U.S. Department of the Interior
U.S. Geological Survey

Ecological Effects on Streams from Forest Fertilization—Literature Review and Conceptual Framework for Future Study in the Western Cascades

By CHAUNCEY W. ANDERSON

Water-Resources Investigations Report 01–4047

Prepared in cooperation with
the BUREAU OF LAND MANAGEMENT

Portland, Oregon: 2002

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, Secretary

U.S. GEOLOGICAL SURVEY
CHARLES G. GROAT, Director

For additional information contact:

District Chief
U.S. Geological Survey
10615 S.E. Cherry Blossom Drive
Portland, OR 97216-3159
E-mail: info-or@usgs.gov
Internet: http://oregon.usgs.gov

Copies of this report can be purchased from:

USGS Information Services
Box 25286, Federal Center
Denver, CO 80225-0046
Telephone: 1-888-ASK-USGS

Suggested citation:

Anderson, C.W., 2002, Ecological effects on streams from forest fertilization—Literature review and conceptual framework for future study in the western Cascades: U.S. Geological Survey Water-Resources Investigations Report 01–4047, 49 p.

CONTENTS

TABLES

FIGURES

CONVERSION FACTORS AND ABBREVIATIONS

Multiply	By	To obtain
acre	0.4047	hectare
pounds per acre per year [(lb/acre)/yr]	1.121	kilograms per hectare per year (kg/ha/yr)

Milligrams per liter (mg/L) is equivalent to parts per million; micrograms per liter (µg/L) is equivalent to parts per billion.

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows: °F=1.8 (°C) +32.

Specific conductance is expressed in microsiemens per centimeter at 25 degrees Celsius (µs/cm at 25°C).

Sea level: In this report, “sea level: refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called Sea Level Datum of 1929.

ACKNOWLEDGMENTS

This project was conducted with the Roseburg Office of the Bureau of Land Management (BLM). Craig Kintop, silviculturalist, was instrumental in obtaining funding and for encouraging credible, objective science about the effects of fertilization on stream biota. BLM hydrologists Lowell Duell and Ed Rumbold provided local logistical support (including maps and stream access), helped with sampling, and provided ideas and interest. Dayne Barron and Anne Boeder were pivotal as managers of the project within the BLM. Many others provided insights into literature and processes of forest fertilization, and stream ecology, including especially Drs. Stan Gregory and Steve Wondzell at Oregon State University. Analytical data were provided by the Cooperative Chemical Analytical Laboratory (CCAL) established by memorandum of understanding number PNW-82–187 between theU.S. Forest Service and the Department of Forest Science, Oregon State University. Within the U.S. Geological Survey, Don Major, Bruce Bury, and Niels Luthold helped brainstorm about effects of fertilizer on biological species, especially amphibians, and acted as guides in the Wolf Creek Basin. Charlie Patton provided advice on analyses of nutrients at low concentrations. Frank Triska, John Duff, and Carol Kendall helped with ideas about how to look for the applied fertilizers and their effects, and Kurt Carpenter, Steve Hinkle, and Joe Rinella participated in many discussions of hydrology, nutrient dynamics, and stream ecology.

Ecological Effects on Streams from Forest Fertilization—Literature Review and Conceptual Framework for Future Study

By Chauncey W. Anderson

Abstract

Fertilization of forests with urea-nitrogen has been studied numerous times for its effects on water quality. Stream nitrogen concentrations following fertilization are typically elevated during winter, including peaks in the tens-of-thousands of parts per billion range, with summer concentrations often returning to background or near-background levels. Despite these increases, water-quality criteria for nitrogen have rarely been exceeded. However, such criteria are targeted at fish toxicity or human health and are not relevant to concentrations that could cause ecological disturbances. Studies of the responses of stream biota to fertilization have been rare and have targeted either immediate, toxicity-based responses or used methods insensitive to ongoing ecological processes. This report reviews water-quality studies following forest fertilizations, emphasizing Cascade streams in the Pacific Northwest and documented biological responses in those streams. A conceptual model predicting potential ecological response to fertilization, which includes effects on algal growth and primary production, is presented. In this model, applied fertilizer nitrogen reaching streams is mostly exported during winter. However, some nitrogen retained in soils or stream and riparian areas may become available to aquatic biota during spring and summer. Biological responses may be minimal in small streams nearest to application because of light limitation, but may be elevated downstream where light is sufficient to allow algal growth. Ultimately, algal response could be greatest in downstream reaches, although ambient nutrient concentrations remain low due to uptake and benthic nutrient recycling. Ground-water flow paths and hyporheic processing could be critical in determining the fate of applied nitrogen. A framework is provided for testing this response in the Little River watershed, a tributary to the North Umpqua River, Oregon, at basic and intensive levels of investigation.

INTRODUCTION

Fertilization of public and private timberlands with nitrogen to boost forest productivity has been common in the Pacific Northwest and elsewhere since the late 1960's (Fredriksen et al., 1975; Binkley et al., 1999), and more frequent use of fertilization is anticipated in the future (National Council of the Paper Industry for Air and Stream Improvement [NCASI], 1999). During 1990–98, over 850,000 acres, or approximately 5 percent of Oregon's timberland, were fertilized (Oregon Department of Forestry, 1999), averaging about 95,000 acres a year. Since 1992, most fertilization has occurred on private timberlands; however, applications averaging 16,000–36,000 acres per year continued to State and Federal lands from 1997–99 (Oregon Department of Forestry, 1999). Regionally, over 120,000 acres of forest lands were fertilized each year in the Pacific Northwest during the late 1980's, and in the southeastern United States over 850,000 acres of pine plantations were fertilized in 1996 alone (Binkley et al., 1999). Forest fertilization also is practiced in other parts of the world, including Japan, Australia, New Zealand, and Sweden.

This report reviews literature on effects of forest fertilization on water quality, emphasizing Cascade streams in the Pacific Northwest and possible ecological effects on aquatic systems in those streams. Although the focus is on streams, the initial discussion describes interactions of fertilizers with soils to the extent that they influence nutrient transport to streams. A brief review of literature on processing of nutrients in underground near-stream (hyporheic and riparian) and in-stream (water and benthic) environments also is presented. Next, a conceptual framework for future evaluation of these effects is developed. Finally, an example study plan is provided for examining the possible operational fertilization of urea-nitrogen to selected areas of the Little River Adaptive Management Area (LRAMA) in southwestern Oregon (fig. 1). Water-quality issues there include occurrences of high pH due to excessive algal productivity and the degree to which nuisance algae are enhanced by forestry. Data from reconnaissance surveys in the LRAMA are provided to indicate stream water-quality and algal conditions prior to public timberland fertilization. The suitability of those areas for studying fertilization's effects also is evaluated.

In this report references are made to streams and watersheds of different sizes that are often nested within larger river basins. To avoid confusion, a consistent set of terminology proposed by McCammon (1994) is used. The term "river basin" is used to refer to the equivalent of a U.S. Geological Survey (USGS) third field accounting unit (Seaber et. al., 1987), generally the largest of the waterbodies, such as the North Umpqua River Basin, considered in the report. The term "watershed" refers to the equivalent of USGS fifth field cataloging units, a subunit of a river basin such as the Little River watershed. Successively smaller hydrologic units are referred to as "subwatersheds" (Wolf Creek subwatershed) and "drainages" (West Fork Wolf Creek drainage).

LITERATURE REVIEW

The possibility of negative effects on stream-water quality from runoff of fertilizer nitrogen has long been recognized (Cole and Gessel, 1965). Forest fertilizer losses to streams and their effects on water quality have been studied often (Fredriksen et al., 1975; Moore, 1975; Bisson et al., 1992; Binkley and Brown, 1993a; Binkley et al., 1999). Water-quality criteria for nutrients have rarely been exceeded as a result of fertilization; however, few studies have examined the more subtle biological effects of fertilizer-nutrient inputs to streams. Table 1 provides an overview of data and findings of the relevant studies from the Pacific Northwest and several from other regions, and is referred to throughout this report.

Forest Processes

A large body of literature exists on forest fertilization, including proceedings from at least three conferences (Gessel et al., 1979; Lousier et al., 1991; Chappell et al., 1992). However, most reports are directed at the efficacy of using fertilizers to enhance tree growth and nutrition (see also Haase and Rose, 1997), forest economics, soil processes, and fate of added nutrients in soils and trees. There are also over 25 reports worldwide on the effects of fertilization on water quality in receiving waters, and periodic reviews (Fredriksen et al., 1975; Moore, 1975; Bisson et al., 1992; Binkley and Brown, 1993a; Binkley et al., 1999). Of these reports, only three evaluated biological effects (Groman, 1972; Meehan et al., 1975; Stay et al., 1979), and none focussed on both soil processes and stream water or linkages between them (Binkley et al., 1999).

Tree growth in the Pacific Northwest and many other locations is generally believed to be constrained by available nitrogen (Cole., 1979; Johnson, 1992; Fenn et al., 1998). For this reason, young (15–40-year-old) commercial forest stands are often fertilized with nitrogen, typically as urea $[(NH_2)_2CO]$ pellets, although ammonium sulfate, ammonium nitrate, and various phosphate fertilizers also have been used (Klock, 1971; Tiedemann et al., 1978; Russel, 1979; Nason and Myrold, 1992). Urea pellets (known as "prill") consist of

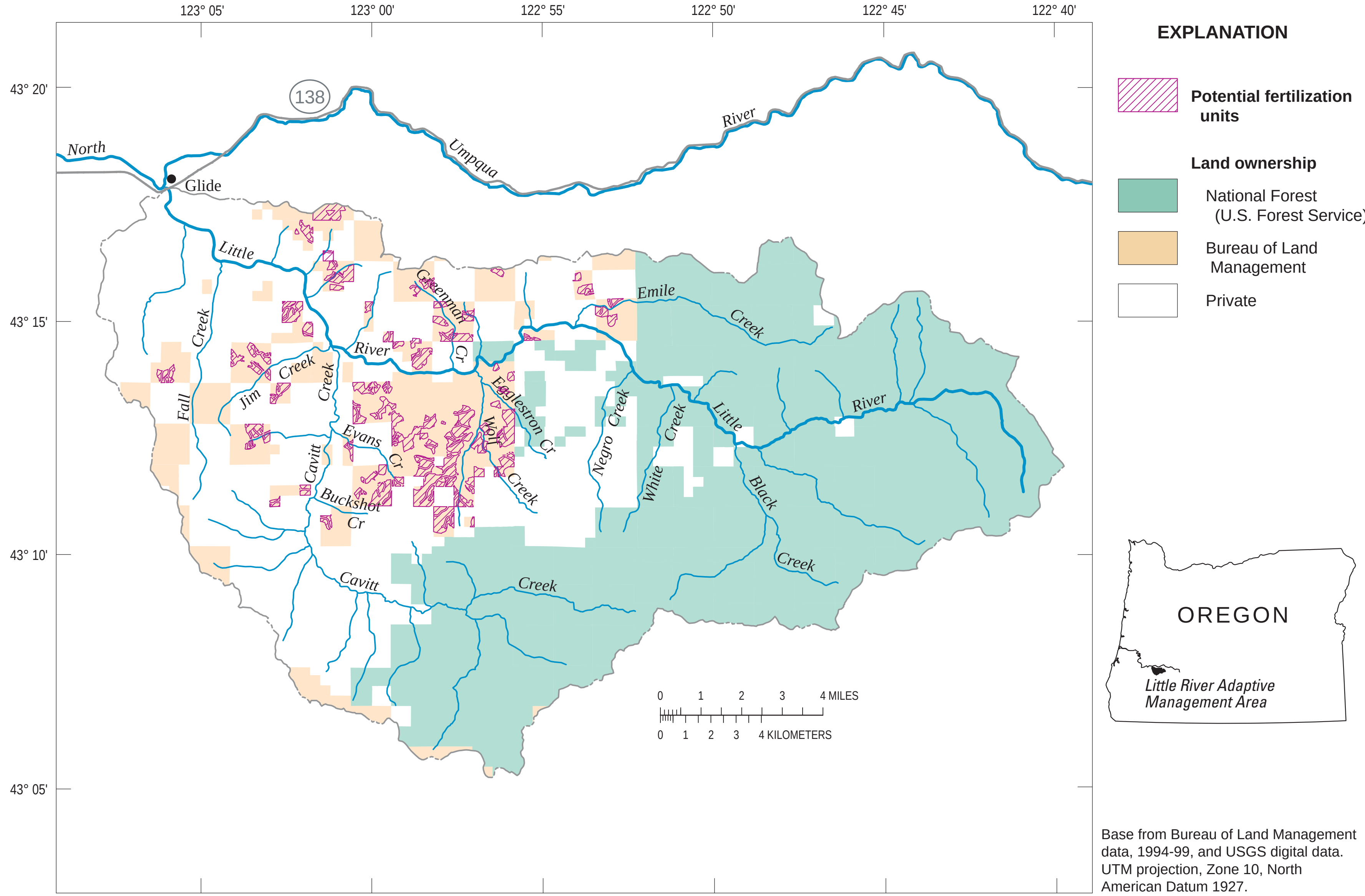


Figure 1. Map showing Little River watershed, Oregon, and proposed Bureau of Land Management fertilization units.

Table 1. Summary of studies of forest fertilization effects on stream-water quality

[µg/L, micrograms per liter; vs, versus; %, percent; NH_3, ammonia; NO_3, nitrate; 1 kg/ha (kilogram per hectare), 0.89 lb/ac (pound per acre); 1kg Urea, 0.46 kg N; NR, not reported; TKN, total Kjeldahl nitrogen; NA, not applicable; d, day; ~, approximately, value is interpreted from report; <, less than; >, greater than; x, times; N, nitrogen; NO_2, nitrite; Tot-N, total-N (sum of TKN and NO_3+NO_2); inverts, invertebrates; chl *a*, chlorophyll *a*; mo, months; yr, year; mg/kg, milligrams per kilogram (parts per million); mg/ L, milligrams per liter; m, meter; km, kilometer;

Geographical area and streams studied	Baseline nitrogen concen-tration (µg/L)	Maximum post-treatment concentration (µg-N/L)	Estimated period and average magnitude of elevated concentration[1], vs control or baseline	% loss to streams	Biological components studied and results	Sampling design / Remarks
Location: Santiam Basin, Oregon; Application rate: 224 kg N/ha urea; Date: May 1969; Objectives: Effects of fertilization on water quality; Reference: Malueg et al., 1972						
Crabtree Creek (control was upstream from study site)	NH_3 <10 NO_3 <10 TKN 400	NH_3 80 NO_3 250 TKN 24,000	NH_3 >100d, ~3x NO_3 >7 mo, 1.5–2x TKN 2d, ~75x	NR	NR	Also performed assays on urea pellets (NH_3-N, 140 mg/kg; NO_2-N, 0.53 mg/kg; NO_3-N, 19.33 mg/kg; TKN, 440,000 mg/kg). Nitrate returned to baseline1 during summer but peaked again in fall with precipitation.
Location: Tahuya River, Kitsap Peninsula, WA.; Application rate: 227 kg N/ha; Date: October 1972; Objectives: Effects of fertilization on water quality; Reference: Cline, 1973						
Site 1, control (upstream)	NH_3 10–80 NO_3 0–200 Urea 0–10	NH_3 <10 NO_3 470 Urea 50	NA	NA	NR	Water–quality data collected at all sites for 1 yr prior to fertilization. In general, NO_3 responded to flow. Conditions were dry for ~31 days after fertilization.
Site 2, treatment (no buffer strip)	NH_3 10–80 NO_3 40–210 Urea 10–20	NH_3 1,400 NO_3 1,830 Urea 27,000	NH_3 25d, ~30–60x NO_3 ~7.5 mo, ~8x Urea 6d, ~40x	NR	NR	Lack of buffers contributed to increased nutrient concentrations compared to site 3. NH_3 peaks were also more immediate than at site 3 due to direct application.
Site 3,treatment (buffer strip)	NH_3 0–60 NO_3 0–260 Urea 10–20	NH_3 160 NO_3 680 Urea 4,300	NH_3 2d, 10–40x NO_3 ~7.5 mo, ~3x Urea 6d, ~40x	NR	NR	NH_3 peaks were delayed by more than a month and lowered due to dry weather in fall.
Sites 4 & 5, downstream sites	NH_3 0–80 NO_3 0–350 Urea 0–30	NH_3 60 NO_3 470 Urea 40	NH_3 ~31d, ~3–5x NO_3 ~31d, ~4x Urea ~3d, ~2x	.45–1%	NR	NH_3 peaks were delayed by more than a month and lowered due to dry weather in fall. Loss of nitrogen reported is for the entire study area (sites 2, 3, 4, and 5) upstream compared to control.
Location: SE Alaska; Application rate: 210 kg urea-N/ha; Date: May 1970; Objectives: MCL, NH_3 toxicity exceedances; Reference: Meehan et al., 1975						
Falls Creek, control	NH_3 ~20 NO_3 ~10	NH_3 ~100 NO_3 ~200	NA	NR	Benthic inverts, algal biomass on Plexiglas slides. No significant differences found between treatment and control. High vari-ability. No species data taken	Application was to recently logged watersheds. Water sampled daily for first month after application, weekly for second month, and monthly for 1.5 yrs. Very low stream temperatures, average pH 6.5–7.2. Three Lakes unit dried up during summer. Phosphorus did not respond to fertilization.
Falls Creek, treatment	NH_3 ~20 NO_3 ~20	NH_3 1,280 NO_3 ~1,600	NH_3 ~ 1.5 mo, 10–20x NO_3 ~ 14 mo, 5–10x	NR		
Three Lakes Creek, control	NH_3 ~20 NO_3 ~20	NH_3 ~100 NO_3 ~300	NA	NR		
Three Lakes Creek, treatment	NH_c ~50 NO_3 ~10	NH_3 ~100 NO_3 2,360	NH_3 ~5d, ~3x NO_3 ~1.5 mo, >5x	NR		
Location: 6 locations in Pacific Northwest; Application rate: 224 kg urea-N/ha; Date: March-April, 1970–72; Objectives: not reported; Reference: Moore, 1971; Fredriksen et al., 1975						
Coyote Creek, South Umpqua Experimental Forest	NH_3 5 NO_3 2 Urea 6	NH_3 48 NO_3 177 Urea 1,390	NH_3 ~5d, ~2x NO_3 ~2 mos, ~5–10x Urea ~15d, ~10x	0.01%	NR	After 3–6 weeks all loss of N was as NO_3. Little or no loss of N during summer months, but NO_3 had a second peak during rains the following fall(~170 µg/L). 92% of N lost during first year was during storms the following fall. 100% of watershed area treated, old-growth mixed conifers.

