rainstorms in late December and early January, and rose again in mid-February. Nitrate concentrations rose slightly but steadily throughout the monitoring period, with brief peaks corresponding to freshets.

Hourly samples were taken in both streams for four days during and immediately after fertilization (Figure 3a-d), and changes in nitrogen concentrations illustrated response differences of the two watersheds. Kjeldahl-N in Louse Creek rose quickly to 60 mg/L, the highest level yet recorded for streams in the Pacific Northwest after a urea application, and then returned quickly to baseline levels. The pattern of Kjeldahl-N increases in Ludwig Creek was different. Three separate peaks were

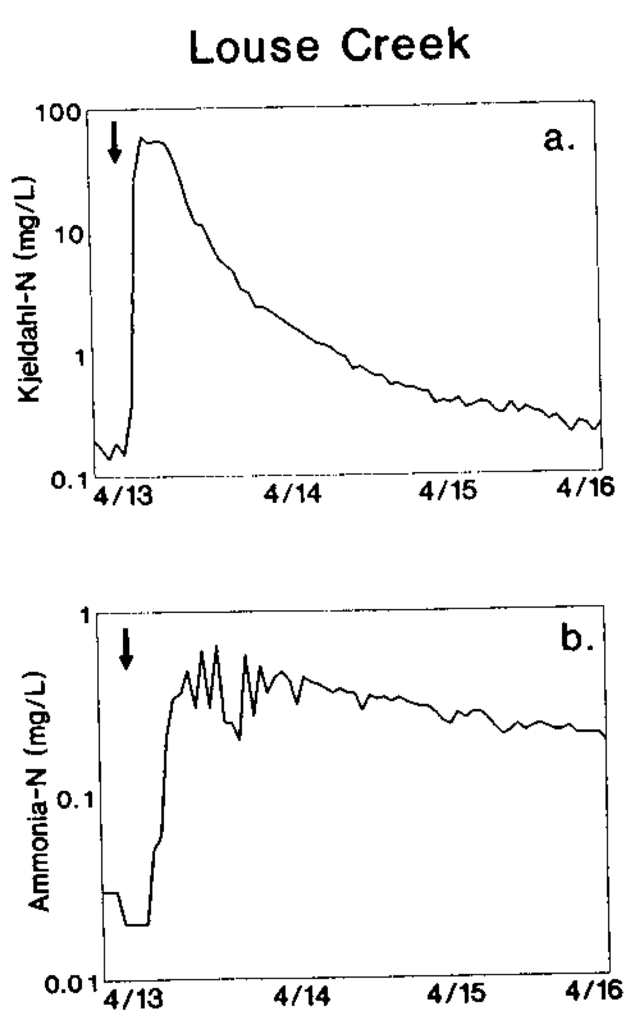

Figure 3. Hourly changes in Kjeldahl-N and ammonia-N during the first three days after urea application in the Louse Creek and Ludwig Creek watersheds. Arrows indicate time of fertilization. From P. Bisson and B. Fransen, unpublished.

noted during the three days following fertilization, although urea application was completed in only a few hours of the first day. No precipitation occurred during this period and the peaks on the second and third day after fertilization could not be explained, but may have been related to the wetlands, beaver ponds, and groundwater seeps that were numerous in the watershed. Likewise, short-term changes in ammonia-N in the two streams differed after fertilization. In Louse Creek, ammonia rose rapidly and declined steadily, while in Ludwig Creek the increase took place in stages, with the highest concentrations measured on the second and third days following urea application. Thus both short- and long-term differences in nitrogen runoff occurred between these two watersheds; however, the general patterns were similar and were consistent with other monitoring studies in the Douglas-fir region (Norris et al. 1983).

Limited water quality monitoring is often a requirement of operational forest fertilization. Typically, a few grab samples are taken before and after urea application to determine whether water quality standards are met. Monitoring protocols also may require grab sampling during freshets in an effort to establish peak concentrations. However, results of intensively monitored sites with sampling intervals as often as hourly suggest that concentration peaks are very transitory (urea) and unpredictable (ammonia and nitrate). Thus grab sampling, especially with a limited frequency, stands very little chance of measuring peak postfertilization concentrations of any of the nitrogen species.

