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Chapter 15

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

PETER A. BISSON, GEORGE G. ICE, CHRIS J. PERRIN, and ROBERT E. BILBY

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

P.A. Bisson is Aquatic Biologist, Weyerhaeuser Company, Tacoma, Washington. G.G. Ice is Forest Hydrologist, National Council of the Paper Industry for Air and Stream Improvement, Corvallis, Oregon. C.J. Perrin is President, Limnotek Research and Development, Vancouver, British Columbia. R.E. Bilby is Aquatic Ecologist, Weyerhaeuser Company, Tacoma. The authors gratefully acknowledge the constructive suggestions of two anonymous referees.

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.

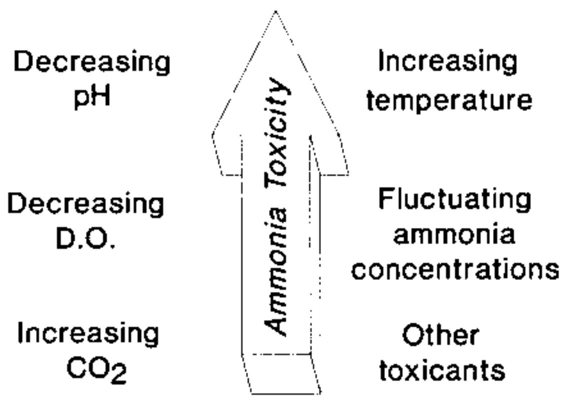

Figure 1. Some environmental factors contributing to increased toxicity of ammonia to fishes. D.O. = dissolved oxygen. Based on several papers by R.V. Thurston and colleagues.