Table 1. Summary of studies of forest fertilization effects on stream-water quality—Continued

[µg/L, micrograms per liter; vs, versus; %, percent; NH_3, ammonia; NO_3, nitrate; 1 kg/ha (kilogram per hectare), 0.89 lb/ac (pound per acre); 1kg Urea, 0.46 kg N; NR, not reported; TKN, total Kjeldahl nitrogen; NA, not applicable; d, day; ~, approximately, value is interpreted from report; <, less than; >, greater than; x, times; N, nitrogen; NO_2, nitrite; Tot-N, total-N (sum of TKN and NO_3+NO_2); inverts, invertebrates; chl *a*, chlorophyll *a*; mo, months; yr, year; mg/kg, milligrams per kilogram (parts per million); mg/ L, milligrams per liter; m, meter; km, kilometer;

Geographical area and streams studied	Baseline nitrogen concentration (µg/L)	Maximum post-treatment concentration (µg-N/L)	Estimated period and average magnitude of elevated concentration[1], vs control or baseline	% loss to streams	Biological components studied and results	Sampling design / Remarks
Location: 6 locations in Pacific Northwest; Application rate: 224 kg urea-N/ha; Date: March-April, 1970–72; Objectives: not reported; Reference: Moore, 1975; Fredriksen et al., 1975—Continued						
Trapper Creek, Olympic National Forest	NH_3 0 NO_3 34 Urea 8	NH_3 10 NO_3 121 Urea 700	NR	NR	NR	<10% of watershed area treated, 40-year-old Douglas fir stands.
Jimmy-Come-Lately Cr., Olympic National Forest	NH_3 0 NO_3 5 Urea 2	NH_3 40 NO_3 42 Urea 708	NO_3 9 weeks	NR	NR	<10% of watershed area treated, 10-year-old Douglas fir stands.
Nelson Creek, Siuslaw River Basin	NH_3 10 NO_3 290 Urea <20	NH_3 320 NO_3 2100 Urea 8,600	NR	NR	NR	100% watershed area treated, young Douglas fir growth.
Dollar Creek, McKenzie River Basin	NH_3 30 NO_3 60 Urea <20	NH_3 490 NO_3 130 Urea 44,400	NR	NR	NR	100% of watershed area treated, young Douglas fir growth.
Pat Creek, Yamhill River Basin	NH_3 7 NO_3 70 Urea 3	NH_3 34 NO_3 388 Urea 3,260	NR	NR	NR	63% of watershed area treated, 35-year-old Douglas fir growth.
Location: 25 Locations on 9 streams in Oakridge Ranger District, Willamette National Forest, Oregon; Application rate: 225 kg N/ha urea; Date: April 1976; Objectives: Determine effects on selected chemical and biological aspects of streams; Reference: Stay et al., 1978, Stay et al., 1979						
Site 25, control	NH_3 5 NO_3 5 TKN 87 Urea ND	NH_3[2] 13 NO_3[2] 5 TKN[2] 63 Urea[2] 20	NA	NR	Algal assays—no response; Periphyton—small increase in chl *a* but not biomass; Macroinverts—no direct change evident; Fish—no mortalities evident	By extending data collection through July 1977, Stay et al. (1979) observed changes from fertilization that were not observed through December 1976 by Stay et al. (1978); these included small increase in NO_3-N in fertilized streams and differences in N-runoff between streams with 30 m and 45 m buffer strips. Some increases were also found in specific conductance and total cation concentrations. Algal assays using a green alga (*Selenastrum capricornutum)* indicate colimitation by N and P. Stay et. al. (1979) state that colimitation by P helped minimize algal response to added N. Invertebrate changes appeared more tied to seasonal variability than to fertilization.
Treatments—24 sites (*Ranges indicate reported concentrations from many sites*)	NH_3 5 NO_3 5–10 TKN 47–100 Urea ND	NH_3 11 NO_3 26 TKN 2,380 Urea 8,000	NH_3 no difference NO_3 ~1 yr, 1–3x TKN <30d, <1–3x Urea <30d, ~1.5x	NR		
Location: Vancouver Island, B.C.; Application rate: 200 kg N/ha; Date: November 1979; Objectives: Effects of fertilization on water quality in streams and downstream lake; Reference: Perrin et al., 1984						
2 control streams	NH_3 <4 NO_3 1–27 Urea <5	NH_3 15 NO_3 110 Urea 20	NA	NA	NR	Lower concentrations and longer transport times observed for streams with buffer strips than without buffer strips. Cold temperatures may have caused reduced nitrification resulting in longer time for urea and NH_3 to return to baseline concentrations (relative to other studies) and lower NO_3 concentrations. Forest fertilization caused shift from N-limitation to P-limitation in downstream lake, & algal blooms.
12 sites on 10 streams draining 3 treatment watersheds entering a lake	NH_3 <4 NO_3 1–58 Urea <5	NH_3 4,780 NO_3 790 Urea 57,000	NH_3 79–136d[3] NO_3 4–84d[3] Urea 102–140d[3]	2.1–5.2%	NR	

Table 1. Summary of studies of forest fertilization effects on stream-water quality—Continued

[µg/L, micrograms per liter; vs, versus; %, percent; NH_3, ammonia; NO_3, nitrate; 1 kg/ha (kilogram per hectare), 0.89 lb/ac (pound per acre); 1kg Urea, 0.46 kg N; NR, not reported; TKN, total Kjeldahl nitrogen; NA, not applicable; d, day; ~, approximately, value is interpreted from report; <, less than; >, greater than; x, times; N, nitrogen; NO_2, nitrite; Tot-N, total-N (sum of TKN and NO_3+NO_2); inverts, invertebrates; chl *a*, chlorophyll *a*; mo, months; yr, year; mg/kg, milligrams per kilogram (parts per million); mg/ L, milligrams per liter; m, meter; km, kilometer;

Geographical area and streams studied	Baseline nitrogen concen-tration (µg/L)	Maximum post-treatment concentration (µg-N/L)	Estimated period and average magnitude of elevated concentration[1], vs control or baseline	% loss to streams	Biological components studied and results	Sampling design / Remarks
Location: Vancouver Island, B.C.; Application rate: 224 kg N/ha; Date: September 1974; Objectives: Effects of fertilization on water quality; Reference: Hetherington, 1985						
TC, control	NH_3 0–131 NO_3 0–10 Urea 0–20	NH_3 61 NO_3 300 Urea 540	NA	NA	NR	Previously fertilized in 1967 at 96 kg N/ha.
16M, control	NH_3 0–93 NO_3 4–109 Urea 0–20	NH_3 22 NO_3 89 Urea 10	NA	NA	NR	Previously fertilized in 1967 at 258 kg N/ha.
TF1 (Lens Creek), treatment 40-year-old plantation	NH_3 0–79 NO_3 7–177 Urea 0–30	NH_3 540 NO_3 2,700 Urea 14,000	NH_3 13d NO_3 >14 mos Urea 6d	5.9%	NR	No buffer strips. 46% of watershed area fertilized. Continually flowing stream. Previously fertilized in 1968 and 1972 at 258 kg N/ha. 98% of N-loss was as nitrate. Fall rains in 1975 caused increases in nitrate and urea.
TF2, treatment	NH_3 0–80 NO_3 28–151 Urea 0–220	NH_3 1,900 NO_3 9,300 Urea 790	NH_3 15d[3,4] NO_3 ~14 mos, ~9x Urea 14d[3]	14.5%	NR	No buffer strips. 80% of watershed area fertilized. Intermittent streamflow. Previously fertilized in 1967 at 96 kg N/ha. 92% of N-loss was as nitrate. Fall rains in 1975 caused increases in nitrate and urea. Wetlands may have contributed to higher N-loss compared to TF1.
L, downstream site (Receives combined flow from both TF1 and TF2)	NH_3 0–119 NO_3 38–215 Urea 0–23	NH_3 360 NO_3 720 Urea 160	NH_3 33d[3] NO_3 ~14 mos[3] Urea 5d[3]	NR	NR	Located ~2 km downstream from TF1. Nitrate and ammonium increases were delayed until November 1974 after first substantial rains.
Location: Western Washington; Application rate: 224 kg urea-N/ha; Date: July 1980; Objectives: determine water-quality effects of annual fertilizations; Reference: Bisson, 1982[5]						
Hook Creek, "control"	NH_3 3 NO_3 262 Tot-N 113	NH_3 25 NO_3 268 Tot-N 488	NA	1.9–9%	NR	"Heavily fertilized within 3 yrs before study" (control).
Willow Creek, Treatment—annual application.	NH_3 6 NO_3 96 Tot-N 91	NH_3 159 NO_3 458 Tot-N 8,597	NH_3 40d, ~5x NO_3 77d, ~1.5x Tot-N 77d, ~3x		NR	"Heavily fertilized within 3 yrs before study", plus applications of 65 kg/ha in first yr of study and annually afterwards (treatment).
Needle Creek, "Control"	NH_3 77 NO_3 1,270 Tot-N 874	NH_3 1,580 NO_3 2,000 Tot-N 4,400	NA	NR	NR	"Heavily fertilized within 3 yrs before study" (control).
Gate Creek, Treatment—65 kg N/ha	NH_3 10 NO_3 1,232 Tot-N 1,168	NH_3 186 NO_3 2,310 Tot-N 9,595	NH_3 40d, ~1.5x NO_3 > 7 mos, ~2x Tot-N > 7 mos,~3x	NR	NR	"Heavily fertilized within 3 yrs of study", plus applications of 65 kg/ha in first yr of study and annually afterwards (treatment). Extensive fertilization history regarded as cause of high N-export through increased nitrification.
Debris Creek[6, 7], Treatment—224 kg N/ha	NH_3 5 NO_3 211 Tot-N 105	NH_3 630 **NO_3 1,570** Tot-N 4,380	NA	NR	NR	"Relatively little past fertilization" plus application of 224 kg N/ha in first yr of study and annual treatment afterwards.
Eleven Creek[6, 7], Treatment—224 kg N/ha	NH_3 2 NO_3 131 Tot-N 44	NH_3 752 NO_3 1,680 Tot-N 37,553	NA	NR	NR	"Relatively little past fertilization" plus application of 224 kg N/ha in first yr of study, and annual treatment afterwards. High Tot-N was urea from direct application. High loss may also have been due to direct application on snow.

Table 1. Summary of studies of forest fertilization effects on stream-water quality—Continued

[µg/L, micrograms per liter; vs, versus; %, percent; NH_3, ammonia; NO_3, nitrate; 1 kg/ha (kilogram per hectare), 0.89 lb/ac (pound per acre); 1kg Urea, 0.46 kg N; NR, not reported; TKN, total Kjeldahl nitrogen; NA, not applicable; d, day; ~, approximately, value is interpreted from report; <, less than; >, greater than; x, times; N, nitrogen; NO_2, nitrite; Tot-N, total-N (sum of TKN and NO_3+NO_2); inverts, invertebrates; chl *a*, chlorophyll *a*; mo, months; yr, year; mg/kg, milligrams per kilogram (parts per million); mg/ L, milligrams per liter; m, meter; km, kilometer;

Geographical area and streams studied	Baseline nitrogen concen-tration (µg/L)	Maximum post-treatment concentration (µg-N/L)	Estimated period and average magnitude of elevated concentration[1], vs control or baseline	% loss to streams	Biological components studied and results	Sampling design / Remarks
Location: Western Washington; Application rate: various (see below); Date: various dates in 1988; Objectives: Drinking water criteria, dissolved N toxicity, and individual objectives in each basin; Reference: Bisson, 1988.						
Forks Creek[7] Treatment averaged 207 kg-N/ha, on 1/88 and 2/88	NH_3 <20 NO_3 30 TKN 80	NH_3 40 NO_3 50 TKN 160	NH_3 40d, ~2x NO_3 2d, ~1.5x TKN >30d ~2x	NR	NR	Project also tested if fertilization jeopardized water quality at downstream fish hatchery or caused algal fouling of water intake system. Sampling only for ~30 days after application.
Spring Creek[7], Treatment averaged 130 kg-N/ha, on 2/88	NH_3 <20 NO_3 1,000 TKN 70	NH_3 <20 NO_3 1,500 TKN 180	NH_3 no increase NO_3 >30d, ~1.5x TKN >15d, ~2x	NR	NR	Tributary to Forks Creek
Silver Lake Basin, Hemlock and Sucker Creeks, Treatment—92 kgN/ha each	NH_3 20 NO_3 800 TKN 300	NH_3 200 NO_3 800 TKN 1,500	NH_3 >100d, 3-4x NO_3 no increase TKN no increase	NR	NR	History of fertilizer application every ~5 yrs since 1969, water quality monitored after each application. Current fertilization in February 1988. Silver lake is eutrophic with extensive macrophyte beds.
Ryderwood, Pair 1[7] (Campbell Creek) Treatment—92 kg N/ha	NH_3 30 NO_3 90 TKN 100	NH_3 275 NO_3 580 TKN 2,000	NH_3 >100d, ~2-5x NO_3 100d, ~2x TKN >100d, ~3x	NR	NR	Tributaries to Cowlitz River. No buffer strips in "treatment" watershed, "control" watershed was recent clear cut. Peak NO_3 concentration in clearcut "control" was higher than in "treatment".
Ryderwood, Pair 2[7] (Arkansas Creek) Treatment—92 kg N/ha	NH_3 20 NO_3 200 TKN 100	NH_3 150 NO_3 600 TKN 3,750	NH_3 >100d, ~3x NO_3 ~75d, ~2x TKN >100d, 2x	NR	pHs averaged 6.5–7.0, increased ~0.3 units	Paired locations on Arkansas Creek (buffered, with unbuffered tributaries); upstream=control, downstream=treatment. High TKN due to direct application
Location: Fernow Exp. Forest, W. Virginia; Application rate: 336 kg N/ha as ammonium nitrate plus 224 kg P/ha as triple superphosphate[8]; Date: April 1976; Objectives: Selected water-quality responses in streams, and cumulative downstream effects, tracked from 3 to 10 years; Reference: Helvey et al., 1989; Edwards et al., 1991						
North and South Facing Watersheds	NO_3 ~500 Ca 2 mg/L	NO_3 ~10,000 Ca 10 mg/L	NO_3 >10 yrs, >5x Ca >3 yrs, ~3x Mg >3 yrs, ~3x	N[9] 23–27% P–<1%	NR	Specific conductance increased from ~28 to 140 µS/cm in fertilized watersheds, remained high after 3 yrs, back to background after 10 yrs. After 10 yrs NO_3-N remained ~40% higher than in control stream. Ca and Mg were back to background after 10 yrs except in one watershed. Average pH's in all streams were around 5.0. No apparent changes in P concentrations.
Location: Western Washington; Application rate: 224 kg N/ha; Date: various in 1988–89; Objectives: Not reported; Reference: Bisson et al., 1992						
Louse Creek (western Cascades, second growth Douglas-fir)	NH_3 ~30 NO_3 ~120 TKN ~100	NH_3 ~800 NO_3 ~1000 TKN 80,000	NH_3 >30d, 5-20x NO_3 >90d, 5-10x TKN ~4d, 10-100x	NR	NR	Fertilized April 1989. Virtually all of watershed's area fertilized. Studies ended after 90d, at onset of summer.
Ludwig Creek (Coast Range, second growth Douglas-fir)	NH_3 ~20 NO_3 ~600 TKN ~200	NH_3 ~400 NO_3 ~4,000 TKN 50,000	NH_3 >60d, 3-10x NO_3 >90d, 2-5x TKN >7d, 10-100x	NR	NR	Fertilized December 1988. Virtually all of watershed's area fertilized. Generally more protracted release of N from fertilization than Louse Creek, but Coast Range may have higher N deposition rates and nitrification rates.

Table 1. Summary of studies of forest fertilization effects on stream-water quality—Continued

[µg/L, micrograms per liter; vs, versus; %, percent; NH_3, ammonia; NO_3, nitrate; 1 kg/ha (kilogram per hectare), 0.89 lb/ac (pound per acre); 1kg Urea, 0.46 kg N; NR, not reported; TKN, total Kjeldahl nitrogen; NA, not applicable; d, day; ~, approximately, value is interpreted from report; <, less than; >, greater than; x, times; N, nitrogen; NO_2, nitrite; Tot-N, total-N (sum of TKN and NO_3+NO_2); inverts, invertebrates; chl *a*, chlorophyll *a*; mo, months; yr, year; mg/kg, milligrams per kilogram (parts per million); mg/ L, milligrams per liter; m, meter; km, kilometer;

Geographical area and streams studied	Baseline nitrogen concentration (µg/L)	Maximum post-treatment concentration (µg-N/L)	Estimated period and average magnitude of elevated concentration[1], vs control or baseline	% loss to streams	Biological components studied and results	Sampling design / Remarks
Location: Central Sweden; Application rate: 150 kg N/ha as ammonium nitrate; Date: August, 1986; Objectives: Simulate acidification from N-deposition, and evaluate effects on water-quality, invertebrates, and fish; Reference: Göthe et al., 1993						
Orranstjärnbäcken	NH_3 5 NO_3 15 Ca 1.8 mg/L	NH_3 11,100 NO_3 8,200 Ca 3.1mg/L	NH_3 >1 yr, 2-4x NO_3 2yrs, ~10x Ca 14d, ~2x,	NR	No long-term change in invertebrate species abundance, but there was an increase in drift in both streams, especially at furthest downstream stations. No mortality or density effects on fish.	Rödtjärnbäcken was fertilized with calcium ammonium nitrate. No acidification effects noted in either stream.
Rödtjärnbäcken	NH_3 5 NO_3 10 Ca NR	NH_3 15,400 NO_3 29,600 Ca NR	NH_3 >14d, >3-20x NO_3 2 yrs, ~6-7x Ca 14d, ~3x	NR		

[1]Concentrations expressed as a relative change in the active nutrient or ingredient, per liter.

[2]Concentrations reported are averages rather than maximums.

[3]Average concentrations not reported.

[4]No streamflow during fertilization at TF2. Ammonia-N and nitrate-N concentrations had peaks attributed to fertilization in October and November 1974 after rainstorms.

[5]Baseline concentrations calculated from Bisson (1982) by C.W. Anderson, USGS, 1999.

[6]Debris Creek and Eleven Creek are paired treatment watersheds, with no control watershed.

[7]"Control" concentrations are baseline concentrations in the same stream prior to fertilization.

[8]Calcium phosphate.

[9]N loss of 27% includes estimated loss in ground water.

46% N, and are usually applied at a rate of 224 kg N/ha (200 lb N/ac). This rate represents several decades of refinement, providing a reasonable economic tradeoff of tree growth to N-loss (Miller and Fight, 1979; Mika et al., 1992) through various processes (described below).

After application, nitrogenous fertilizers undergo numerous reactions, and substantial literature has identified and quantified these reactions, including N-incorporation into trees and (or) N-losses. Some of those reactions are summarized here, with an emphasis on urea fertilizers and processes that could affect stream water quality.

Reactions of applied urea-N include volatilization as ammonia, foliar uptake, hydrolysis to ammonium-N (NH_4^+-N)[1], rapid immobilization into soil organic fractions, ion exchange, mineralization, nitrification, plant (root) uptake, denitrification, and leaching to deeper soils and ground water (fig. 2). The fraction of NH_3-N or urea-N not initially volatilized or taken up into foliage (including trees) is usually hydrolyzed to NH_4^+-N. Subsequently, NH_4^+-N can be immobilized by conversion to soil organic material or through ion exchange (Matzner et al., 1983; Edwards et al., 1991; Moldan and Wright; 1998), taken up by tree roots, or converted to NO_3-N (nitrified). Hydrolysis may temporarily increase soil pH, which can in turn enhance nitrification (Ochtere-Boateng, 1979). Nitrate is relatively mobile and can be rapidly leached into shallow or deeper ground-water systems, where it can be utilized by root systems or exit the forest through ground water or runoff processes. Mineralization is the conversion of soil organic nitrogen to inorganic N.