Overall, most monitoring studies have shown that forest fertilization causes elevated urea (organic nitrogen) concentrations over a period of days, elevated ammonia concentrations over a period of a few weeks to a few months, and elevated nitrate concentrations over a period of up to one year or more. The general pattern of nitrogen runoff observed during studies in the 1960s and 1970s has been upheld by research in the 1980s, although the duration of nitrate concentration increases has been shown in some cases to be longer than was measured in the early studies.

Annual Losses of Fertilizer Nitrogen

Many water quality monitoring projects have attempted to determine the fraction of applied nitrogen lost to stream-water runoff. Early research on the transport characteristics of nitrogen amendments through the soil (Cole and Gessel 1965) had suggested that fertilizer nitrogen was not easily removed, and therefore that little if any added nitrogen would be lost to streams. Water quality studies during the 1960s and 1970s seemed to support this conclusion (Table 4), with annualized estimates of nitrogen loss usually amounting to less than 1% of the total amount applied to the watershed (Moore 1972, 1975*b*).

More recent studies, however, have produced loss estimates considerably greater than those of the 1970s (Table 4). Preston et al. (1990) were able to account for only about 50% of the labeled ^{15}N added to a forest site in coastal British Columbia; the rest was believed to have been lost through leaching, denitrification, or volatilization. Nitrogen export in stream water within one year of fertilizer application has been estimated to be greater than 10% in one small watershed in British Columbia (Hetherington 1985), and two other studies in the Pacific Northwest have placed annual nitrogen losses at between about 2% and 10%. Fertilization with ammonium nitrate led to estimated losses to stream water of 27.5% over three years in the Fernow Experimental Forest of the central Appalachian Mountains in West Virginia (Edwards et al. 1991). Likewise, annual nitro-

Table 4—Estimated percentages of fertilizer nitrogen lost to stream runoff in one year, from studies of forest fertilization in the Pacific Northwest and Western Europe.

Location	Percentage of Nitrogen Lost In One Year	Reference
Pacific Northwest		
Western Oregon	0.17	Moore (1972)
Western Washington	0.45	Cline (1973)[1]
Oregon and Washington	2 - 3	Moore (1975*a*)[2]
Western Washington	0.20 - 0.26	Moore (1975*b*)
Oregon and Washington	<0.5	Fredriksen et al. (1975)
Western Washington	1.9 - 9.0	Bisson and Marosy (1981, unpub.)[3]
Vancouver Island	2.1 - 5.2	Perrin et al. (1984)[4]
Vancouver Island	5.9 - 14.5	Hetherington (1985)[5]
Western Europe		
Scotland	9	Harriman (1978)[6]
Norway	10	Ogner (1982)[7]

[1]Cited by Hetherington (1985).

[2]Estimated losses after early fertilizer applications that gave little protection to watercourses.

[3]Based on studies of four small watersheds that had one to three previous urea applications.

[4]Believed by the authors to be an underestimate of the actual amount of exported nitrogen.

[5]Total losses measured over 14 months, rather than 12 months.

[6]Fertilization with ammonium nitrate; loss estimate based on 10-month period.

[7]Fertilization with ammonium nitrate; loss estimate based on 15-month period.

gen export from European watersheds fertilized with ammonium nitrate has been estimated at about 10% (Table 4).

Feller (1977) has suggested a mechanism that might explain the apparent discrepancy between the hypothesis that nearly all fertilizer nitrogen will be retained in the soil (Cole and Gessel 1965; Cole et al. 1975) and the results of recent monitoring studies that led to higher loss estimates than earlier predictions (P. Bisson and M. Marosy, 1981, unpublished; Perrin et al. 1984; Hetherington 1985). Rapid transport of solutes through soil macropores (decayed root channels) was believed by Feller (1977) to potentially result in much greater nitrogen losses to streams than were predicted by lysimeter measurements of water percolating through the soil matrix. Water movement through macropores is difficult to sample, but may be an important pathway by which fertilizer nitrogen enters streams without being retained in forest soils (Hetherington 1985).