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-

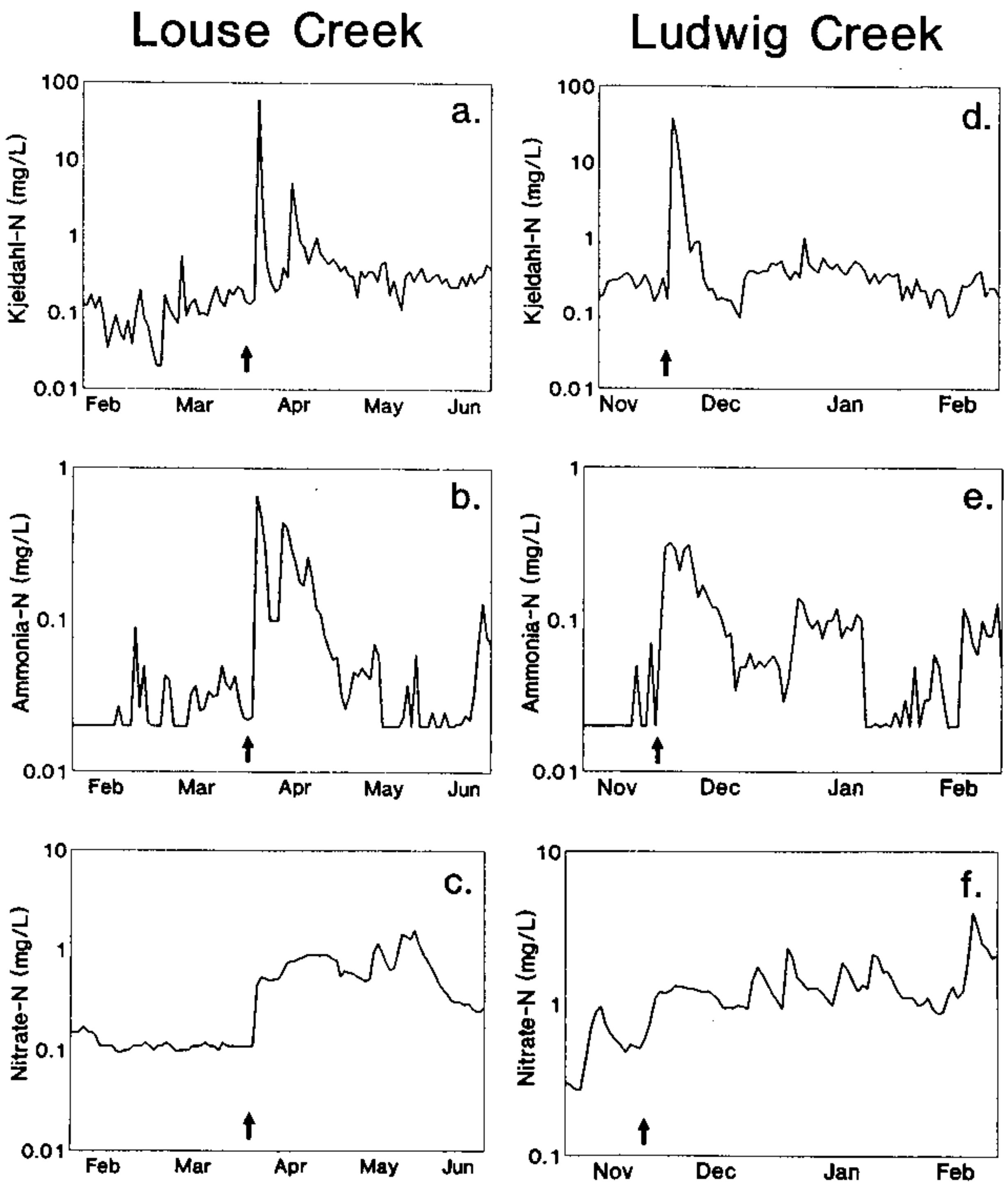

Figure 2. Changes in dissolved nitrogen species after urea application (224 kg N/ha) in Louse Creek, a midelevation stream in the Cascade Range, and Ludwig Creek, a low elevation stream in the Coast Range. Arrows indicate time of fertilization. From P. Bisson and B. Fransen, unpublished.

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

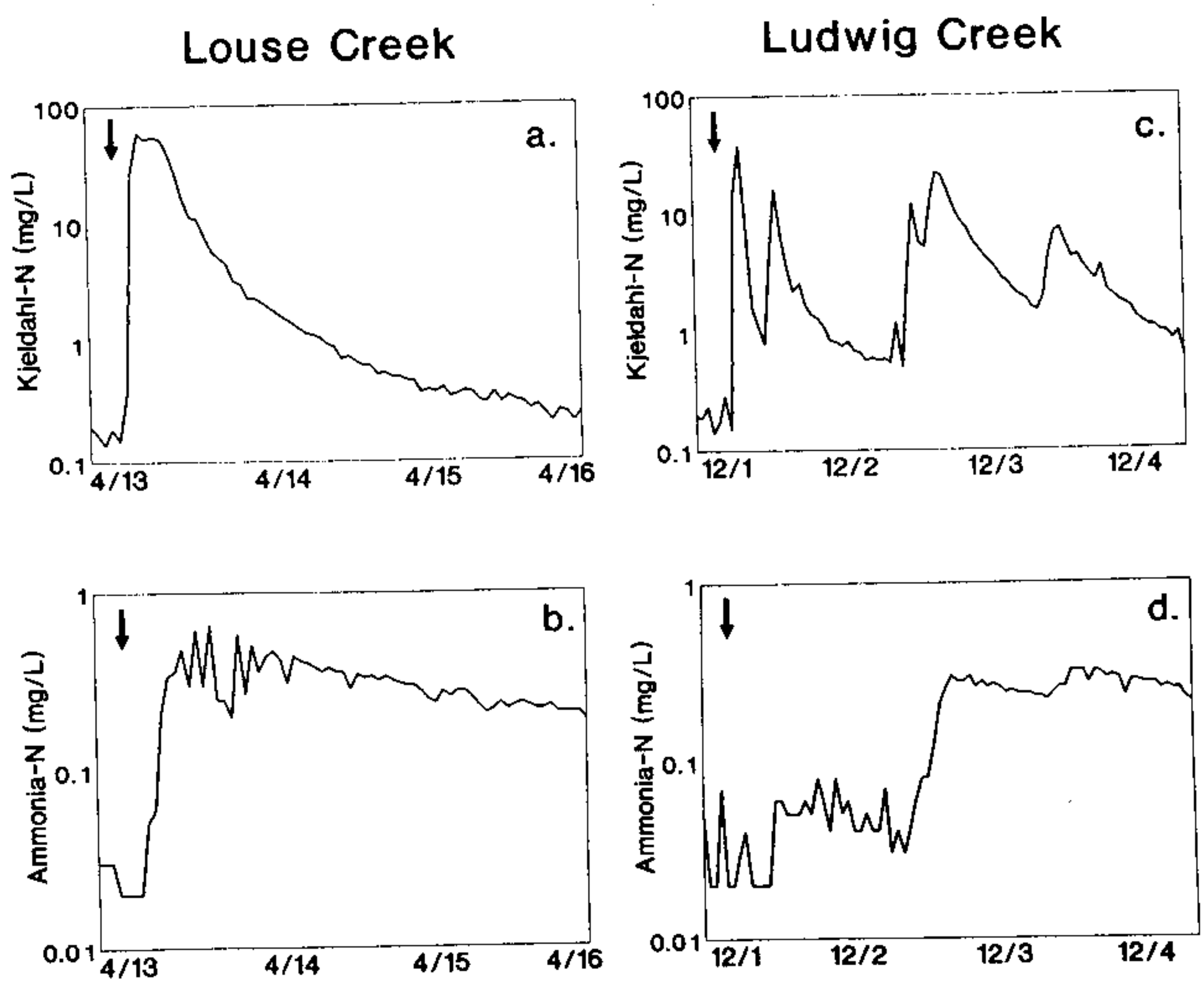

Figure 3. Hourly changes in Kjeldahl-N and ammonia-N during the first three days after urea application in the Louse Creek and Ludwig Creek watersheds. Arrows indicate time of fertilization. From P. Bisson and B. Fransen, unpublished.

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.