The amount of urea-N entering trees, through foliar and plant uptake, ranges from 10 to 30% of that applied (Binkley, 1986; Nason and Myrold, 1992). Thus, up to 70–90% is potentially retained in forest soils or other vegetation, volatilized, or passed from the forest ecosystem (Nason and Myrold, 1992), all of which are considered losses of applied fertilizer. These loss terms are highly

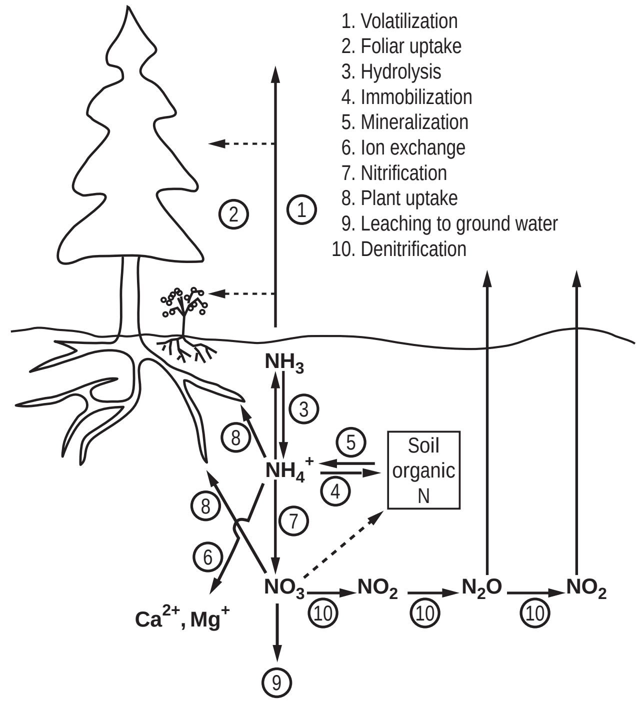


Figure 2. Forest cycling pathways representing major processes and fates of nitrogen fertilizers. (Modified from Nason and Myrold, 1992.)

variable, but the largest are usually volatilization (<1–46%) (Craig and Wollum, 1982; Marshall, 1986, 1991; Nason and Myrold, 1992) and immobilization (averaging about 50–60%) (Nason and Myrold, 1992). Much of the immobilized N apparently remains unavailable (Miller, 1986), so mineralization is likely small. The amount lost to streams, a topic explored in the following section, generally is less than 10% of applied N but can be larger (up to 27%) (table 1) under certain soil and moisture conditions. Denitrification is usually insignificant (Nason and Myrold, 1992), although in anoxic, saturated riparian soils, denitrification can be several orders of magnitude higher than in unsaturated upland soils (Cirmo and McDonnell, 1997). The relative extent of reactions listed above are highly dependent on the type of fertilizer applied. Nitrate-based fertilizers can have substantially different loss rates compared with the urea-based rates given above (Marshall, 1986, 1991). The organic-carbon content of soils and pore-waters is also a critical determinant of N-retention and transformations, both in upland areas as well as riparian regions (Cirmo and McDonnell, 1997; Dahm et. al, 1998; Chestnut and McDowell, 2000).

[1] In this report, the term NH_3-N is used in a general sense to represent the sum of free and ionized ammonia concentrations. Ionized ammonia, also known as ammonium, is referred to specifically as NH_4^+-N when it is intended to mean only the ammonium (ionized) form.

The timing of urea application to forests is partly determined by efforts to minimize losses, especially volatilization and immobilization, the largest loss components. Volatilization is enhanced by higher temperatures, wind, and soil pH, and by low intensity rainfall. High intensity rainfall (though not so high as to significantly increase erosion and runoff) can reduce volatilization. Immobilization, which includes conversion of both fertilizer N and soil inorganic-N into organic forms, is minimized by cool, moist weather (Nason and Myrold, 1992). As a result, urea is typically applied during fall in the Pacific Northwest as weather conditions become cool and wet. (Mika et al., 1992).

Water-Quality Effects on Streams from Forest Fertilization

Immediate Nutrient Runoff

All studies in which fertilizers were applied by helicopter reported some violation of stream buffers, direct applications to water, or inadvertent fertilization of control watersheds. Aerial application of fertilizer to forests is not exact, although recent advances in navigation using global positioning systems (GPS) should enhance the ability of pilots to minimize unintended application to areas not intended for fertilization. Actual application rates using helicopters, as measured on the ground, can vary by 20–60% from the targeted rate (Fredriksen et al., 1975; Binkley, 1986). Direct applications to water have invariably resulted in immediate and relatively high-concentration pulses of urea-N (often measured as organic- or Kjeldahl-N) and NH_3-N (table 1). Peak urea-N concentrations usually exceed 1,000 µg/L (micrograms per liter) and have been reported as high as 80,000 µg/L (Bisson et al., 1992). Peak NH_3-N concentrations have been as high as 4,780 µg/L (Perrin, et al., 1984). In the latter case, ammonia toxicity criteria concentrations (U.S. Environmental Protection Agency, 1986) might have been temporarily exceeded (although temperature and pH data required to make this assessment were not reported). With this exception, however, ammonia toxicity problems have not been reported, though ammonia toxicity is an obvious consideration and hence NH_3-N almost always is monitored.

Typically, pulses of urea-N and NH_3-N decline in concentration and are short-lived following fertilization, usually lasting less than 1 month and often just a few days (table 1), depending on rainfall conditions. Maximum periods of elevation for urea and NH_3-N have been more than 100 days (Malueg et al., 1972; Perrin et al., 1984). Extreme cases (several months or years) (Bisson, 1982; Edwards et al., 1991) were reported in instances where applications were in forests previously perturbed by forest management or excessive N-deposition. Reductions in urea and NH_3-N are generally coincident with soil nitrification, when their supply available for runoff becomes greatly diminished, often to pretreatment or control levels (Bisson et al., 1992; Binkley et al., 1999).

Longer-Term Nutrient Losses to Streams

Concentrations of nitrate-N (NO_3-N) in streams typically remain low immediately after urea application, but increase rapidly during subsequent rainstorms as nitrification proceeds. Peak concentrations (table 1) have ranged from less than 100 µg/L (for example, Fredriksen et al., 1975) to greater than 9,000 µg/L (Hetherington, 1985). In two studies, peak NO_3-N concentrations exceeded EPA drinking water standards (10,000 µg/L); however, these studies were at the Fernow Experimental Forest in West Virginia (Helvey et al., 1989; Edwards et al., 1991) and in Sweden (Göthe et al., 1993), where nitrogen-saturated soils associated with excess atmospheric nitrogen deposition are well-documented (Vitousek et al., 1997; Fenn et al., 1998).

Whereas peak NO_3-N concentrations occur chiefly during high runoff, ambient NO_3-N concentrations in streams draining fertilized watersheds also are typically elevated as much as two- to tenfold, often for the entire winter and spring following a fall fertilization (Bisson et al., 1992). In most cases, however, NO_3-N concentrations essentially return to background levels by summer due to uptake in soils and (possibly) uptake in stream water (Mulholland, 1992; Mulholland and Rosemond, 1992). Where sampling continued beyond the summer following fertilization into subsequent fall and winter periods (Malueg et al., 1972; Moore, 1971; Fredriksen et al., 1975; Meehan et al., 1975; Stay et al., 1979; Hetherington, 1985), a fall NO_3-N peak, which is elevated relative to that in control streams, has been observed. This secondary peak indicates that applied fertilizer nitrogen remains available for leaching to streams beyond the spring and summer growing seasons. An extreme example of long-term availability was reported at Fernow Experimental Forest, where NO_3-N remained elevated relative to control streams 10 years after fertilization with ammonium nitrate at 336 Kg N/ha (Edwards et al., 1991). Aside from the Fernow experiment and a long term watershed acidification study in Maine (Norton et al., 1994; Fry et al., 1995), no studies have followed fertilizer-nitrogen runoff for more than 1–1.5 years.

The amount of applied nitrogen lost to streams varies from less than 0.5% to 14.5%, with the exception of Fernow Experimental Forest, where losses to streams were as high as 27% (table 1). Most of the losses occur as NO_3-N, extending over a protracted period, although in some cases immediate losses of urea-N or NH_3-N accounted for as much as 50% of the loss of applied N (Moore, 1975; Perrin et al., 1984)

Reasons for differences among studies in losses of applied N and concentrations of NO_3-N in streams vary, but often appear to be related to land-management history, nitrogen status of soils, or differences in the fertilizer applications (Bisson et al., 1992). In a study originally intended to investigate the importance of repeated fertilizations, Bisson (1982) evaluated N-runoff to streams in six fertilized watersheds, four of which had been "heavily" fertilized (rates not reported) less than 3 years prior to the study. Each stream had NO_3-N concentrations prior to the study fertilization higher than might be expected (100–1,200 µg/L) for forested streams in the Cascades, and even greater NO_3-N concentrations after fertilization. "Extensive" prior fertilization was considered the cause of high N-export (Bisson, 1982). One cause of this increase may be that fertilization stimulates growth of nitrifying bacteria in soils such that nitrification is increased if fertilizations are repeated within a few years (Bisson, 1982; Miegroet et al., 1990; Johnson, 1992). Enhancement of soil nitrification can also occur simply from large increases in atmospheric N-deposition (Fenn et al., 1998). Others have noted long-term increases in soil-N availability from single or multiple fertilizations (Binkley and Reid, 1985; Prescott et al., 1995; Norton et al., 1994; Moldan and Wright, 1998) resulting from increases in N recycling, nitrification, or mineralization. In an experiment at a midwestern agricultural field, using isotopically enriched N, Wilkison et. al (2000) found applied fertilizer N in runoff for several years following fertilization. Additional fertilizations in succeeding years would be expected to increase leaching of N to ground water.

From these studies it is apparent that a variety of factors related to the nitrogen status of soils determine a watershed's response to urea fertilization, just as many factors are considered in predicting the response of tree growth from fertilization (Klinka, 1991; Carter, 1992). For watersheds, these factors include not just the N-pool but the amount of organic material, nitrification potential, cation exchange capacity (Mitchell et al., 1996; Edwards et al., 1991), tree types and stand ages, N-deposition rates, previous fertilization history, precipitation quantity and timing, and others. In watersheds in the eastern USA, with high N-deposition rates and N-saturated soils (Aber et al., 1989; Stoddard, 1994; Fenn et al., 1998), it is not surprising that N-fertilization resulted in stream NO_3-N concentrations that nearly violated water-quality standards. Such results are much less likely in the western Cascades, where N-limitation in forest soils, and probable uptake in the largely N-limited forest streams (see below), help maintain ambient stream NO_3-N concentrations well below existing nutrient standards.

Water-Quality Criteria

Comparison of stream nutrient concentrations resulting from forest fertilization with currently available criteria is arguably insufficient to evaluate fertilizations effects, particularly in streams in Cascade streams of the Pacific Northwest, where primary production is often limited by the supply of nitrogen in water (Triska et al., 1983; Gregory et al, 1987; Bothwell, 1992; Borchardt, 1996). It is now well established that NO_3-N concentrations resulting from forest fertilization rarely exceed the U.S. Environmental Protection Agency's (EPA's) 10 mg/L (milligram per liter) standard, and ammonia toxicity has rarely been observed (Bisson et al., 1992; Binkley et al., 1999), despite sometimes high concentrations (Göthe et al., 1993). However, the NO_3-N standard is targeted towards human health protection in drinking water and is not intended to protect against ecosystem degradation. Furthermore, there are no criteria for phosphorus in streams, despite evidence that phosphorus concentrations exceeding 25 µg/L in lakes can produce eutrophic conditions (Welch, 1992), and in streams similar concentrations may result in nuisance growth of periphyton (attached algae) (Dodds et al, 1997, 1998; Correll, 1998).

The EPA has recently developed guidelines for the States to use in setting regional nutrient criteria (U.S. Environmental Protection Agency, 2000a). The nutrient levels suggested to prevent nuisance conditions, however, are considerably lower than the existing criteria targeting drinking water and ammonia toxicity. For instance, upper limits for total nitrogen (TN), dissolved inorganic nitrogen (largely NO_3-N and NH_3-N), and total phosphorus could be approximately 650, 400, and 38 µg/L, respectively (U.S. Environmental Protection Agency, 2000a). For the Cascades subecoregion, the suggested values are even lower—approximately 55, 5, and 9 µg/L, respectively [U.S. Environmental Protection Agency, 2000b]. These values are one to several orders of magnitude less than existing criteria and well below concentrations frequently observed following fertilization. However, few criteria have actually been set, and it will be left to States to do so, meaning that the establishment of criteria is likely to happen several years in the future, and will be variable among States and ecoregions. Thus, rather than using drinking-water criteria as a basis for decision making for forest streams, it might be more relevant at this point to increase attention on the ecological consequences of fertilization.

Ecological Effects on Streams from Forest Fertilizations

Many researchers have expressed the need for more direct investigation of fertilizer effects on stream ecology (Groman, 1972; Bisson, 1988; Binkley et al., 1999), though few studies (Groman, 1972; Meehan et al., 1975; Stay et al., 1979; Göthe et al, 1993) actually examined any biological responses. These studies indicated that toxic effects to aquatic invertebrates and fish are unlikely. However, broad conclusions about the ecological effects of N-fertilization are tenuous because (1) conditions specific to individual studies may not be applicable in other settings, (2) sampling techniques may have been insensitive to the questions being asked, or (3) the understanding of hydrological and biological processes available at the times of the studies were limited and precluded examination of subtle, yet potentially important ecological processes. In some cases, changes in nutrient inputs were measured in ephemeral streams too small to support substantial increases in algal biomass, and (or) algal growth might have been limited more by light availability than nutrient concentrations. In these cases, added nutrients would be transported downstream to larger, less light-limited reaches, where complex processes of nutrient uptake by benthic organisms (periphyton or bacteria) and hyporheic processing might greatly reduce water column concentrations. Some streams, particularly those like the Little River in the western Cascades (fig. 1), might be especially susceptible to negative effects because of local physical and water-quality conditions or combined effects from other upstream land use. Results from the few studies with biological components are discussed below with respect to their applicability to the Little River watershed.

Following spring fertilization of two previously clearcut watersheds (total 1,500 acres) in Alaska, Meehan et al. (1975) found no increase in periphyton biomass on Plexiglas slides, and no difference from natural variation in stream invertebrates despite five- to tenfold increases in stream NO_3-N concentrations (table 1). Although these results indicate minimal biological effects from fertilization, biomass of periphyton growth alone may not adequately represent changes in algal community due to high variability and complicating effects from invertebrate grazing, scour, or changes in taxonomic composition. (Stevenson, 1996a). The authors did not report data on algal special composition, which can indicate differences in water quality (Lowe and Pan, 1996), so there is no way to assess whether changes in nutrient concentrations changed the algal communities. Also, artificial substrates such as Plexiglas slides are often poor indicators of natural periphyton because they tend to underestimate growth of green and blue-green algae (Cattaneo and Amireault, 1992), which are common in many Cascade streams. Neither was the potential for nutrient or light limitation reported. The first- and second-order study streams had very low flow (streams in one subbasin were dry during summer), so primary production in these ephemeral streams was likely insignificant anyway. Furthermore, no information was provided on downstream sites. Finally, fertilization took place in spring, to clearcuts, as opposed to fall applications to 10-40 year-old stands, the current operational practice. As a result, nutrient dynamics in the forest floor and hydrologic conditions that contribute to nutrient runoff likely differed from those in locations like the Little River watershed.

After fertilization in the Cascade foothills of the Santiam River Basin, Oregon, Stay et al. (1979) found colimitation of Crabtree Creek water by N and P, using algal bioassays with the planktonic green algae *Selenastrum capricornutum*. There was no response to added N alone (table 1). Although valuable, laboratory demonstrations of colimitation in flasks by *S. capricornutum,* a planktonic green alga of European origin that lives in still or slow moving water, have limited transferability to Cascade streams dominated by periphyton. Reasons for this lack of transferability include the enhancing effect of moderate water velocities on periphyton metabolism (Stevenson, 1996b) and algal-grazer interactions (Steinman, 1996). In the same study, variation in populations of benthic invertebrates before and after treatment were indistinguishable due to high natural variability, and fish assays showed no mortality due to fertilization. The invertebrate studies, however, were designed to detect short-term changes from toxicity and could have been confounded by flow change. However, the study's approach would have been insensitive to longer-term shifts in invertebrate assemblages. Finally, calculations of invertebrate diversity indicated already perturbed conditions prior to treatment, suggesting that sensitive invertebrate species may have already been eliminated, and remaining species may have been insensitive to additional effects from fertilization.

Göthe et al. (1993), after fertilizing parts of two watersheds to evaluate stream acidification, found a temporary increase in drift of benthic invertebrates that was attributed to high concentrations of un-ionized ammonia. Drift was highest at the downstream stations, indicating cumulative upstream effects, and daytime drift was as large as nighttime drift, which may have rendered invertebrates more susceptible to daytime predation. No effects on fish density or mortality were noted. Though this study suggests some possible short-term, toxicity-based effects for invertebrates, it did not address longer-term changes that could result from more indirect changes in habitat and food quality due to altered nutrient regimes. The authors also did not evaluate primary producers, which would have been directly affected by nutrient additions from fertilization.

Some investigators acknowledge that increases in stream primary production from fertilization may occur (Binkley et al., 1999), occasionally postulating that such increases could stimulate food web changes that enhance fish production (Fredriksen et al., 1975; Malueg et al., 1972; Harriman, 1978; Hetherington, 1985; Bisson et al., 1992). In parts of an oligotrophic lake in British Columbia, nitrogen input from upstream forest fertilization with urea was sufficient to cause a temporary shift from nitrogen limitation to phosphorus limitation and an increase in plankton biomass (Perrin et al. 1984). Thus, the potential for ecological modification has long been recognized, and there is some evidence of its having occurred in certain instances. No investigators, however, have followed the movement of applied fertilizer nitrogen from the forest floor, through soil profiles and ground-water regions, to streams and into aquatic food webs, so the link between terrestrial and aquatic processes remains poorly understood (Binkley et al., 1999).

Stream Nutrient Dynamics

The processing of nutrients in streams is complex, and mostly beyond the scope of this review. However, advances in the understanding of ecological processes and several recent conceptual developments are critical to understanding the possible effects of fertilization on streams, and potentially to investigating those effects in streams of the Cascade Mountains. These concepts include nutrient speciation and limitation, nutrient uptake by stream algae, hyporheic processing of nutrients, algal indicators of environmental changes, food web interactions between primary producers and higher trophic levels, and the use of stable isotopes as tracers of nutrient transfer in aquatic food webs.

Stream nitrogen budgets, though complex to measure, have been determined in several forested streams. In the Pacific Northwest, almost all have been in the Andrews Experimental Forest, in the Cascade Range of Oregon. In all cases, organic nitrogen has been an important component of both N-input and output in undisturbed streams. Sollins et al. (1980) found that nitrogen was biologically limiting in an old-growth watershed at the Andrews Experimental Forest, and that DON or NH_3-N were the dominant forms of N in solution annually. In the same watershed, Triska et al. (1984) found that over 96% of the nitrogen leaving the outlet stream on an annual basis was as organic nitrogen, with 77% as DON, whereas less than 4% was as NO_3-N. Similarly, DON was the largest component and NO_3-N the smallest (95% and 0.2%, respectively) of annual N-export in temperate, old-growth forests in Chile (Hedin et al., 1995). Overall, the relative amount of organic nitrogen (dissolved and particulate) output from subbasins in the Andrews Experimental Forest ranges from 28% to 85%, with the highest DON proportions occurring in fall (K. Vanderbilt, Oregon State University, unpub. data, 1999). Interestingly, NH_3-N export is roughly twice that of NO_3-N in undisturbed watersheds in the Andrews Experimental Forest, whereas NO_3-N export is greater in disturbed (though unfertilized) watersheds (K. Vanderbilt, Oregon State University, unpub. data, 1999). Wondzell and Swanson (1996) found that a conifer-dominated floodplain was the largest source of nitrogen to fourth-order McRae Creek, and more than 50% of that nitrogen was as DON. In contrast, inorganic nitrogen made up over 50% of the nitrogen entering McRae Creek through an alder-dominated gravel bar, with most of the remainder being as DON.