Effects of Multiple Fertilizer Applications

Some watersheds in the Douglas-fir region have been fertilized several times, yet the effects of repeated applications on water quality are poorly known. In one unpublished study, baseline nitrogen concentrations in watersheds with different fertilization histories but which were otherwise similar with respect to major soil type, site index, slope, and stand age were compared (F. Guerrero and P. Bisson, Weyerhaeuser Company, Tacoma, Washington). Streams draining watersheds that had been previously fertilized had higher fall baseline nitrate-N concentrations than streams draining unfertilized watersheds, and there appeared to be a trend toward increasing baseline levels with successive fertilizer applications (Figure 4). A possible mechanism for the increase in stream-water nitrate in the fall, if in fact it represents a general pattern, is the flushing of nitrate from watersheds where multiple fertilizer applications have enhanced the rate of nitrification in forest soils (Wagenet et al. 1977). In such watersheds, long-term export of nitrogen may be increased as a result of multiple fertilization.

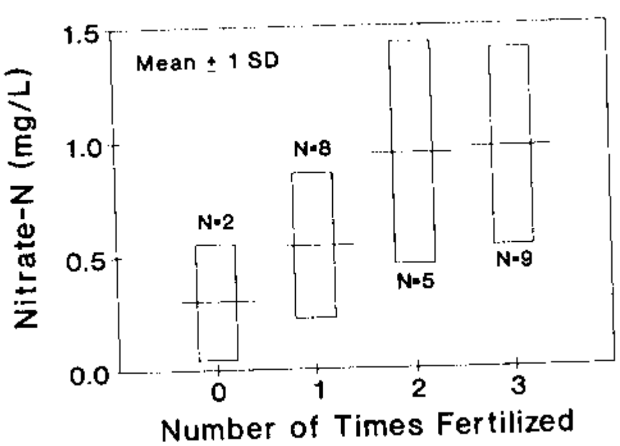

Figure 4. Baseline autumn nitrate-N concentrations in streams in the Silver Lake basin of southwestern Washington flowing from watersheds with different fertilization histories. Sample size refers to number of watersheds. From F. Guerrero and P. Bisson, unpublished.

Biological Effects of Forest Fertilization

Fertilizer Nitrogen as a Pollutant

Relative to the nitrate and ammonia drinking water standards and to the estimated ammonia toxicity threshold (Table 2), peak nitrogen concentrations in streams after urea applications have usually been well below recommended water quality limits (Table 5). In most cases, maximum ammonia-N levels have been less than 0.5 mg/L and nitrate-N levels have been less than 3 mg/L, although one study on Vancouver Island recorded a nitrate-N peak of 9.3 mg/L in a small tributary (Hetherington 1985). In part, these relatively low peak concentrations have been aided by naturally low baseline nitrogen concentrations in Pacific Northwest streams. The results of over 20 years of monitoring suggest that operational forest fertilization in the Douglas-fir region very rarely exceeds drinking water standards in streams.

In the Mohun Lake study on Vancouver Island (Perrin et al. 1984), total ammonia-N concentrations after fertilization briefly exceeded potentially toxic levels in several small watersheds where streams were not protected by unfertilized buffer strips. Peak ammonia-N concentrations in five of seven watersheds without buffer strips were greater than 1.2 mg/L, and four of the seven exceeded 3.0 mg/L. The high ammonia concentrations were attributed to direct introduction of urea to the streams, as peaks occurred within one week of application. However, long-term urea and ammonia elevation (up to 144 days) was also noted in this study. Perrin et al. (1984) speculated that late fall and winter fertilizer

Table 5—Typical peak concentrations of ammonia-N and nitrate-N in streams after urea fertilization in the Douglas-fir region, and their relationship to the estimated threshold for aquatic toxicity (ammonia) and drinking water toxicity (nitrate).

Nitrogen Species	Typical Peak Concentration (mg/L)	Percentage of Toxicity Threshold
Ammonia-N	0.1 - 0.5	10 - 40
Nitrate-N	0.5 - 3.0	5 - 30

applications, including some applications on snow cover, could result in extended periods of relatively high urea and ammonia levels in streams because temperature-dependent nitrification rates would be reduced (Wollum and Davey 1975; Otchere-Boateng and Ballard 1978). Meehan et al. (1975) also measured high ammonia concentrations in streams and delayed nitrification after urea fertilization in southeastern Alaska. Some water quality benefits may therefore be achieved by not applying urea at very low temperatures, particularly where snow is present. Preston et al. (1990) caution against fertilization on snow with nitrate as opposed to other forms, since the potential for losses due to leaching is great.

Another potential concern is that forest fertilization could promote nuisance growths of algae in streams and lakes (Groman 1972; Tamm et al. 1974). This could occur if nitrogen was the primary nutrient limiting algal production and if other factors such as scouring during freshets, consumption by invertebrates, low temperatures, and high suspended sediment were not controlling algal biomass. Whether or not nitrogen will act as the primary limiting nutrient will depend, in most cases, on the atomic ratio of available nitrogen to phosphorus as well as the actual concentrations of both. Critical N:P ratios as low as 7:1 or as high as 33:1 have been suggested as criteria for establishing nitrogen or phosphorus limitation to algae in natural waters; however, many workers in the Pacific Northwest favor a ratio of approximately 10-15:1 (Thut and Haydu 1971; Stockner and Shortreed 1978; Perrin et al. 1987). That is, N:P atomic ratios below this range indicate that aquatic plants will be nitrogen limited, while ratios greater than this range indicate that plants are likely to be phosphorus limited.

Although nitrogen has been identified as a limiting nutrient in some streams in the Douglas-fir region (Thut and Haydu 1971; Gregory 1980; Gregory et al. 1987; Munn and Meyer 1990), increased algal growth in streams after fertilizer applications has not been reported to reach nuisance or harmful levels. Most forested streams in watersheds receiving nitrogen amendments are relatively steep, have short water residence times, and are heavily shaded. Accumulation of periphyton in such streams is often limited by light levels (Gregory 1980), water currents (Horner and Welch 1981), the grazing activities of invertebrates (Lamberti and Resh 1983; Gregory 1983; Power 1990), and other factors unrelated to nutrient concentrations.

Nitrogen runoff into a highly oligotrophic lake can potentially increase phytoplankton production if the lake is nitrogen limited. Dense phytoplankton blooms could reduce water clarity and impair the pristine quality of these systems. However, the occurrence of nitrogen limitation in lakes is relatively rare and tends to be transitory (Schindler 1977). More often, phosphorus is the primary limiting nutrient in lakes in the Pacific Northwest (Perrin et al. 1984; Stockner and Shortreed 1985). As with streams, there have been no clearly documented cases of forest fertilizer runoff causing algae in lakes to increase to nuisance levels in the Douglas-fir region.

Fertilizer Nitrogen as an Aquatic Nutrient

Nutrient additions to streams have been shown in certain cases to increase the production of fishery resources (salmon and trout) by increasing the availability of food organisms (Huntsman 1948; Warren et al. 1964; Mundie et al. 1983). Similarly, fish production in lakes has been positively correlated to nutrients, phytoplankton, and zooplankton (Northcote and Larkin 1956; Ryder et al. 1974; Le Brasseur et al. 1978). Because nitrogen may be a limiting nutrient to primary production in some stream ecosystems in the Pacific Northwest (Gregory et al. 1987), introduction of nitrogen from forest fertilization has the potential of promoting the growth of algae, leading to greater abundance of aquatic invertebrates, which in turn increases the availability of food for fishes.

Several studies have examined the response of different trophic levels in streams to nutrient additions in the Pacific Northwest and Alaska (Table 6). Gregory (1980) found that nitrogen was the primary limiting nutrient in a small stream in the H. G. Andrews Experimental Forest of western Oregon; however, he noted that light limitation overrode nutrient limitation in this heavily canopied old-growth system, and only by experimentally increasing light levels would added nitrogen produce an increase in algal growth. Bisson et al. (1975) continuously added nitrate for two years to uncanopied experimental stream channels in the Cascade Range of Washington. They, too, found nitrogen to be the primary limiting nutrient, but were not able to show that increased primary production led to increased production of rainbow trout. Stockner and Shortreed (1978) tested both nitrogen and phosphorus additions to flowing-water troughs in Carnation Creek, a small west coast Vancouver Island stream. They found that phosphorus, not nitrogen, was the primary limiting nutrient. In one of the largest whole-river experimental enrichment study in the Pacific Northwest to date, a mixture of nitrogen and phosphorus was continuously added to the Keogh River, a medium-sized river on the east coast

Table 6—Responses of stream and lake biota to nutrient enrichment along the Pacific coast of North America.