One importance of N-speciation is related to the types of algae able to utilize that nitrogen, and the possible shifts in community structure that might occur if the amount or form of nitrogen changes. For instance, DON is usually assumed to be biologically unavailable (Sollins and McCorison, 1981). However, some diatom species are able to utilize DON for energy (heterotrophy) and (or) for nutrition, although this process is relatively inefficient because energy is expended in breaking down organic molecules to liberate energy and reduce N (Hellebust and Lewin, 1977). Inorganic nitrogen is more easily assimilated than DON by most algae, particularly green algae. NH_3-N assimilation is the most energetically favorable but NO_3-N can readily be reduced as well. Where available nitrogen is scarce, algae that can fix elemental nitrogen (N_2) from air or water have a competitive advantage and are often more abundant than in nitrogen-replete waters (Biggs et al., 1998). However, the importance of DON as a source of N to algae also may be elevated in these situations (Mulholland, 1992). N-fixing algae are typically blue-green species but can also include certain diatom species containing cyanobacterial inclusions (Floener and Bothe, 1980). In the North Umpqua and Little River watersheds, the colonial blue-green alga, *Nostoc*, and the diatom *Epithemia Sorex*, are commonly observed in nitrogen-poor environments (Anderson and Carpenter, 1998).

Hence, it is reasonable to hypothesize that a fertilization-induced spring/summer shift of the predominant dissolved nitrogen form, to NO_3-N from DON or NH_3-N, could induce a shift in algal community structure. The algal community might change from heterotrophic and N-fixing species to a community dominated by non-N-fixing diatoms, non-heterocystous blue-greens, and possibly filamentous green algae. Such shifts in community were observed after NH_4^+-N addition to a nitrogen limited, fifth-order stream (Lookout Creek) in the Andrews Experimental Forest, despite relatively unchanged algal biomass in the enriched sections (Lundberg, 1996). In that study, the changes in algal assemblages due to N enrichment were hypothesized to affect invertebrate grazers because certain species were known to prefer the epithemiacean (N-fixing) diatoms present prior to enrichment. Recent efforts to model benthic algal and invertebrate processes indicate that an increase in a limiting nutrient that causes a decrease in algal food quality can indirectly affect various invertebrate functional groups (McIntire et al., 1996).

A decrease in N concentrations in surface water during summer does not necessarily confirm that fertilizer N is not entering streams, nor does it confirm that benthic communities are unaffected. Concentrations of stream NO_3-N could be reduced to low levels during summer because of increased plant uptake in upslope areas. Yet periphyton uptake can also increase nitrogen retention in streams (Kim et al., 1992; Peterson et al., 2001), and hence reduce dissolved-nitrogen concentrations. Thus, if dissolved-N input during summer is through the riverbed as ground water is important, subsequent algal uptake and spiraling (Newbold et al., 1981) is likely to reduce the ability to measure the added nitrogen in the water column (Peterson et al., 2001), even though it may be transported downstream by recycling in benthic layers or as sloughed algae. Additionally, added N in a hyporheic zone could increase heterotrophic metabolism (Mullholand et al., 1997, 1999; Storey et al., 1999) while obscuring the increased N input in streams.

Hyporheic Processing

In recent years, hydrologic exchange and nutrient processing in hyporheic zones (subsurface and riparian near-shore environments) along streams have received increased attention. Extensive syntheses of various hyporheic processes and findings have been published by Cirmo and McDonnell (1997), Boulton et al. (1998), Dahm et al. (1998), and Storey et al. (1999). Hyporheic function can be critical in determining hydrologic flow paths and nutrient exchange, as well as transformations of carbon and nitrogen. These aspects are discussed briefly below with respect to forest fertilization and its effects on aquatic systems.

Interactions of ground water with streams can follow several types of paths, including gaining, losing, parallel flow, and through flow (Woessner, 2000), each of which may be intermittently present depending on stratigraphic and fluvial character of individual reaches or streams (Stanford and Ward, 1993). These flow paths have potentially diverse implications for ecological processes. In gaining streams the inflowing water is derived from shallow to regional ground-water flow and will likely increase net stream solute transport. Benthic processes will be partly dependent on the quality of incoming hyporheic water, which in turn will be dependent on physical conditions in stream margins and on the quality of the ground-water source. In losing streams, benthic processes will be more predictably dependent on surface-water chemistry and physical conditions, and will in turn control hyporheic metabolic process (Boulton et. al, 1998). Parallel flow occurs when head gradients between surface and ground water are not distinct, with intermittent exchange occurring in both directions between stream and hyporheic zones depending on local variations in channel gradient and bed material. In these cases, water in hyporheic zones and streams may be of similar quality, and net solute transport may be relatively unaffected by the exchange processes. Through flow is likely to occur between bends in alluvial-stream reaches as a short-circuiting of ground-water flow from upgradient to downgradient.

Transformations of solutes in hyporheic zones are often controlled by the hydraulic residence time and the amount of organic material in those zones (Boulton et. al, 1998; Wondzell and Swanson, 1996). Transformations of nitrogen are complex, depending also on redox conditions and available oxygen and organic carbon (Cirmo and McDonnell, 1997), and are seasonally variable (Wondzell and Swanson, 1996) (fig 2). During winter and high-flow periods, temperatures and storage time are typically low, so nitrification of incoming DON (including urea or NH_3-N) from upland areas is likely to be minimized. Nitrogen entering the stream will be a combination of DON, NH_3-N, and NO_3-N (possibly nitrified in upland terrestrial soil). As temperatures rise and streamflows decrease during spring and summer, nitrification will increase such that the predominant form of N entering the stream will be nitrate. Along with these warm weather transformations will be losses of nitrogen resulting from microbial uptake (Boulton et. al, 1998; Wondzell and Swanson, 1996) and possibly denitrification (Cirmo and McDonnell, 1997), so the overall load of nitrogen entering the stream will be reduced compared to that in winter and early spring. If the hyporheic zone contains relatively high amounts of DOC, microbial metabolism and hence nitrification rates may be further enhanced, and denitrification can be enhanced in anoxic environments.

The form, timing, and relative amount of fertilizer-derived nitrogen entering streams will therefore depend partly on hyporheic zone processing, which itself will depend on subsurface flow paths and degree of saturation in upland and riparian areas, as well as hydraulic conductivity, stratigraphy, and amount of organic material in riparian areas. In regions as geologically complex as the Little River watershed and the Wolf Creek subwatershed, heterogeneity may reduce the ability to make broad spatial generalizations. Nonetheless, if summer streamflow has an important ground-water component, fertilizer nitrogen could stimulate microbial nitrification and (or) denitrification in hyporheic zones and benthic algal growth at the interface between stream and hyporheic inputs.

Conceptual Model of Ecological Processing of Fertilizer Nitrogen

There are a variety of possible ecological effects of fertilizer nitrogen in Cascade streams, depending on local conditions of hydrology, water quality, stream morphology, and aquatic biota. For the Little River watershed, a hypothetical process scenario following fall-winter urea fertilization is illustrated in figure 3, p. 18–19. The lettered descriptions below correspond to the respective lettered parts of the figure. Pie charts indicate hypothetical relative stream nutrient concentrations (areas of circles) and speciation (pie slices).

A. Following fall fertilization, rains cause immediate overland and (or) subsurface runoff of organic and ammonium nitrogen to small streams draining fertilized stands, most of which is rapidly transported to larger streams (>4th order) such as Little River, and further downstream to successively larger rivers. Streams draining reference areas remain higher in organic nitrogen than NO_3-N (Sollins and McCorrison, 1981), and have lower nitrogen concentrations overall.

B. Winter rains and nitrification of applied nitrogen enhance longer-term inputs of NO_3-N to small streams, which also is mostly transported out of the fertilized stands downstream to the Little River and farther. Some sequestration of nutrients during the spring may occur in stream biota, other organic material, or in hyporheic zones and shallow ground water.

C. During late spring and early summer, nitrate concentrations in streams affected by, and downstream of, fertilization decline in conjunction with declines in stream discharge. Reasons for these reductions include uptake by plants and trees in forest soils, denitrification or sequestration in hyporheic environments, and uptake by periphyton in streams (Mulholland, 1992; Mulholland and Rosemond, 1992). Dissolved organic nitrogen (DON) remains proportionally high (Sollins and McCorrison, 1981; Triska et. al, 1984), especially in unfertilized basins.

D. Additional input of fertilizer nitrogen during summer is reduced but may not be altogether eliminated. Shallow ground-water flow from upslope may transport NO_3-N or DON to periphyton or benthic bacteria through riparian and hyporheic zones. Nutrients could be transported from smaller streams in fertilized stands, where algal growth may be light limited, to higher order reaches where more open canopies allow greater algal growth (Gregory et al, 1987). Regional ground-water flow patterns (Harvey and Bencala, 1993) might discharge water and nutrients from fertilized areas well downstream of fertilized stands.

E. Algal biomass in small, lower-order streams remains low during summer because of light limitation, though changes in species may occur if nitrogen input is increased. Biomass downstream may increase, or algal species may change, as light availability increases. Benthic uptake continues to keep water column nitrogen concentrations low. Nuisance algal growth occurs in some places where substrate and light conditions are favorable, enhancing overall primary production.

The N-transport and transformation processes postulated here would not be identical in all forests, even within the Pacific Northwest. Some factors might make certain streams more sensitive than others to nitrogen inputs, predisposing them to perturbation. For instance, coastal mountain regions of the Pacific Northwest often have higher ambient nitrate concentrations than Cascade streams, possibly because of differences in geology, weather patterns, and the predominance of red alder (*Alnus rubra*) (Brown et al., 1973; Miegroet and Cole, 1988). In contrast, relatively high concentrations of available phosphorus, often found in streams of young volcanic origin (Dillon and Kirchner, 1975) such as the Cascades, provide adequate phosphorus (10–40 µg/L) necessary to grow periphyton (Bothwell, 1988; Borchardt, 1996), making periphyton communities more dependent on nitrogen supplies. In low alkalinity streams, a given algal productivity or biomass could cause larger fluctuations in pH and higher daily maximum pH than in streams with high alkalinity (Teal and Kanwisher, 1966; Beyers, 1970). Also, more stable beds composed of relatively large, stable substrates (ranging from cobbles to bedrock) can often accumulate higher algal biomass, because the algal mats are able to withstand higher flows that would scour attached algae in streams with less stable surfaces (Peterson, 1996).

Enhanced diel fluctuations of pH and DO resulting from increased primary production may be compounded, in streams dominated by bedrock or otherwise armored, if hydrologic exchange or discharge through hyporheic zones also is reduced. Because these zones are important areas for nutrient transformations and heterotrophic activity in streams (Mulholland et al., 1997 & 1999; Naegeli and Uehlinger, 1997; Boulton et al., 1998; Chafiq et al., 1999), heterotrophic respiration may be reduced in streams with smaller hyporheic zones (Mulholland et al., 1997 & 1999), potentially allowing higher pH maxima. Although maximum DO concentrations could be increased if hyporheic respiration is reduced, DO in small, high-gradient streams appears to be reaerated more rapidly by physical processes than does carbon dioxide (Guasch et al., 1998), so stream DO might not be altered to the same extent as pH. In a study of the North Umpqua River, DO was apparently controlled by temperature and reaeration, despite diel pH changes that were indicative of primary production (Anderson and Carpenter, 1998).

The influences of upstream land-management practices could play a vital role in determining stream response to N-application. Nutrient concentrations (including NO_3-N) are known to be elevated by many forestry operations (Tamm et al., 1974; Sollins and McCorison, 1981; Tiedemann et al., 1988; Adams and Stack, 1989; Binkley and Brown, 1993a, 1993b), with various subsequent effects on stream productivity (Gregory et al., 1987). Other site-dependent factors that could influence algal production include soil characteristics, ground-water flow paths, light availability, water temperatures, and the amount of invertebrate grazing.

Suggested Approaches to Evaluate Ecological Effects of Forest Fertilization

Care will be needed to differentiate the effects on streams of fertilization from those of other land-management practices and from natural variability. The susceptibility of streams to eutrophication and other ecological effects will depend on numerous watershed conditions, including hydrology, geology and stream morphology, geochemistry, nitrogen status of soils and streams, canopy cover and aspect, and history of land use. Many forested watersheds in the West have already been perturbed by historical resource management. As a result, added fertilizer N could contribute to cumulative stimulatory effects of forestry on primary production in streams (Norris et al., 1991). Forest management practices that can increase nutrient runoff to streams include road construction and logging (Fredriksen et al., 1975; Sollins and McCorison, 1981; Binkley and Brown, 1993a, 1993b; Brown and Binkley, 1994), history of fire or fire prevention (Brown et al., 1973; Tiedemann et al., 1978; Norris et al., 1991) and fertilization history (Bisson et al., 1992). Additional conditions that could compound the effects of fertilizer nutrient inputs include (1) increased sediment transport, which can contribute nutrients and scour channels, (2) increased temperature, which can enhance algal growth (DeNicola, 1996) and accelerate invertebrate hatches, possibly decreasing grazing pressure on algae, and (3) increased light penetration, where small buffer strips might cause a shift from light to nutrient limitation.

The occurrence of several or all of the above conditions can contribute to possible cumulative effects on streams and make differentiation of the effects from an individual practice difficult to discern. Thus, streams already experiencing increases in eutrophication, manifested as increases in nutrient concentrations, changes in algal growth or algal populations, elevated temperatures, increased light penetration, or exacerbation of diel DO and pH cycles resulting from elevated primary production, may be poor locations to examine effects from forest fertilization.

Despite these complications, there are several ways to determine, more definitively than in the past, whether fertilizer nitrogen is transported to streams during growing seasons or contributes to benthic production. Potential methods include (1) measurements of nutrient processing and transport in hyporheic zones, (2) assessments of benthic metabolism either in chambers (Bott et al., 1997; Harvey et al., 1998) or by measurements of whole-stream metabolism (Marzolf et al., 1994, 1998;

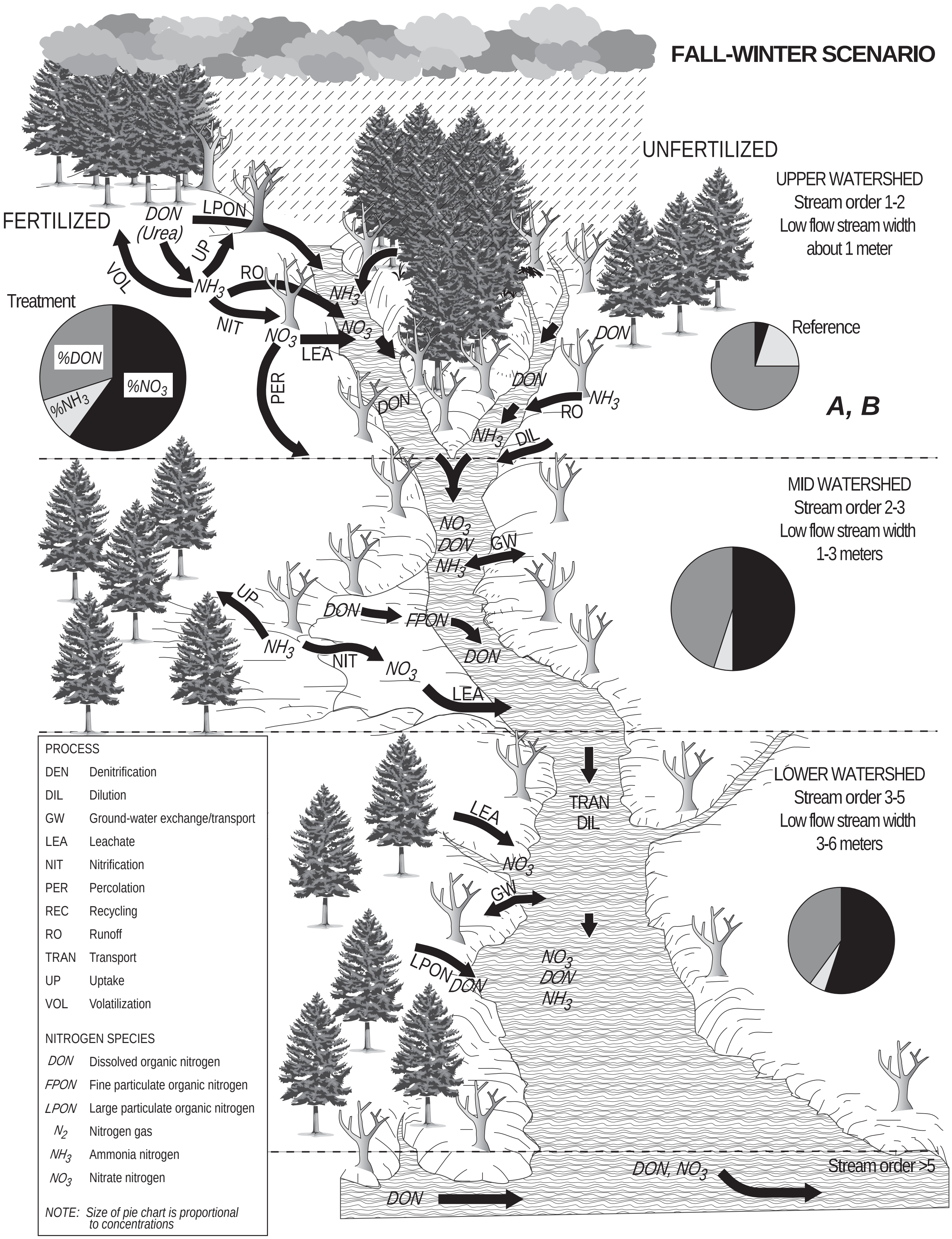
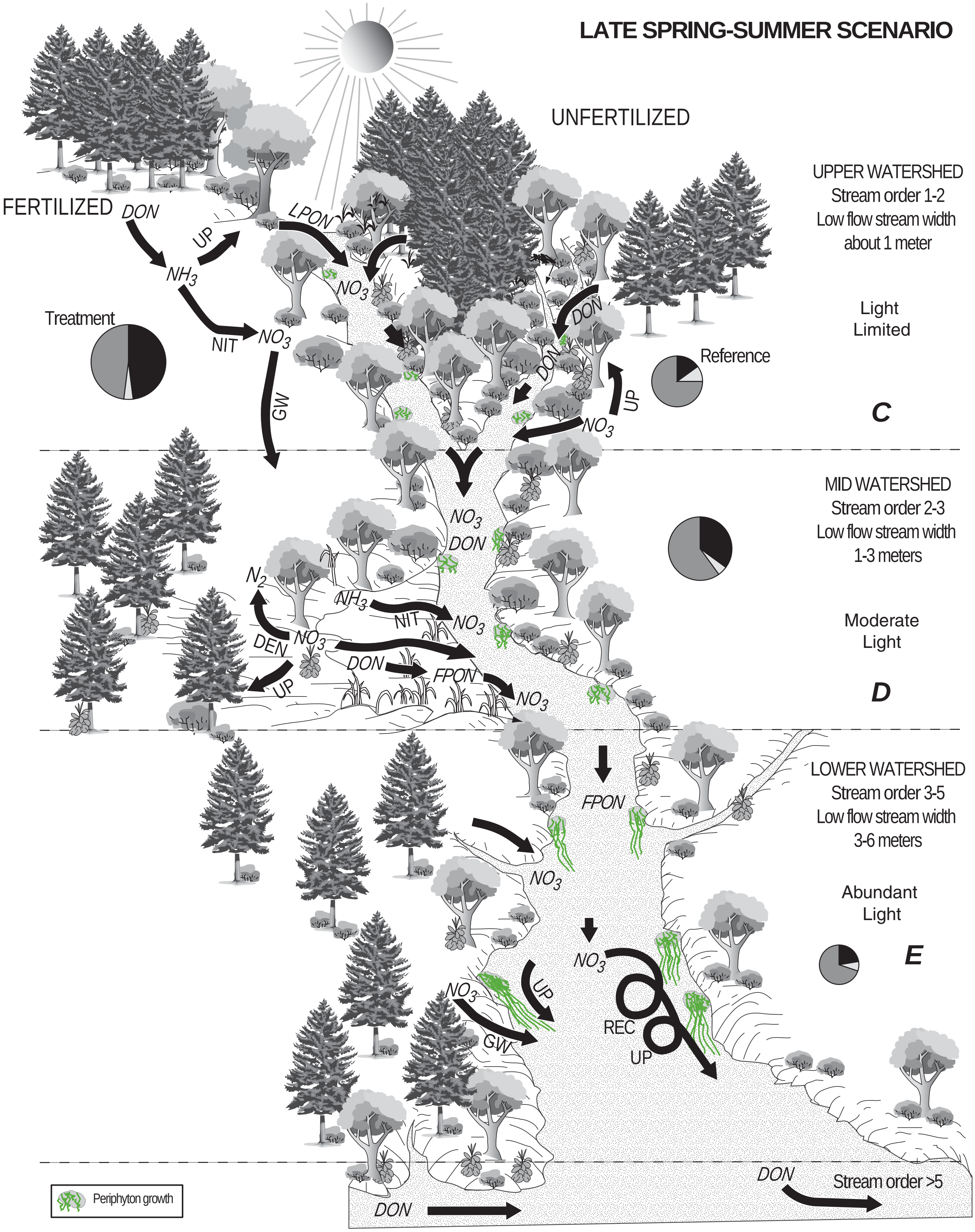


Figure 3. Hypothetical transport pathways, dominant processes, and relative concentrations of nitrogen in response to urea explained in the text.)



fertilization in forested catchments of the western Cascades, Oregon, during fall/winter and late spring/summer. (Parts A-E are

Parkhill and Gulliver, 1998), (3) experiments using nutrient-diffusing substrates to determine growth-limiting factors (Fairchild et al., 1985; Pringle and Triska, 1996; Tank and Webster, 1998), (4) measurements of cellular-nutrient content and sloughing algal material in transport to more accurately account for N inputs and outputs in streams, (5) use of stable isotopes of nitrogen, including possible additions of labelled ^{15}N in urea fertilizer (Kahl et al., 1993), to track incorporation of applied N into different algal and food-web compartments (Harvey et al., 1998); and (6) assessments of benthic algal communities using autecological evaluations of water quality (Lowe and Pan, 1996; Pan et al., 1996; Anderson and Carpenter, 1998) and (or) biomass in cumulative assessments longitudinally and temporally in stream systems. In addition, in sensitive systems, continuous monitoring of DO and pH during selected periods might provide data to determine whether production is increasing and whether that production is having potentially deleterious effects on stream ecosystems. Most of these methods will require similar evaluations in control or untreated streams for comparison to determine the magnitude of response to fertilizer treatment.