Location	Nutrient(s)	Effect	Reference
		STREAMS	
Oregon			
Andrews Forest	N, P, light	Increased algal growth (N and light only)	Gregory (1980)
Washington			
Kalama River	N, trace elements	Increased algal growth (N only)	Bisson et al. (1975)
British Columbia			
Carnation Creek	N, P	Increased algal growth (P only)	Stockner and Shortreed (1978)
Keogh River	N, P	Increased algal growth;increased salmonid growth	Perrin et al. (1987) Johnston et al. (1990)
Thompson River	P	Increased algal growth	Bothwell (1985, 1989)
Alaska			
Kuparuk River	P	Increased algal growth;increased salmonid growth	Peterson et al. (1985 and unpub.)
		LAKES	
British Columbia			
Mohun Lake	N, P	Increased phytoplankton production (mostly P)	Perrin et al. (1984)
17 coastal lakes	N, P	Increased phytoplankton production (mostly P); increased salmonid growth	Stockner and Shortreed (1985) Hyatt and Stockner (1985)

of Vancouver Island. The nutrients were added in spring and summer specifically for the purpose of enhancing the production of anadromous salmonids. Results of this study showed that enrichment improved algal growth (Perrin et al. 1987), and that the size of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*O. mykiss*) significantly increased after fertilization (Johnston et al. 1990). Continuous phosphorus enrichment of the Kuparuk River, an Alaska tundra stream, has also demonstrated that nutrient additions can enhance primary production (Peterson et al. 1985) and increase fish growth, in this case the growth of grayling (*Thymallus arcticus*).

Experimental fertilization of Great Central Lake on Vancouver Island with a mixture of nitrogen and phosphorus demonstrated that the run of adult sockeye salmon (*O. nerka*) could be enhanced (Le Brasseur et al. 1978). Because of the success of this pilot project, a number of oligotrophic lakes (Table 6) in coastal British Columbia have been fertilized to improve the growth and survival of juvenile sockeye salmon (Shortreed et al. 1984). In general, the lakes have been limited more by available phosphorus than by available nitrogen (Stockner and Shortreed 1985). Only in the Mohun Lake study has forest fertilizer runoff been considered as a potential nutrient source for enhanced fish production (Perrin et al. 1984). These authors concluded that nitrogen derived from forest fertilization improved phytoplankton production in Mohun Lake in the spring, but that phosphorus was the primary limiting nutrient for most of the year.

Although elevated nitrogen levels in streams and lakes can potentially lead to increased fish production in nitrogen-limited waters, there have been no studies in the Douglas-fir region that have clearly demonstrated a positive association between urea applications and fish populations. In some cases, streams have responded to mixtures of both nitrogen and phosphorus, and in other cases nitrogen additions did not raise primary production without a corresponding increase in light levels. Similarly, lake enrichment has improved fish production, but only when both nitrogen and phosphorus have been added. At this point, it appears possible that fertilizer runoff could enhance the growth and survival of salmon and trout in certain waters, but definitive evidence is lacking.

Potential for Downstream Cumulative Effects

Little is known about the additive effects of multiple fertilizer applications within a river system on downstream aquatic resources, and this issue deserves more study. Where forestry is the major land use, fertilization has the potential to contribute significantly to total nitrogen loading in a river if a large area of the drainage basin is fertilized and baseline nitrogen concentrations are low. Unfortunately, we are aware of no studies that have addressed the cumulative effects of widespread fertilizer applications on the aquatic resources of a larger stream or river (greater than fourth-order). This lack of information has sometimes been a barrier to the acceptance of forest fertilization as an environmentally safe forest management practice.

In many river basins of the Douglas-fir region, forest management is one of several anthropogenic sources of nutrients. Significant inputs can come from agricultural practices and a variety of urban and industrial stormwater and wastewater discharges. Partly because of the tremendous expense involved in constructing a nitrogen budget for an entire river basin, and because the relative contributions of nitrogen from different types of land use are generally not known, it is often impossible to assess the downstream cumulative effects of forest fertilization within large river systems.

Although there is no sure way to identify surface waters that might respond in either a desirable or undesirable way to forest fertilization, resource managers should begin by evaluating the current condition of the drainage of interest. This evaluation should include the relative concentrations of baseline nitrogen and phosphorus, and the extent of eutrophication of rivers and lakes within the system. The occurrence of valuable fishery resources, hatchery water intakes, and drinking water supplies downstream from fertilizer applications should be documented so that an appropriate monitoring program can be established if necessary. The presence of potential sinks (lakes, sloughs, wetlands, estuaries) for nitrogen transported downstream should be noted, and inputs of nitrogen from other sources should be identified, even if actual amounts cannot be quantified. Aesthetics should also be considered. Some pristine, oligotrophic waters, particularly those with significant recreational use, may have value apart from fish production, and this value should be taken into account.

Environmental Protection

State and provincial forest chemical regulations usually prescribe minimizing direct application of fertilizer to streams. This is accomplished by leaving unfertilized strips along streambanks. In locations that include very steep terrain and large numbers of small first- and second-order channels, the ability of helicopter application methods to effectively buffer small streams is limited. Reasons for the inability to effectively leave unfertilized strips along some headwater streams include the steepness of the hill slopes, often requiring that the flight line be perpendicular rather than parallel to the axis of the stream, and the difficulty in locating the channel beneath a coniferous forest canopy. Larger streams are easier to buffer; they are located in more gentle terrain, the channel is more readily visible, and they are often bordered by deciduous trees.

Few studies have examined the effectiveness of unfertilized buffer strips in preventing nitrogen from entering streams. Perrin et al. (1984) found that 50 m buffer strips reduced peak urea and ammonia concentrations by about an order of magnitude, while nitrate peaks were reduced by more than 50% (Table 7). In addition, buffer strips delayed the time for peak urea and ammonia concentrations to occur. The authors attributed the success of the unfertilized buffer strips to reduced direct application of urea prill to the streams. They also found that high concentrations of both urea (57.6 mg/L) and ammonia (4.78 mg/L) in one stream apparently originated from a fertilizer spill at the heliport. These results illustrate the importance of locating fertilizer storage and loading areas away from watercourses or ditches that may carry spilled fertilizer to perennial stream channels. Because urea dissolves rapidly, spills should be cleaned as quickly as possible.

Accuracy of aerial urea prill application was investigated in an open field by Terry and Goedhard (1989). A row of open containers (pans) collected prill applied by helicopter from heights of about 30 and 90 m (100 and 300 ft). The effective swath width was defined as the total width of the swath in which prill were collected in the containers. The maximum swath width was the absolute maximum width of all observed fertilizer, including prill that were so infrequent as to remain unsampled in the containers. Trials were conducted when conditions were relatively calm and when conditions included a crosswind of approximately 25 km/hr (10-15 mph). At normal flight altitudes (30 m), Terry and Goedhard (1989) found that the effective swath width was about 60 m (200 ft). Increasing the flight altitude to 90 m expanded both the effective and maximum swath widths somewhat, and the presence of a crosswind displaced the swath laterally from the flight center line (Table 8). Based in part on these results, the Washington State Department of Natural Resources has recently recommended that buffer strips be left where a

Table 7—Effects of 50 m unfertilized buffer strips on peak nitrogen concentrations in the Mohun Lake forest fertilization study. From Perrin et al. (1984).

	Peak Concentration (mg/L)	
Nitrogen Species	Without 50 m Buffer Strip	With 50 m Buffer Strip
Urea-N	25.2	0.3
Ammonia-N	2.83	0.28
Nitrate-N	0.39	0.16

Table 8—Effective and maximum widths (see text) of the swath of urea prill applied by helicopter from two different altitudes. From Terry and Goedhard (1989).