LITTLE RIVER WATERSHED

This section presents a framework for evaluating an operational application of urea fertilizer by the Bureau of Land Management (BLM) to selected stands in the Little River watershed, a nitrogen-limited tributary to the North Umpqua River, Oregon (fig. 4). The previous literature review is used as a basis to explore possible ecological effects of fertilization in the context of known geological and hydrological conditions in the basin. Urea-based fertilizer would be applied during late fall to individual 15–40 year old stands on BLM lands, at a rate of 200 pounds N/acre (224 kg N/ha). The study would examine water-quality and possible ecological effects of added fertilizer nitrogen, including changes in biomass of periphytic algae, dissolved oxygen (DO) and pH in streams, shifts in algal species and community composition, and changes in secondary grazer (macroinvertebrate) communities or food-web structure and function resulting from shifts in algal community composition and biomass. Data from reconnaissance samplings are included to provide an indication of water-quality and algal conditions prior to fertilization. Various portions of private timberlands in the watershed were fertilized in 1998 or in 1999, but were not sampled prior to fertilization.

Physiographic Setting

The Little River watershed (fig. 1) is approximately 206 square miles in area, with elevations ranging from 730 to 5,275 feet above sea level (U.S. Forest Service and Bureau of Land Management, 1995). Most of the watershed (83%), including almost all of the land east of Cavitt Creek, is located within the western Cascades geologic province (McFarland, 1983). Soils in this province tend to be high in nutrient content, particularly phosphorus, as they are derived from many layers of volcanic rock. In many places, streams have eroded deeply incised channels through the volcanic layers to surficial bedrock, and the potential for landslides is typically high on the steep slopes. Recharge and well yields in the province are typically low. In some places, including the eastern Wolf Creek subwatershed, earthflows have created a poorly defined drainage network where ponds and seeps are common, and where subsurface flow can exceed surface flow (U.S. Forest Service and Bureau of Land Management, 1995). Rocks of the Klamath Mountains comprise about 11% of the watershed, mostly in the southwest. Exposed rocks in this province include granitic and sedimentary rocks (composed of altered submarine volcanic flows, tuffs, flow-breccia, and agglomerates) and ultramafic rocks (including large outcroppings of peridotite and serpentinite). Well yields in this province tend to be somewhat higher than in the western Cascades (McFarland, 1983). The final 6% of the watershed, located in the extreme northwestern corner near the mouth of Little River, is made up of tertiary rocks from the Coast Range province, which are predominantly marine sedimentary rocks.

The Wolf Creek subwatershed is located in the central portion of the watershed in what is known as the “Wolf Plateau Vicinity” (U.S. Forest Service and Bureau of Land Management, 1995), which also includes Negro Creek and White Creek. The vicinity is characterized by broad, gently sloping uplands formed from resistant ash-flow tuff, creating a rocky bluff along the northern, western, and

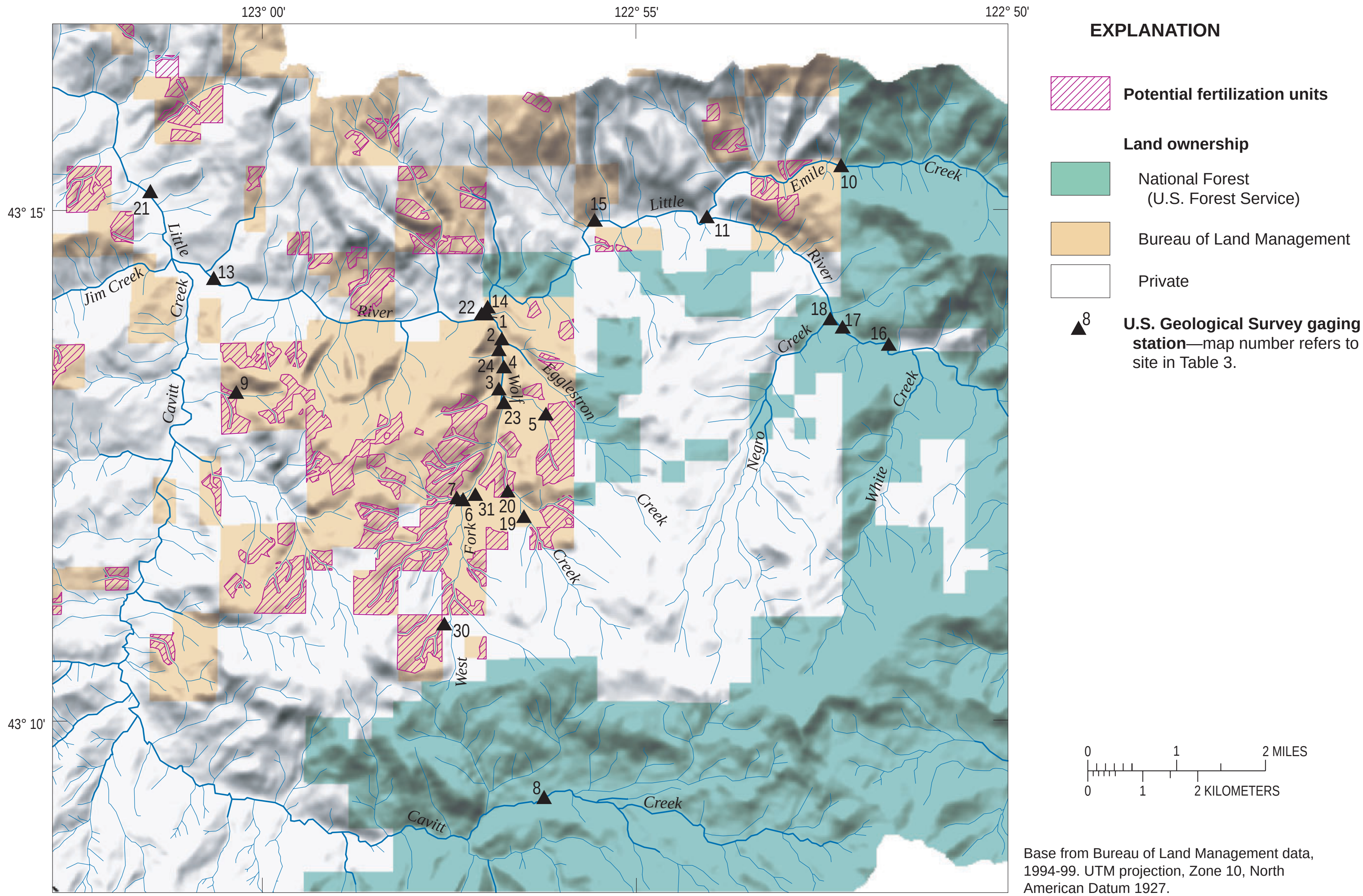


Figure 4. Sampling locations in Little River Watershed, Oregon 1998.

southern edges. The plateau is dissected by drainage systems that have begun to cut through this resistant layer, creating steep gradients with numerous waterfalls that act as natural barriers to fish passage. The plateau is capped by a variety of volcanic deposits such as lava flows, mudflows, flow-breccias, and tuffs. Weathering of these deposits has generated fine, clay-rich, relatively impermeable soils. Mass-failure processes are likely where channel incision has undermined adjacent banks, and during seasonal peak flows chronic sediment is delivered from these channels. Localized debris avalanches, slumps, and earthflows are found along steeper hillslopes (U.S. Forest Service and Bureau of Land Management, 1995).

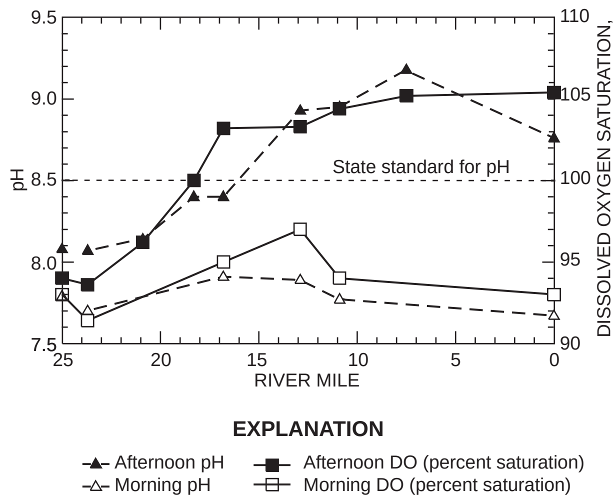


Figure 5. Morning and afternoon pH and dissolved oxygen saturation in the Little River, July 28, 1998. (Source: Powell, 1998.)

Water-Quality Issues in the Little River Watershed

Previous studies in the Umpqua Basin have indicated that several streams, including the South Umpqua River (Tanner and Anderson, 1996), North Umpqua River (Anderson and Carpenter, 1998), and Little River (Powell, 1995, 1998) experience nuisance growths of periphytic algae during summer low flow periods. In many locations, resultant photosynthesis has raised pH values higher than the State of Oregon standard of 8.5, with maximum values in the North Umpqua Basin reaching as high as 9.1 (Anderson and Carpenter, 1998; Powell, 1995, 1998). Little River is listed on the State of Oregon’s 303(d) list of water-quality limited streams for pH, temperature, sedimentation, and habitat modification (Oregon Department of Environmental Quality, 1999). Longitudinal surveys in various streams in the basin have shown general increases in pH in a downstream direction, and there are substantial diel variations in pH and DO (fig. 5) characteristic of excessive primary production. Mechanisms that may account for nuisance algal growth and (or) exaggerated diel pH and DO cycles include increased primary production resulting from timber operations (Gregory et al., 1987; Powell, 1995) and lack of functioning hyporheic zones that could help to reduce pH through benthic respiration (Storey et al., 1999; Mullholand et al., 1997, 1999). These effects are likely exacerbated when streams have low buffering capacity (alkalinity), and geochemical processes may be important locally in controlling stream pH.

Problems with water-quality and ecosystem processes in the Little River watershed extend beyond exceedances of State water-quality standards. There is a variety of aquatic species of concern in the Little River (table 2), including both anadromous fish and amphibians (John Raby, Bureau of Land Management, written commun., August 1999; Dr. R. Bruce Bury, U.S. Geological Survey, written commun., August 1999). These species could be affected by disruption of food supplies resulting from changed inputs of nutrients. To the extent that management activities on forest lands in the basin affect these sensitive species, public land agencies are mandated to strive to balance those activities with the needs of aquatic biota. Thus, it is important to understand the effects of forest management, including fertilization, on aquatic communities in the Little River watershed and in similar basins throughout the Pacific Northwest.

Table 2. Aquatic species of concern in the Little River watershed, Oregon

Species	Federal designation	State of Oregon designation
Umpqua River cutthroat trout	Endangered	Sensitive/Vulnerable
Oregon Coast coho salmon	Threatened	Sensitive/Critical
Oregon Coast steelhead trout	Candidate	Sensitive/Vulnerable
Pacific lamprey	Species of Concern	Sensitive/Vulnerable
Red legged frog	Sensitive[1]	No Designation

[1]listed on the National Forest Sensitive Species List

Land uses in the Little River Watershed and Potential Effects on Water Quality

Forestry is by far the dominant land use in the Little River watershed. Roughly 60 percent of the watershed's area had been harvested for timber and reforested by 1995. The watershed's 206 square miles (approximately 132,000 acres) is predominantly (>63%) Federal forest, of which 76% (63,575 acres) is administered by the Forest Service and 24% (19,802 acres) is administered by the BLM. The watershed is sparsely settled (population approximately 1,200), and over 70% of the private lands are managed for commercial timber production. About 78% of the Wolf Plateau Vicinity has been harvested, making it the most intensively logged vicinity in the Little River watershed. There are 960 miles of road in the watershed, the majority of which are used for forest management. Extensive road building and timber harvesting, especially prior to 1970 when techniques were poor, may have contributed to the Wolf Plateau Vicinity's having the second highest frequency of land-management related landslides (5.2 per square mile) in the watershed (U.S. Forest Service and Bureau of Land Management, 1995). Fertilization continued on selected Forest Service lands until 1990, but no BLM lands have been fertilized in the watershed since 1975 (C. Kinntop, Bureau of Land Management, written commun., 2000). Timber operations were recently identified as being among potential nutrient sources in the larger North Umpqua River Basin (Anderson and Carpenter, 1998). In that study, nutrient enrichment, excessive benthic algal growth, and maximum pH were evaluated in conjunction with timber and hydropower operations.

In 1994, the public land in the watershed was collectively designated as one of 10 Adaptive Management Areas (AMAs) under the President's Northwest Forest Plan (U.S. Forest Service and Bureau of Land Management, 1994). The specific emphasis of the Little River Adaptive Management Area (LRAMA) is "the development and testing of approaches to integration of intensive timber production with restoration and maintenance of high quality riparian habitat" (U.S. Forest Service and Bureau of Land Management, 1995). As a result, evaluating management effects on water quality is the major emphasis of the LRAMA.

A complicating aspect to the proposed study is the patchwork pattern of land ownership in the watershed (fig. 4), a common feature in forested lands of the western United States. The upper portions of the Little River watershed are located entirely on National Forest lands, whereas private timberlands intermix in alternating square-mile sections with BLM and Forest Service lands in the middle portions of the LRAMA, particularly in the Wolf, Negro, and Cavitt Creek subwatersheds. This pattern renders many of the perennial streams in these subwatersheds subject to influences from a mixture of upstream land uses and management practices, including previous fertilizations on much of the private timberlands during fall in 1998 and 1999. For this reason, care will be needed to differentiate fertilization effects to perennial streams on BLM lands and effects from other land uses. This problem necessitates including smaller scale research (reach and subwatershed), in addition to investigating larger streams, to minimize confounding effects from upstream treatments.

In addition to forestry, there is a small amount of agriculture, a permanent population of about 1,200, 2 camps that house a combined population of 250–400 people at a time, and least 19 federally managed camping areas along the Little River (U.S. Forest Service and Bureau of Land Management, 1995). Of the two camps, one has a small on-site wastewater treatment plant (WWTP), the other a large septic drainfield. Private residences in the watershed predominantly use septic systems, and recreation areas use either pit toilets or septic systems, some or all of which could provide nutrients to the river depending on their state of maintenance. With summertime streamflows dipping to less than 30 ft^3/s, the cumulative nonpoint nutrient inputs from these sources can potentially contribute to stream eutrophication.

The two established camps, the Wolf Creek Job Corps Center and the Little River Christian Camp, are located just upstream from Wolf Creek at approximately river miles 12 and 13, respectively. The Job Corps Center houses between 230 and 275 people, and the Christian Camp averages about 90 during summer. The Job Corp Center's WWTP discharges an average of 21,000 gallons per day. Coincidentally, both camps are located immediately upstream of Wolf Creek, the tributary with the largest number of forested stands planned for fertilization by the BLM and the most likely location in which to investigate fertilizer effects on streams. This fact, and possible effects from other upstream land uses such as recreation and forestry operations, complicates the assessment of the effects to the Little River from fertilization within and outside the Wolf Creek subwatershed due to the lack of an appropriate upstream reference.

Water-Quality Conditions

Data were collected by the USGS and BLM during reconnaissance investigations in August and November 1998 (table 4), and August 1999 (table 5). The August surveys were to assess conditions during low flow prior to fertilization, and the November survey was intended to assess runoff during a fertilization to private timber land in the Wolf Creek subwatershed. Ultimately, the November samples were collected prior to fertilization during a storm. In the discussion below, emphasis is placed on data from the August 1999 survey because it is a more complete dataset and more quality assurance data were collected during that survey. Data from 1998 are referenced where they support or contradict findings from August 1999.

Samples from all locations (fig. 4, table 3) were analyzed at the U.S. Geological Survey's National Water Quality Laboratory (NWQL) in Denver, Colorado. A subset of samples from the August 1999 survey also was submitted for nutrient analyses to the Central Chemical Analytical Laboratory (CCAL) in the Forest Sciences Laboratory at Oregon State University. This laboratory comparison was an attempt to assure the quality of CCAL for low-level nutrient analyses, particularly nitrogen species, because the NWQL does not have methods available to analyze organic nitrogen at the low concentrations routinely found in the Little River watershed (tables 4 and 5). Organic nitrogen is a potentially important nitrogen species in the adjacent North Umpqua River Basin and typically constitutes the largest portion of nitrogen budgets in forested streams in the Cascades (Triska et al., 1984).

Methods

Discharge was measured using Price AA or pygmy current meters by established techniques (Rantz et al., 1982) wherever possible, although in some places water depths or velocities were too low to use meters. In these instances, discharge was measured by directing the entire flow of the stream into a bucket and measuring the volume of water filling the bucket in a known amount of time. Frequently this was done on the downstream ends of culverts under logging roads. Field parameters (temperature, specific conductance, pH, and dissolved oxygen) were measured in-place using Hydrolab® multiparameter instruments that were calibrated in the field according to manufacturer's specifications. Where possible, field measurements were timed to document pH near its daily maximum (about 4:00–6:00 p.m.), and sampling during 1999 specifically included late afternoon measurements at each site for this purpose.