Flight Altitude (m)	Effective Swath Width (m)	Maximum Swath Width (m)
30	61[1]	72
90	73	90

[1]25 km/hr crosswind displaced swath 6 to 9 m from flight centerline.

flight path parallel to the stream is possible, and that the centerline of the flight path be 150 ft (45 m) from the water's edge when winds are calm or blowing away from the stream. When winds are blowing toward the stream, the recommended flight centerline increases to 200 ft (60 m) from the water's edge.

Summary

Recent investigations have shown that the amount and duration of fertilizer nitrogen entering streams are highly variable, but may be greater than was suggested by early water quality monitoring studies. Repeated watershed fertilization, urea applications during cold weather, water movement through soil macropores, and the possibility of long-term increases in nitrification rates in forest soils may contribute to elevated nitrogen runoff in the Douglas-fir region. However, peak concentrations of urea-N, ammonia-N, and nitrate-N in streams are, in nearly all routine fertilizer applications, less than 50% of the recommended limits for drinking water and the protection of salmonid fishes. There is a possibility that primary production and, in turn, fish production will be increased in streams and lakes where nitrogen is the principal limiting nutrient, but such an effect has not yet been clearly demonstrated and will be strongly influenced by phosphorus availability. Unfertilized buffer strips are an effective means of reducing peak nitrogen concentrations, particularly of urea and ammonia, during and soon after fertilizer application.

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Questions and Answers

Have there been any studies of the effects of fertilization on macroinvertebrates in stream systems?

According to the EPA recommendations for protection of cold-water biota (EPA 1986), the sensitivity of fishes to elevated nitrogen, especially ammonia, is equal to or greater than the sensitivity of aquatic invertebrates. Acute (96 hr) toxicity thresholds of *un-ionized* ammonia-N for salmonids have been reported to range from 0.083 to 1.09 mg/L, while chronic (>4 days) toxicity thresholds for macroinvertebrates range from 0.304 to 1.2 mg/L. Therefore, protection of water quality for fishes should provide adequate protection for macroinvertebrates.

Several of the studies cited in Table 6 documented increases in the density of aquatic invertebrates when algae responded to nutrient additions. The types of invertebrates most likely to benefit from increased autotrophic production would be members of feeding guilds able to make direct use of periphyton and algal-based detritus (Gregory et al. 1987). In Pacific Northwest streams, some Ephemeroptera (mayflies) and many members of the Chironomidae (midges) are dominant within these feeding guilds. These insect larvae readily enter the drift and are heavily used by salmonids. It should be pointed out, however, that invertebrate response to forest fertilization has not yet been demonstrated.

With regard to the question "Can macroinvertebrates be used to monitor the impacts of fertilization?" we are reluctant to advocate using these organisms as indicators of the response of stream ecosystems to fertilizer applications. Stay et al. (1979) examined aquatic invertebrates after fertilization and found no significant shifts in community structure. Likewise, Meehan et al. (1975) were unable to detect any changes in benthic invertebrates in southeastern Alaska streams. Aquatic invertebrates are notoriously patchy in their distribution and are therefore difficult to sample quantitatively; in addition, processing invertebrate samples is very time consuming and expensive. In theory, macroinvertebrate abundance should reflect significant changes in availability of algae and other organic matter sources, but any macroinvertebrate monitoring program should not be undertaken lightly.

Compared to spikes caused by fertilization, how significant are nutrient releases caused by decomposition of salmon carcasses?

Nutrients released by decomposing salmon can be a significant source of nitrogen within a stream system (e.g., see Kline et al. 1990). In the Douglas-fir region, and particularly in coastal Oregon, Washington, and the Columbia River Basin, returns of naturally spawning salmon have generally declined to levels far below what they were historically (Nehlsen et al. 1991). With fewer salmon returning to streams and with the short residence time of the carcasses (many carcasses are removed from streams by scavenging wildlife), it is unlikely that either peak nitrogen concentrations or long-term nitrogen increase caused by carcasses or dead eggs exceed the magnitude of increases associated with forest fertilization. However, this generalization may not hold for streams with very large runs of species such as sockeye salmon, chum salmon (*O. keta*), and pink salmon (*O. gorbuscha*) that tend to spawn at high densities in limited areas.