Water chemistry samples for nutrients and major ions were taken by grab sampling in most cases. Grab sampling was considered representative where streamflows were low, there was little suspended material, and the streams were well mixed due to high gradients. Also, in many of the small catchments, the low streamflow would have prevented the use of larger depth- and width-integrating samplers. Samples from the main-stem Little River were collected using depth- and width-integrating techniques. Samples for whole-water (unfiltered) nutrients (total phosphorus [TP] and total organic-plus-ammonium nitrogen [TKN]) were collected in acid-washed bottles. Those to be analyzed by the NWQL were immediately preserved with 0.2N H_2SO_4, whereas those being analyzed by CCAL were unpreserved. For filtered-water nutrients (dissolved organic plus ammonium nitrogen [DKN], NH_3-N, nitrate plus nitrite nitrogen [predominantly NO_3-N], nitrite nitrogen, soluble reactive phosphorus [SRP], and digested dissolved phosphorus [TDP]), water was subsampled on site by peristaltic pump with in-line, prewashed, disposable capsule filters (0.45 µm [micrometer] pore-size) into sample bottles and stored chilled and unpreserved. Samples for major chemistry were filtered into two polypropylene bottles; water analyzed for cations was preserved with nitric acid, and water analyzed for anions was unpreserved. Samples for alkalinity during August 1999 were filtered in the field and stored chilled until titrated at the U.S. Geological Survey's Oregon District Laboratory (about 3–4 days). Based on good agreement in cation and anion balances, storage time did not appear to affect the alkalinity results. Analytical methods and detection levels for nutrient samples analyzed at the NWQL and CCAL are given in table 6.

Table 3. Sites sampled in the Little River watershed by the U.S. Geological Survey (USGS) in 1998 and 1999×

[ID, identification; BLM, Bureau of Land Management; USFS, U.S. Forest Service; Latitude/Longitude given in degrees, minutes, and seconds]

Map ID	Site name	USGS 7-1/2 minute topographic map	Range/ township/- section	Latitude/ longitude	Dates sampled		
					August 1998	November 1998	August 1999
1	Wolf Creek at mouth	Red Butte	R2W/T27S-9	431358/1225700	X	X	X
2	Egglestron Creek at mouth	Red Butte	R2W/T27S-16	431341/1225648	X	X	X
3	Unnamed tributary to Fork Wolf Creek at west bank near Wolf Creek Falls	Red Butte	R2W/T27S-16	431314/1225650	X	X	
4	Wolf Creek above Egglestron ×Creek	Red Butte	R2W/T27S-16	431327/1225646	X		X
5	Unnamed tributary to Wolf Creek on east bank upstream of Egglestron Creek on BLM 16.0 road	Red Butte	R2W/T27S-16	431253/1225619	X	X	
6	West Fork Wolf Creek	Red Butte	R2W/T27S-20	431209/1225719	X	X	X
7	Unnamed tributary to West Fork Wolf Creek at west bank on BLM 16.0 road	Red Butte	R2W/T27S-20	431210/1225724	X	X	X
8	Cavitt Creek above Withrow Creek on road 25	Red Butte	R2W/T28S-9	430914/1225614	X		
9	Unnamed tributary to Cavitt Creek below Evarts Creek	Lane Mountain	R3W/T27S-13	431312/1230021	X		
10	Emile Creek at USFS/BLM boundary	Mace Mountain	R1W/T26S-31	431525/1225215	X		
11	Emile Creek at mouth	Old Fairview	R2W/T27S -2	431455/1225403	X		
13	Little River above Cavitt Creek at new bridge	Lane Mountain	R3W/T27S-11	431419/1230039			X
14	Little River above Wolf Creek	Red Butte	R2W/T27S-9	431402/1225659	X	X	X
15	Little River above Job Corps at road 2701 bridge (Old Red Butte Road)	Red Butte	R2W/T27S-3	431453/1225533	X	X	X
16	Little River at White Creek Campground	Taft Mountain	R1W/T27S-7	431340/1225137	X		X
17	Little River at Coolwater Campground	Taft Mountain	R1W/T27S-7	431350/1225214		X	
18	Negro Creek near mouth	Taft Mountain	R2W/T27S-12	431355/1225224		X	X
19	Wolf Creek near Red Butte below gravel quarry at BLM 14.1 road	Red Butte	R2W/T27S-28	431159/1225630		X	
20	Wolf Creek at BLM road 16.0	Red Butte	R2W/T27S-21	431214/1225643		X	X
21	Little River at Peel (USGS station ID 14318000)	Glide	R3W/T27S-2	431510/ 123013	X		X
22	Little River below Wolf Creek	Red Butte	R2W/T27S-9	431358/1225704			X
23	Wolf Creek above Wolf Creek Falls	Red Butte	R2W/T27S-16	431306/1225646			X
24	Reference tributary to Wolf Creek above Egglestron Creek	Red Butte	R2W/T27S-16	431337/1225650			X
30	Unnamed tributary to West Fork Wolf Creek in section 32 at road 32.0	Red Butte	R2W/T27S-32	431056/1225734			X
31	Unnamed tributary to West Fork Wolf Creek, #2, at east bank at BLM road 16.0	Red Butte	R2W/T27S-20	431212/1225709			X

Table 4. Nutrient and field data in the Little River and tributaries from reconnaissance samplings during August and November, 1998

[Sites are listed in downstream order. Map ID number refers to locations on figure 4. All nutrient samples were analyzed at the U.S. Geological Survey laboratory in Denver, CO. ft^3/s, cubic feet per second; °C, degrees Celsius; (µS/cm), microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter; µg/L, micrograms per liter; BP, Barometric pressure; mm Hg, millimeters mercury; DO, dissolved oxygen; %, percent; µg/L, micrograms per liter; NH_3, ammonia; NO_2, nitrite; DKN, dissolved Kjeldahl nitrogen; TKN, total Kjeldahl nitrogen; NO_3, nitrate + nitrite; TP, total phosphorus; TDP, total dissolved phosphorus; SRP, soluble reactive phosphorus; <, actual value is less than the indicated amount; USFS, U.S. Forest Service; BLM, Bureau of Land Management; --, no data]

Station Name	Map ID number	Date	Time	Flow (ft^3/s)	Water temperature (x°C)	Specific conductance (µS/cm)	BP (mm Hg)	DO (mg/L)	DO saturation (%)	pH	DKN (µg/L as N)	TKN (µg/L as N)	NH_3 (µg/L as N)	NO_3 (µg/L as N)	NO_2 (µg/L as N)	TP (µg/L as P)	TDP (µg/L as P)	SRP (µg/L as P)
Little River at White Creek Campground	16	8/26/98	1835	8.8	16.4	88	--	9.3	--	8.2	--	100	<2	<5	<1	8	6	<1
Little River above Negro Creek at Coolwater Campground	17	11/7/98	1500	--	--	69	--	--	--	7.6	--	110	--	--	--	32	--	--
"	17	11/10/98	1500	120	6.5	64	725	11.7	100	7.7	<100	260	<2	8	--	33	--	7
Negro Creek near mouth	18	11/7/98	1400	--	--	85	--	--	--	7.6	100	270	<2	9	--	36	--	10
"	18	11/10/98	1420	18	6.4	80	723	11.2	96	7.8	120	280	2	6	--	40	--	9
Emile Creek at USFS/BLM Boundary	10	8/26/98	1915	.99	14.9	56	--	9.1	--	7.5	--	<100	2	37	<1	12	9	5
Emile Creek at mouth	11	8/25/98	1930	1.1	16.2	68	--	9.2	--	7.7	--	<100	<2	<5	<1	11	11	8
Little River above Christian Camp at USFS road 2701 bridge	15	8/26/98	1745	14	18.7	92	--	9.3	--	8.8	--	<100	<2	<5	<1	9	8	<1
Little River above Wolf Creek	14	8/24/98	1200	17	17.4	95	735	10.4	113	8.6	--	<100	<2	<5	<1	9	18	5
"	14	11/9/98	1345	133	7.1	64	738	11.7	100	7.8	140	320	5	9	--	34	--	6
Wolf Creek at BLM road 14.1	20(?)	11/9/98	1435	5.8	7.1	53	--	--	--	7.6	<100	170	<2	96	--	18	--	8
West Fork Wolf Creek	6	8/27/98	1400	.64	12.8	129	--	9.4	--	7.9	--	<100	<2	11	<1	12	14	11
Unnamed tributary to Wolf Creek east bank	5	8/25/98	1615	.01	14	90	--	9	--	7.4	--	<100	16	13	1	12	10	--
Unnamed tributary to West Fork Wolf Creek	7	8/27/98	1320	.09	13.1	101	--	9.1	--	7.6	--	<100	<2	8	<1	8	13	4
"	7	11/9/98	1735	--	--	--	--	--	--	--	<100	210	<2	15	--	32	--	20
"	7	11/10/98	1200	6.2	7.2	41	629	9.8	98	7.3	--	<100	--	--	--	20	--	--
Unnamed tributary to Wolf Creek west bank below Wolf Creek Falls	3	8/25/98	1415	.01	11.2	26	734	10.3	97	6.8	--	<100	12	6	<1	7	7	4
"	3	11/9/98	1120	.31	9.3	26	735	10.6	96	7.5	<100	<100	<2	1	--	33	--	19
Wolf Creek above Egglestron Creek	4	8/25/98	1435	2	13.8	120	--	9.7	--	8	--	<100	<2	31	<1	9	9	6
Egglestron Creek at mouth	2	8/24/98	1600	.42	14.6	213	--	9.4	--	8.2	--	<100	<2	7	<1	15	12	8
"	2	11/9/98	1210	3.7	7.1	84	737	11.4	97	7.8	<100	190	<2	5	--	29	--	8
Wolf Creek near Red Butte	19	11/9/98	1230	4	6.2	91	--	--	--	7.6	130	280	5	7	--	44	--	8
Wolf Creek at mouth	1	8/24/98	1400	2.9	15.2	135	--	9.9	--	8.4	--	<100	<2	84	<1	12	11	7
"	1	11/7/98	1100	--		74	--	--	--	7.4	<100	<100	<2	6	--	20	--	11
"	1	11/9/98	1310	18	7.2	61	739	11.4	98	7.7	--	<100	--	--	--	27	--	--
Little River above Cavitt Creek at new bridge	13	8/26/98	1645	16	19.5	98	--	9.3	--	8.7	--	<100	<2	<5	<1	16	14	10
Cavitt Creek above Withrow Creek	8	8/26/98	1407	2.3	12.7	138	699	9.5	98	8.2	--	<100	<2	18	<1	25	23	19
Unnamed tributary to Cavitt Creek	9	8/27/98	1145	0	14.4	68	724	8.4	86	7	--	<100	<2	34	<1	21	22	18
Little River at Peel	21	8/26/98	1545	26	18.5	111	740	10.1	111	8.4	--	<100	<2	<5	<1	19	18	15

Table 5. Nutrient and field data for sites in the Little River and tributaries, August 1999

[Sites are listed in downstream order. Map ID number refers to locations on figure 4. Sample types— f, field parameters (temperature, dissolved oxygen, pH, specific conductance) only; C, Cooperative Chemical Analytical Laboratory at Oregon State University; U, U.S. Geological Survey Laboratory in Denver, Colorado; °C, degrees Celsius; mm Hg, millimeters of mercury; mg/L, milligrams per liter; µg/L, micrograms per liter; BP, Barometric pressure; ft^3/s, cubic feet per second; (µS/cm), microsiemens per centimeter at 25 degrees Celsius; DO, dissolved oxygen; Alk, alkalinity, as $CaCO_3$; NH_3, ammonia; NO_2, nitrite+nitrate; DKN, dissolved Kjeldahl nitrogen; TKN, total Kjeldahl nitrogen; NO_3, nitrate + nitrite; TP, total phosphorus; TDP, total dissolved phosphorus; SRP, soluble reactive phosphorus; (E), value is estimated; <, actual value is less than the indicated amount; --, no data; USFS, U.S. Forest Service; BLM, Bureau of Land Management]

Station name	Map ID number	Sample type	Dates	Times	Water temperature (°C)	BP (mm Hg)	Flow (ft^3/s)	Specific conductance (µS/cm)	DO (mg/L)	pH	Alk (mg/L)	NH_3 (µg/L as N)	NO_2 (µg/L as N)	DKN (µg/L as N)	TKN (µg/L as N)	NO_3 (µg/L as N)	TP (µg/L as P)	TDP (µg/L as P)	SRP (µg/L as P)
Little River at White Creek Campground	16	f	19990817	1705	17.0	727	--	82	9.1	7.9	--	--	--	--	--	--	--	--	--
Little River at White Creek Campground	16	C	19990818	1500	--	--	--	--	--	--	--	<10	--	50	40	3	40	--	16
Little River at White Creek Campground	16	U	19990818	1500	17.3	741	16.5	80	9.5	8.2	39	3	1	102	E 78	7	18	26	16
Little River at Coolwater Campground	17	f	19990817	1725	16.7	727	--	88	9.1	7.9	--	--	--	--	--	--	--	--	--
Little River at Coolwater Campground	17	f	19990818	1700	17.2	731	--	86	9.3	8.2	--	--	--	--	--	--	--	--	--
Negro Creek near mouth	18	f	19990817	1755	14.5	727	--	110	8.9	7.9	--	--	--	--	--	--	--	--	--
Negro Creek near mouth	18	C	19990818	1200	--	--	--	--	--	--	--	<10		40	30	38	37		11
Negro Creek near mouth	18	U	19990818	1200	13.6	741	4.55	106	10.0	8.1	57	<2	<1	103	E 68	49	12	16	10
Emile Creek at mouth	11	f	19990818	1705	17.7	729	--	65	8.6	7.8	--	--	--	--	--	--	--	--	--
Little River below Emile Creek	--	f	19990818	1705	18.8	729	--	89	8.9	8.3	--	--	--	--	--	--	--	--	--
Little River at USFS road 2701 bridge	15	C	19990817	1530	--	--	--	--	--	--	--	<10	--	40	60	2	42	--	12
Little River at USFS road 2701 bridge	15	U	19990817	1530	19.4	737	25.5	86	9.1	8.5	40	<2	1	<100	<100	6	18	18	10
Little River at USFS road 2701 bridge	15	f	19990817	1735	19.6	727	--	88	8.6	8.3	--	--	--	--	--	--	--	--	--
Little River at USFS road 2701 bridge	15	f	19990818	1715	19.7	731	--	86	9.1	8.6	--	--	--	--	--	--	--	--	--
Little River above Wolf Creek	14	C	19990816	1100	--	--	--	--	--	--	--	38	--	80	100	20	26	--	8
Little River above Wolf Creek	14	U	19990816	1100	15.7	738	20.6	90	9.9	8.1	40	36	1	<100	111	12	19	10	8
Little River above Wolf Creek	14	f	19990816	1850	19.1	735	--	89	8.6	8.2	--	--	--	--	--	--	--	--	--
Little River above Wolf Creek	14	f	19990817	1730	20.1	735 (E)	--	87	8.9	8.6	--	--	--	--	--	--	--	--	--
Little River above Wolf Creek	14	f	19990817	1815	19.2	735 (E)	--	90	8.5	8.2	--	--	--	--	--	--	--	--	--
Little River above Wolf Creek	14	f	19990818	1730	19.9	735 (E)	--	91	8.7	8.6	--	--	--	--	--	--	--	--	--
Wolf Creek at mouth	1	C	19990817	1100	--	--	--	--	--	--	--	<10	--	40	50	24	33	--	9
Wolf Creek at mouth	1	U	19990817	1100	14.6	737	1.86	135	10.0	8.2	68	7	1	148	<100	5	12	14	7
Wolf Creek at mouth	1	f	19990818	1735	16.4	735 (E)	--	141	9.1	8.2	--	--	--	--	--	--	--	--	--
Egglestron Creek at mouth	2	U	19990816	1620	14.8	738	.81	208	9.4	8.1	100	17	<1	<100	E 77	13	20	13	13
Reference tributary to Wolf Creek above Egglestron Creek	24	C	19990816	1430	--	--	--	--	--	--	--	<10	--	40	50	14	18	--	4
Reference tributary to Wolf Creek above Egglestron Creek	24	U	19990816	1430	14.8	734	.02	20	9.0	7.4	9	13	<1	<100	E 85	<5	11	<4	1
Wolf Creek above Egglestron Creek	4	U	19990816	1420	14.5	738	1.47	126	9.5	7.9	58	11	<1	<100	E 85	12	12	8	4

Table 5. Nutrient and field data for sites in the Little River and tributaries, August 1999

[Sites are listed in downstream order. Map ID number refers to locations on figure 4. Sample types— f, field parameters (temperature, dissolved oxygen, pH, specific conductance) only; C, Cooperative Chemical Analytical Laboratory at Oregon State University; U, U.S. Geological Survey Laboratory in Denver, Colorado; °C, degrees Celsius; mm Hg, millimeters of mercury; mg/L, milligrams per liter; μg/L, micrograms per liter; BP, Barometric pressure; ft^3/s, cubic feet per second; (μS/cm), microsiemens per centimeter at 25 degrees Celsius; DO, dissolved oxygen; Alk, alkalinity, as $CaCO_3$; NH_3, ammonia; NO_2, nitrite+nitrate; DKN, dissolved Kjeldahl nitrogen; TKN, total Kjeldahl nitrogen; NO_3, nitrate + nitrite; TP, total phosphorus; TDP, total dissolved phosphorus; SRP, soluble reactive phosphorus; (E), value is estimated; <, actual value is less than the indicated amount; --, no data; USFS, U.S. Forest Service; BLM, Bureau of Land Management]

Station name	Map ID number	Sample type	Dates	Times	Water temperature (°C)	BP (mm Hg)	Flow (ft^3/s)	Specific conductance (μS/cm)	DO (mg/L)	pH	Alk (mg/L)	NH_3 (μg/L as N)	NO_2 (μg/L as N)	DKN (μg/L as N)	TKN (μg/L as N)	NO_3 (μg/L as N)	TP (μg/L as P)	TDP (μg/L as P)	SRP (μg/L as P)
Wolf Creek above Wolf Creek Falls	23	U	19990816	1730	14.2	727	--	131	9.5	8.1	65	11	<1	<100	267	39	16	8	9
Unnamed tributary to West Fork Wolf Creek at BLM road 16.0	31	U	19990817	1340	12.6	714	.01	77	9.0	7.7	41	5	<1	<100	<100	<5	9	13	2
West Fork Wolf Creek	6	C	19990817	1530	--	--	--	--	--	--	--	<10	--	30	40	63	35	--	9
West Fork Wolf Creek	6	U	19990817	1530	13.5	714	.44	135	8.9	7.8	68	15	<1	105	<100	30	14	21	8
Unnamed tributary to West Fork Wolf Creek	7	U	19990817	1720	13.8	714	.07	80	8.6	8.3	51	7	1	105	<100	<5	16	19	9
Wolf Creek at BLM road 14.1	19	C	19990818	1330	--	--	--	--	--	--	--	<10	--	40	40	24	35	--	10
Wolf Creek at BLM road 14.1	19	U	19990818	1330	12.7	698	.35	174	9.1	8.0	89	<2	1	<100	188	19	19	18	10
Unnamed tributary to West Fork Wolf Creek in section 32 at road 32.0	30	U	19990818	1110	11.9	691	.18	56	8.7	7.5	35	2	<1	102	E 63	8	6	11	2
Little River below Wolf Creek	22	C	19990817	1130	--	--	--	--	--	--	--	<10	--	6	90	13	31	--	9
Little River below Wolf Creek	22	U	19990817	1130	18.3	737	24.3	91	10.0	8.5	42	6	2	<100	<100	<5	13	15	5
Little River below Wolf Creek	22	f	19990818	1745	19.6	--	--	96	8.6	8.6	--	--	--	--	--	--	--	--	--
Little River at Cavitt Creek bridge	13	f	19990817	1835	21.1	--	--	94	8.6	8.5	--	--	--	--	--	--	--	--	--
Little River at Cavitt Creek bridge	13	f	19990818	1800	19.4	739	--	108	9.2	8.6	--	--	--	--	--	--	--	--	--
Little River at New Cavitt Creek bridge	13	f	19990818	1740	19.9	738	--	90	9.5	8.8	--	--	--	--	--	--	--	--	--
Little River at Peel	21	f	19990818	1830	19.5	741 (E)	--	112	9.0	8.4	--	--	--	--	--	--	--	--	--
Little River at Peel	21	C	19990818	1000	--	--	--	--	--	--	--	<10	--	12	100	3	32	--	4
Little River at Peel	21	U	19990818	1000	18.7	741	32	109	9.0	8.0	51	10	1	E 78	210	<5	10	12	2
Little River Highway 27 bridge below Peel	--	f	19990817	1900	20.3	--	--	112	8.7	8.3	--	--	--	--	--	--	--	--	--

Table 6. Analytical methods and detection levels for nutrient analyses performed at the U.S. Geological Survey National Water Quality Laboratory (NWQL) and Oregon State University Cooperative Chemical Analytical Laboratory (CCAL)
[EPA, Environmental Protection Agency]

	NWQL			CCAL	
Constituent	Method	Detection level (µg/L)	Reference	Method	Detection level (µg/L)
Total phosphorus (TP)	EPA 365.1	8		EPA 424C	1
Total dissolved phosphorus (TDP)	EPA 365.1	6		--	--
Soluble reactive phosphorus (SRP)	I260689	1	Fishman, 1993	EPA 424F	1
Total organic + ammonium nitrogen (TKN)	I451591	100	Patton and Truitt, 2000	Kjeldahl, Nessler finish	10
Dissolved organic + ammonium nitrogen (DKN)	I261091	100	Patton and Truitt, 2000	Kjeldahl, Nessler finish	10
Ammonium nitrogen (NH_4-N)	I252589	2	Fishman, 1993	EPA 417F	10
Nitrate + nitrite nitrogen (NO_3-N)	I254691	5	Fishman, 1993	EPA 418F	1
Nitrite nitrogen (NO_2-N)	I254289	1	Fishman, 1993	--	--

Algal samples from 1999 were collected by scraping known areas from rocks into jars, homogenizing the sample in a blender, and subsampling the resulting slurry for algal biomass and chlorophyll *a*. Analysis of samples for ash free dry mass (AFDM) and chlorophyll *a* used standard methods (American Public Health Association, 1998). AFDM and chlorophyll *a* were analyzed in triplicate in the Oregon District Laboratory.

Quality Assurance

Nutrient and major-ion sample results were checked for bias through the use of blank samples. Blanks were prepared in the field using the same equipment as environmental samples, with inorganic-free water obtained from the USGS's Ocala Field Services warehouse. Laboratory blank and standard reference samples to check for bias and accuracy in low-level nutrient analyses were made in the Oregon District Laboratory, using Standard Reference Materials traceable to the National Institute of Standards and Technology (NIST-SRMs) in inorganic-free water. These quality-assurance samples were simultaneously submitted in triplicate to the NWQL and to CCAL. Precision was evaluated in field replicate samples that were submitted to both laboratories as well.

Data from the August 1998 sampling indicated a possible contamination of dissolved organic nitrogen because dissolved Kjeldahl nitrogen (DKN) values were substantially higher than those for than TKN at more than half of the sites. TKN and DKN values less than 100 µg/L were expected on the basis of previous samplings in the watershed (Anderson and Carpenter, 1998) and previous data from the Oregon Department of Environmental Quality (D. Ades, Oregon Department of Environmental Quality, 1998, written commun.). Data for DKN from August 1998 were subsequently deleted from the database.

Results of reference sample analysis (table 7) indicated that both laboratories performed comparably and well for phosphorus analysis but each had small bias from contamination of NH_3-N, with the NWQL apparently having a somewhat higher bias (average ~12 µg/L) than CCAL (average ~6 µg/L). Bias in NH_3-N measurement is not unusual because contamination is notoriously difficult to avoid (Holmes et al., 1999). Although new methods to cleanly sample and analyze for NH_3-N have recently been developed (Holmes et al., 1999), they have not yet been adopted by the NWQL or CCAL. The largest difference was apparent in analysis of DKN. The NWQL's reporting limit for this analysis (as for TKN) is 100 µg/L, although it will report values as estimates for detections between 50 and 100 µg/L; whereas CCAL's analytical detection limit for DKN and TKN is approximately 10 µg/L. For the DKN samples, NWQL analysis had a highly variable positive bias ranging from 46–110 µg N/L.

Among environmental samples analyzed between the two laboratories, there was considerable disagreement for phosphorus analysis (table 5), despite the favorable comparison of previous standard reference samples. Concentrations for TP from the NWQL were approximately half those reported from the CCAL, although results were comparable for SRP. Results for NO_3-N and

Table 7. Comparison of nutrient concentrations from standard reference samples analyzed at the U.S. Geological Survey's National Water Quality Laboratory and Oregon State University's Cooperative Chemical Analytical Laboratory, August 1998

[Reference samples (R1, R2, R3) were prepared with only organic nitrogen and organic phosphorus; samples were also analyzed for ammonium nitrogen to check for decomposition of the organic nitrogen and to evaluate possible bias from contamination in low level ammonium data. NWQL, National Water Quality Laboratory; CCAL, Cooperative Chemical Analytical Laboratory; *, Replicate analyses performed by CCAL; (E) Concentrations are below the method reporting limit and are considered estimates; µg/L - micrograms per liter]

	Organic P (dissolved digested P)			Ammonia-nitrogen			Organic-N + ammonia N (dissolved Kjeldahl N)		
Sample	Nominal value (expected) (µg/L)	NWQL (reported) (µg/L)	CCAL (reported) (µg/L)	Nominal value (expected (µg/L)	NWQL (reported) (µg/L)	CCAL (reported) (µg/L)	Nominal value (expected) (µg/L)	NWQL (reported) (µg/L)	CCAL (reported) (µg/L)
Blank	0	<4	<1	0	3	<10	0	(E) 71	<10
Blank	0	<4	*1/1	0	9	<10	0	(E) 88	<10
Blank	0	<4	<1	0	7	<10	0	(E) 95	<10
R1	21	21	22	0	13	0	43	(E) 89	40
R2	21	19	22	0	12	*4/7	43	134	40
R3	21	21	24	0	10	7	43	153	50

NH_3-N in environmental samples were also comparable, although the small positive contamination from NH_3-N was evident in some samples. On the basis of QA samples (table 7), data on organic nitrogen (TKN and DKN) from CCAL were considered more reliable than those from the NWQL; however differences in TP concentrations between the two labs remain unresolved. For most of the discussion here, data from the NWQL are used because samples were not submitted to CCAL from all sites; however reference is made to CCAL data where interpretation of nutrient status would be different.

Environmental Data

Precipitation during 1999 in the Little River watershed was slightly higher than normal (Owenby and Ezell, 1992), and temperatures were near normal. Nonetheless, many tributaries located directly within the BLM's proposed fertilization units were dry during sampling in August 1999 and an earlier reconnaissance trip in July 1999. The lack of water necessitated sampling farther downstream to find adequate water. However, as a result of relocating downstream from the fertilization units, samples contained water from additional tributaries that had entered the river from upstream subwatersheds containing mixed private and Federal (BLM) timberlands. In general, discharge was low in the Wolf Creek subwatershed, ranging from less than 0.01 ft^3/s in some of the upland tributaries to 1.9 ft^3/s at the mouth of Wolf Creek. These discharges were similar to those in August 1998, although Wolf Creek had 2.9 ft^3/s at the mouth during 1998. Most tributary streams were covered by dense canopies of alder and low brush, and could often be straddled. An attempt was made to mass balance flows in Wolf Creek to evaluate possible ground-water discharge. Though most significant flows were measured, Wolf Creek upstream of the confluence with West Fork Wolf Creek, with visible flow, was not measured due to inaccessibility. It is therefore unclear if the approximately 0.5 ft^3/s gain from upstream to downstream resulted from unmeasured inflows of tributaries or ground water. Discharge in the Little River main stem increased from 16.5 ft^3/s at White Creek Campground (the upstream border between BLM- and USFS-managed lands) to 32 ft^3/s at the USGS gaging station at Peel (table 5), a value equal to the average monthly discharge for August at the Peel gage during a previous period (1953–1987) when the station was in operation (Moffatt et al., 1990).

Where there was enough water to sample in the Wolf Creek subwatershed and its tributaries, field parameter data indicated few overt water-quality problems. Maximum temperatures were less than 16.5°C (degrees Celsius) at all sites and as low as 11.9°C at some sites. Maximum pH was as high as 8.4 at the mouth of Wolf Creek in 1998, and 8.3 in West Fork Wolf Creek in 1999, but at all other sites ranged from 7.4 to 8.2. Dissolved oxygen (DO) concentrations were lowest (8.7 mg/L) at site #30, an unnamed tributary high in the West Fork Wolf Creek drainage; however, this investigation did not target the early morning period, when DO could be lowest if significantly affected by periphyton respiration. Therefore, the minimum DO conditions during August 1998 are unknown. Algal biomass was low and difficult to observe without magnification, at almost all locations in the Wolf Creek subwatershed except for two areas with open canopies. These were the mouth of Wolf Creek (site #1) and the upright wall of Wolf Creek Falls.

Wolf Creek Falls is unique because of its physical structure, in which much of the water disperses and trickles thinly over a long, wide slab of rock. This slab has a western aspect and open canopy, so solar exposure is relatively good. As a result, the slab had a continuous and consistently thick film of healthy, bubbling filamentous green algae covering it as the water slowly poured over it, resembling a trickling-filter apparatus in a wastewater treatment plant. Due to air exposure, no grazing aquatic insects such as the stone-case building caddisfly *Dicosmoecus*, (observed elsewhere in the lower reaches of Wolf Creek and the Little River) colonized this mat. Thus the waterfall provides a naturally occurring habitat for filamentous algal growth with minimal grazing pressures. This phenomenon undoubtedly acts to reduce nutrient concentrations through uptake during certain times of the year, and may also increase pH in Wolf Creek upstream of the mouth such as previously observed (Powell, 1995). During mid-August 1999, the falls increased pH only about 0.1 units from top to bottom.

The mouth of Wolf Creek (site #1) is covered by steps of bedrock with little alluvial material, and at this location solar exposure to the stream is the greatest in the subwatershed. There, luxuriant mats of filamentous green algae were observed, with individual strands exceeding 5 feet in length, during late August-early September 1999.

In contrast to the Wolf Creek subwatershed, field parameter data in the main-stem Little River appear to reflect the effects of upstream land uses. During August 1999, daily maximum pH in the Little River (fig. 6) increased from 8.2 at White Creek Campground (site # 16) to 8.8 above Cavitt Creek (site # 13), with the largest increase (0.3 units) occurring in the 1.5 mile reach between the mouth of Emile Creek (river mile 14.9, site not shown on fig. 4) and the USFS road 2701 bridge (site #15). Similarly, during August 1998 pH was higher than the State standard of 8.5 at main-stem sites between the USFS road 2701 bridge and Cavitt Creek (table 4). These patterns were similar to those observed by Powell (1998) in figure 4. Like pH, maximum water temperatures (fig. 7) did not meet the State standard (17.8°C) from below Emile Creek (river mile 14.9) to the Highway 27 bridge below Peel (river mile 3.2, not shown on fig. 4). Temperatures in the Little River during August 1998 did not meet the standard at the USFS road 2701 bridge, above Cavitt Creek, or at the Peel gage; temperature at the site above Wolf Creek may have met the standard because it was measured around noon rather than in late afternoon as most other main-stem sites were. Dissolved oxygen concentrations met the State standard of 8.0 mg/L at all stations, but were not investigated during the early morning to evaluate diel variation associated with periphyton metabolism.

Geologic or land-use differences in the area may be reflected in the water quality. In particular, the smaller streams draining the western slopes of the Wolf Creek subwatershed (sites 24, 7, 30, and 31) had distinctly different chemical signatures, as measured by major-ion concentrations (fig. 8) and specific conductance (table 5), from other sites in Little River and Wolf Creek. Although the major anion at all sites was almost exclusively bicarbonate, and calcium and magnesium were the dominant cations at most sites, sodium was increased both in concentration and percent of total cations at these four sites and calcium concentrations were reduced. The same four sites also were the most dilute in the Wolf Creek subwatershed (specific conductances 20–79 µS/cm [microsiemens per centimeter]). In contrast, the two sites draining the east side of the subwatershed (site 2—Egglestron Creek, and site 20—Wolf Creek at the BLM 14.1 road) had the highest specific conductances (208 and 174 µS/cm, respectively) measured during the 1999 survey. Major ions were not measured during 1998, but specific conductances during August 1998 were similar to those during the August 1999 survey. The causes of the differences among sites, or their possible effects on nutrient retention in soils and (or) transport in the streams, have not been expressly investigated.

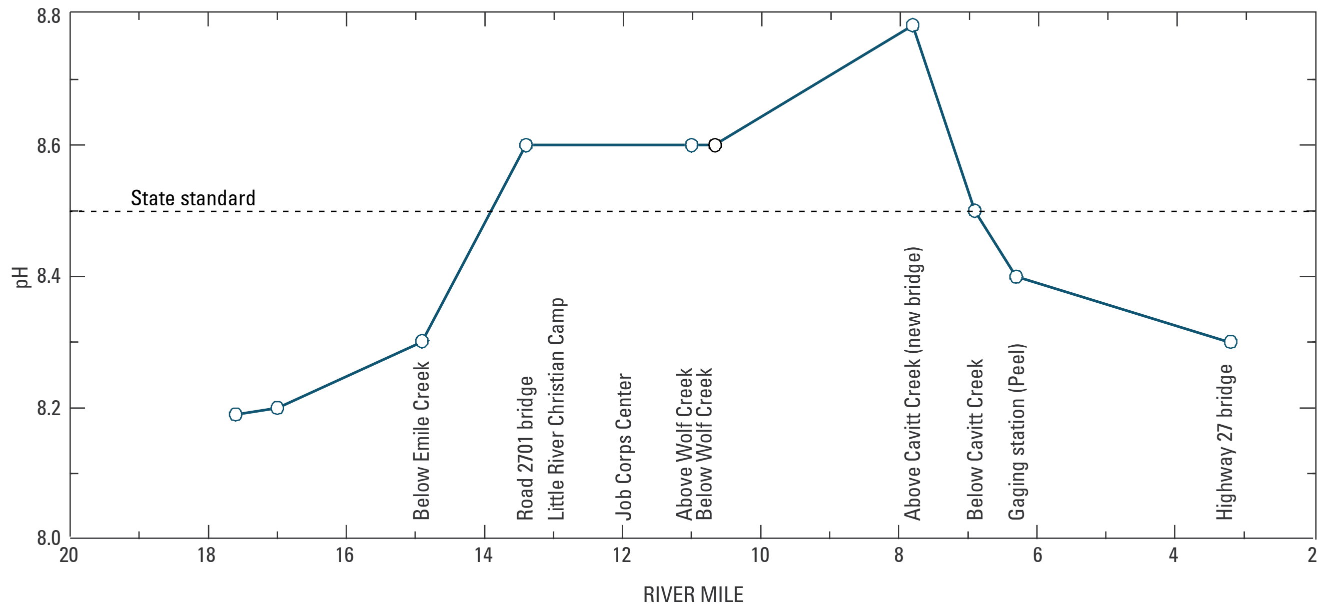


Figure 6. Afternoon pH in the main stem of the Little River, August 1999.

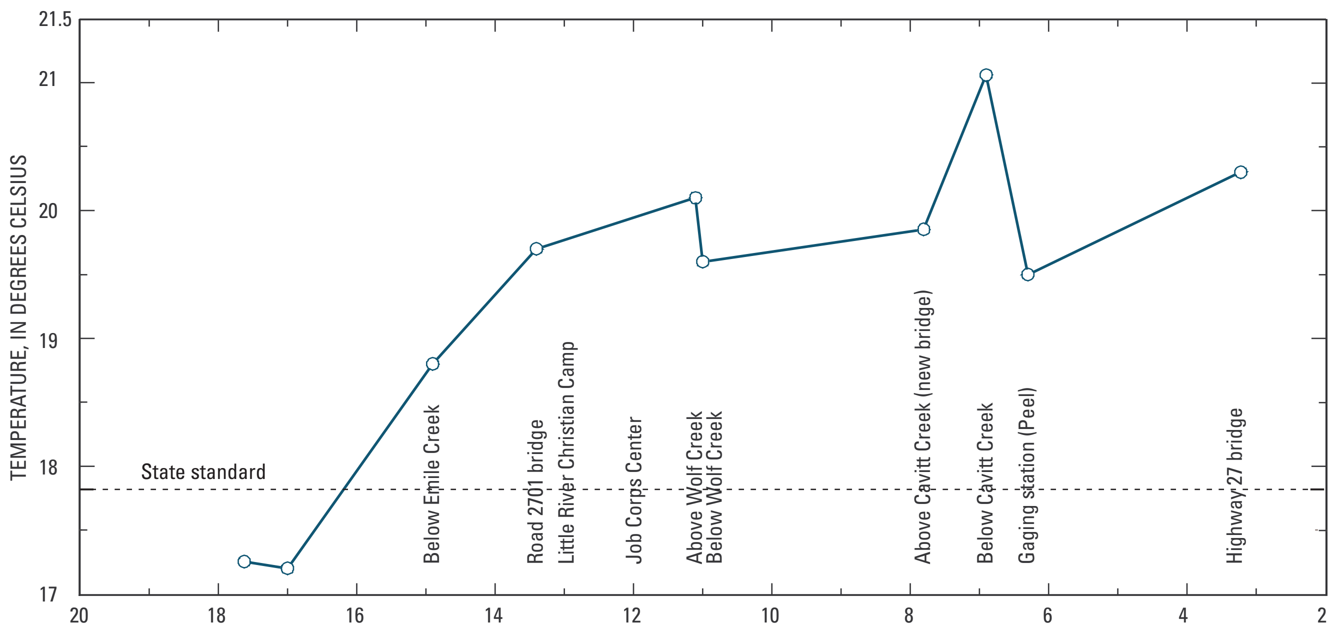


Figure 7. Afternoon temperature in the main stem of the Little River, August 1999.

Nutrient concentrations in the Little River watershed and Wolf Creek subwatershed were typically low during August 1999, consistent with nitrogen limitation as previously observed (Anderson and Carpenter, 1998) (fig. 9). Median concentrations for all dissolved inorganic nitrogen species were less than 20 µg/L, and TP concentrations (median 20 µg/L) were typically high enough to saturate algal growth (Bothwell, 1989). Nitrate-N and NH_3-N concentrations were slightly higher in streams of the Wolf Creek subwatershed than in the main stem Little River, possibly reflecting uptake of nitrogen in Little River and (or) light limitation in Wolf Creek. In contrast SRP concentrations, and to a lesser extent TP concentrations, were somewhat higher in Little River than in Wolf Creek. Organic plus ammonium nitrogen, measured as TKN or DKN, was generally less than 100 µg/L (using data from CCAL).

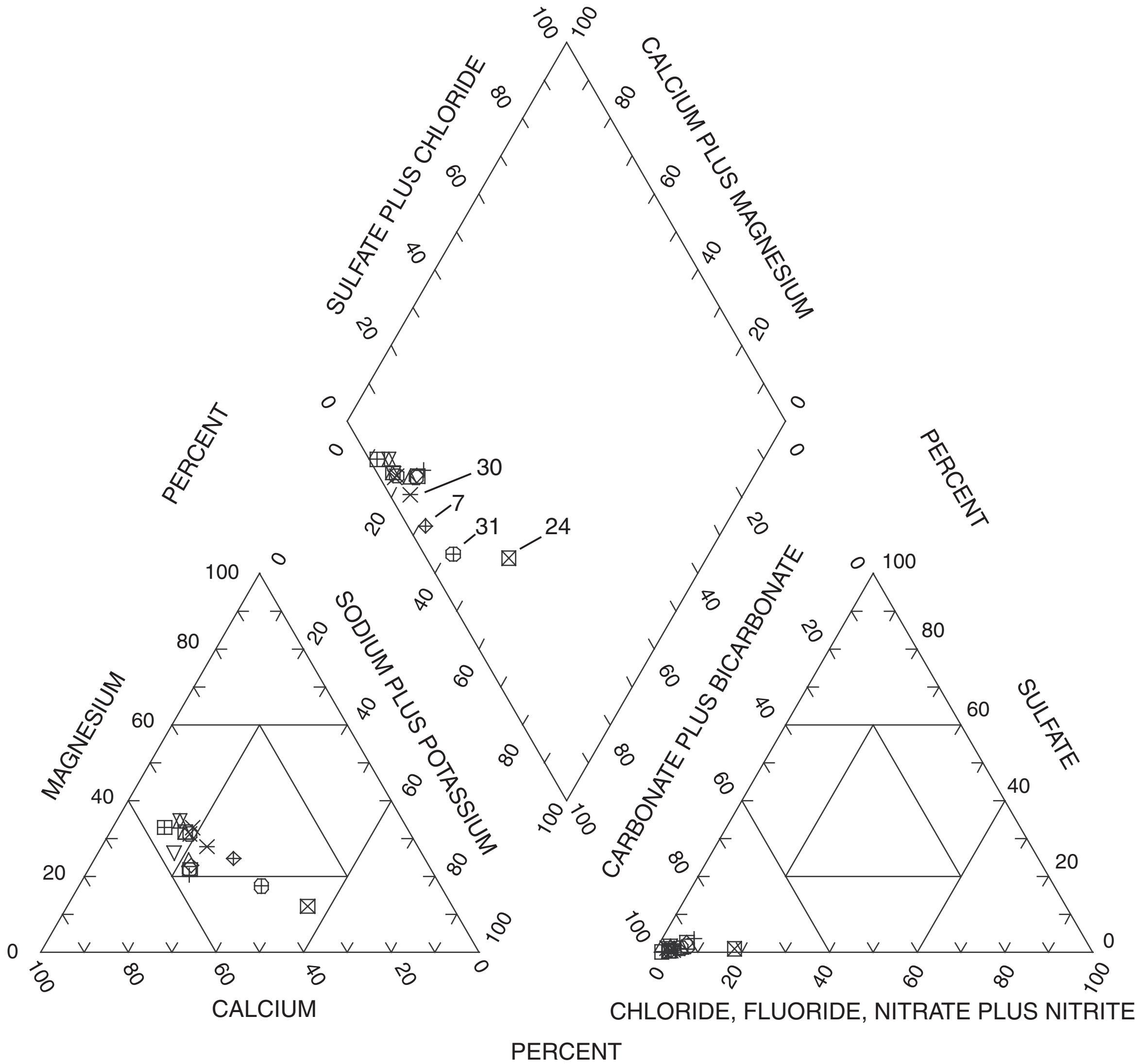


Figure 8. Major ion chemistry in Little River and Wolf Creek. (Sites that are the most chemically distinct are the reference tributary to Wolf Creek above Egglestron Creek [site 24], an unnamed tributary east of the West Fork of Wolf Creek on road 16.1 [site 30], an unnamed tributary west of the West Fork of Wolf Creek on road 16.1 [7], and an unnamed tributary to West Fork Wolf Creek in section 32 at road 32.0 [site 31]. See figure 4 for locations.)

Typically N:P ratios <7 (on an atomic weight basis) are considered indicative of nitrogen limitation, whereas N:P >10 could indicate phosphorus limitation (Wetzel, 1983; Hillebrand and Sommer, 1999). For this report, ratios of total nitrogen tototal phosphorus (TN:TP) during 1999 depend on the source of the data (NWQL or CCAL) and the treatment of censored data (that is, nondetections). Using NWQL data only, and taking nondetected concentrations at the value of their respective reporting limits (an approach that overestimates nitrogen in nondetected samples), the median TN:TP was 9.3 (range 5.1–24). In contrast the median TN:TP was 2.3 (range 1.1–4.6) using data only from CCAL, but this source accounts for only 12 of 17 samples because not all sites had samples submitted to CCAL. By substituting NWQL organic nitrogen data with CCAL data where available, the median TN:TP is 5.9 (range 2.6–15.5). Thus, in a few cases, phosphorus limitation might be indicated using censored data from the NWQL, but using data from CCAL, with less positive bias in TKN and higher TP concentrations, nitrogen limitation is almost universally indicated. Similar results are obtained for the ratio of dissolved inorganic nitrogen to soluble reactive

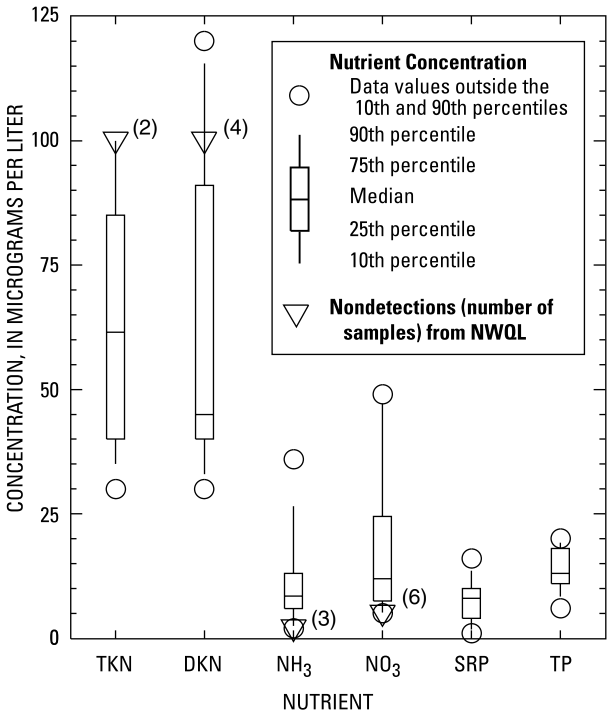


Figure 9. Distribution of nutrient concentrations in the Little River and tributaries during August, 1999. (All data are from the U.S. Geological Survey's National Water Quality Laboratory (NWQL) in Denver, Colorado, except dissolved Kjeldahl nitrogen (DKN) and total Kjeldahl nitrogen (TKN), which include data from both NWQL and Cooperative Chemical Analytical Laboratory (CCAL) at Oregon State University.)

phosphorus (DIN:SRP) (table 5). Similar conclusions can also be drawn from the August 1998 data. Although median TP concentrations during 1998 (~12 µg/L) were lower than from 1999, N-limitation is supported by a median DIN:SRP ratio of ~1.

Little River watershed nitrogen concentrations (tables 4 and 5) are lower than the national median flow-weighted NO_3-N and total N (NO_3-N + TKN) concentrations (87 and 260 µg/L, respectively) reported by Clark et al. (2000) for relatively undeveloped streams. In contrast, Little River watershed phosphorus concentrations are roughly equivalent to national median flow-weighted concentrations in those streams (TP=22 µg/L, SRP = 10 µg/L). These comparisons are made with caution, however, because data are not available to determine flow-weighted median nutrient concentrations for the Little River watershed. Also, data for the Little River watershed are from summer, when benthic uptake and the lack of particulate matter probably cause concentrations to be lower than flow-weighted median concentrations would be. Even so, during the stormflows sampled in November 1998, nitrate and total N concentrations were equivalent to or lower than those reported by Clark et al. (2000), and TP was only slightly elevated. Together these comparisons support the suggestion that streams in the Little River watershed are strongly nitrogen limited, indicating that algal growth could be stimulated by inputs of fertilizer nitrogen to streams, even in small quantities.

Periphytic algal biomass (reported as ash free dry mass, or AFDM) was generally low in streams of the Wolf Creek subwatershed, but relatively abundant in Little River (table 8). In many cases, rocks in the smaller Wolf Creek streams lacked visible algal growth, though chlorophyll *a* analysis indicated nominal growth. Algal biomass within the Wolf Creek subwatershed was visibly heaviest on the rock wall of Wolf Creek Falls, which was not sampled during this survey. The highest measured biomass in the subwatershed, as AFDM, was at the mouth of Wolf Creek, although chlorophyll *a* in Egglestron Creek was similar to that at the mouth of Wolf Creek. Algal biomass in Little River was highest below Wolf Creek and at the Peel gaging station, and chlorophyll *a* was also highest below Wolf Creek. During the August survey, algal nuisance conditions (mats with filaments several feet long) were not observed; however nuisance growths of green algae, with filaments up to several feet long, were observed during a brief inspection in early September 1999 at the mouth of Wolf Creek, and in isolated mats in Little River above and below Wolf Creek. Macroinvertebrate grazers (a case building caddisfly of the genus *Dicosmoecus*) were abundant in Little River and the lower reaches of Wolf Creek in early August, and likely contributed to keeping algal biomass low. Their subsequent emergence in the warm waters sampled during mid-August may have allowed algal growth to accelerate afterwards.

Spatial patterns of water quality in the watershed (fig. 10) generally followed the conceptual model proposed previously (fig. 3). For instance, NO_3-N concentrations during the 1998 survey were generally higher at upstream tributary sites where dense riparian shading prevented light penetration, somewhat lower at downstream tributary sites where more light was available, and lowest in the main stem where light is not limiting and algal uptake apparently reduces nutrient concentrations.

Table 8. Algal biomass and chlorophyll *a* measured in the Little River watershed, August 1999
[AFDM, ash free dry mass; chl *a*, chlorophyll *a*; g/m^2, grams per meter squared; mg/m^2, milligrams per meter squared]

Sampling site	Site number	AFDM g/m^2	Chl *a* mg/m^2
Unnamed tributary to West Fork Wolf Creek in section 32 at road 32.0	30	2.2	2.6
Wolf Creek at USFS road 14.0	19	3.7	9.4
Wolf Creek above falls	23	2.6	7.8
Wolf Creek below falls	33	4.0	14
Wolf Creek above Egglestron Creek	4	6.5	19
Wolf Creek at mouth	1	17	24
West Fork Wolf Creek	6	2.1	11
Unnamed tributary to West Fork Wolf Creek	7	2.3	5.5
Reference tributary to Wolf Creek above Egglestron Creek	24	4.9	7.0
Unnamed tributary to West Fork Wolf Creek, number 2	31	2.8	2.9
Egglestron at mouth	2	3.3	27
Negro Creek at mouth	18	2.7	11
Little River at White Creek Campground	16	14	21
Little River above Job Corp	15	16	35
Little River above Wolf Creek	14	18	60
Little River below Wolf Creek	22	20	68
Little River at Peel	21	22	57

The one main-stem site at which NO_3-N was greater than 10 µg/L was just upstream of Wolf Creek, illustrating possible NO_3-N inputs from the youth camp, Job Corps Center, or individual streamside residences. In contrast, total-phosphorus concentrations were moderate at most sites except the smallest tributaries sampled and in Little River at sites below Wolf Creek, where concentrations were again reduced by uptake. Daily maximum temperature and pH were each typically higher in the Little River than in the tributaries, except at the most upstream sites surveyed in Little River. Maximum pH was lower overall at upstream, shaded sites and somewhat higher at downstream sites. Patterns of chlorophyll *a* from 1999 followed those of pH. Chlorophyll *a* and pH were highest in the main stem of the Little River below Emile Creek, where an open canopy likely indicates that algal growth is not limited by light availability, and nutrient sources are apparently adequate to support relatively high primary production. During August 1999, NO_3-N was low (<10 µg/L) throughout the upper main stem Little River, but it was relatively high (> 40 µg/L) at the mouth of the shaded but heavily timbered (and possibly recently fertilized) Negro Creek subwatershed. However, associations between the water-quality parameters in figure 10 and stream order or elevation were not statistically significant, indicating that a variety of processes and conditions must be considered in the Little River watershed for studies of forest fertilization to be conclusive.

FRAMEWORK FOR FERTILIZATION STUDY IN WOLF CREEK AND LITTLE RIVER WATERSHED

In keeping with its mission under the President's Northwest Forest Plan (U.S. Forest Service and Bureau of Land Management, 1994), the BLM plans to study the effects of forest fertilization with urea-N on water quality and stream ecology in the LRAMA to determine if fertilization adversely affects water quality and stream biota. Specifically, the objectives of the study are to determine:

1. Effects of fertilizer nutrient inputs on the aquatic ecosystem, including algae and higher trophic levels, such as macroinvertebrates and (or) amphibians,

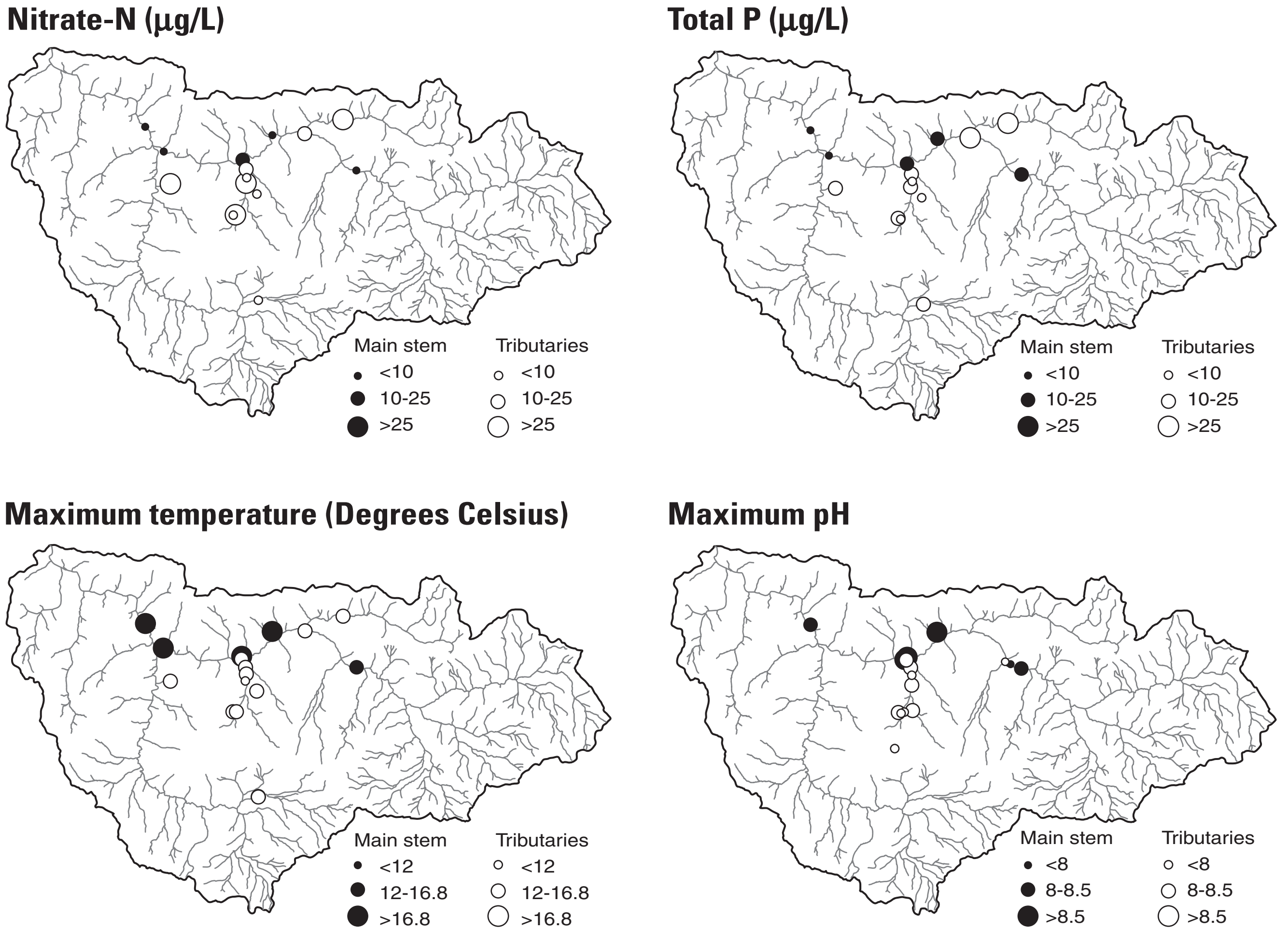


Figure 10. Concentrations of nitrate-nitrogen and total phosphorus, and daily maximum temperature and pH in the Little River Basin during August, 1998. (Sampling sites are given in table 3 and figure 5, and water quality data are provided in tables 4 and 5.)

2. Relations in the Little River watershed among nutrient inputs (natural and anthropogenic), watershed characteristics, and water quality, particularly pH, DO, nutrients, and temperature, and

3. Downstream cumulative effects, both spatial and temporal, of forest fertilization on water quality and aquatic-biological systems.

An investigation into ecological effects from forest fertilization in the LRAMA will require care and detail to distinguish effects of fertilization from natural processes and other land uses. Ancillary factors include the history of timber production on alternating federal and private lands, mixed-age stands (including recent clearcuts), recent forest fertilizations on some private lands, and residential and recreational land adjacent to the Little River. Existing extremes of pH and temperature that already suggest possible degradation from natural conditions may confound interpretation of water-quality data.

An obvious approach would be to used paired drainages, or sets of pairs, to provide controls and treatments. Where possible, this approach is suggested for fertilization studies. Unfortunately, few, if any, drainages in the Little River watershed's study area are small enough to contain somewhat homogeneous upstream land uses (especially without private timberland in the subbasins) and yet large enough to avoid drying in summer. The best choices for control streams include one small drainage in the Wolf Creek subwatershed with predominantly old-growth trees, and several possible locations in tributaries to Little River well upstream of Wolf Creek. Furthermore, benthic growth in the small tributary streams and immediately below the areas to be fertilized is most likely limited by light rather than nutrients (fig. 3). Thus, fertilizer nutrients are expected to be transported downstream to lower reaches of Wolf Creek and Little River, where investigation of cumulative biological effects would be most confounded. This lack of suitable subbasin pairs necessitates an alternative approach, with emphasis on comparison of conditions before and after fertilization and on longitudinal changes in study streams. Longitudinal evaluations involving an upstream-downstream comparison need to take into account the possible inputs from upstream private timber lands with unknown histories (fig. 4) and the likely changes in stream function along an elevational or stream-order gradient (fig. 3).

Adherence to water-quality criteria for nutrients is not perceived as an effective benchmark to determine fertilization's effects on water quality in this case, with the possible exception of pulsed ammonia toxicity during rainstorms immediately following fertilization. Rather, the focus will primarily be on biological endpoints such as measures of biomass (AFDM and chlorophyll *a*), changes in algal community structure (autecology, species diversity, and dominant species types) and function (primary production, nutrient sequestration and uptake), secondary effects on water quality (DO and pH), and possibly secondary interactions with higher trophic levels (macroinvertebrate grazing or amphibian abundance). Nutrient processes will be investigated to provide insights into relevant ecological processes, evaluate transport, and make comparisons among stations.

Ideally, incorporation of fertilizer-nutrient into biological tissues could be traced using unique signatures of naturally occurring ^{15}N (herein termed "natural-^{15}N"), the stable isotope of nitrogen (Lajtha and Michener, 1994; Kendall and McDonnell, 1998). Natural isotopes of oxygen (^{18}O) can also be used in conjunction with ^{15}N to determine hydrologic flow paths and water sources (Kendall and McDonnell, 1998). However, urea fertilizer typically has a natural-^{15}N signature that generally cannot be differentiated from natural-^{15}N found in forest soils, with $\delta^{15}N$ values near 0 ‰[2]. It is possible that volatilization, nitrification, and uptake processes in forest floors and along hydrologic pathways would cause fractionation of the urea's ^{15}N to a heavier fraction, producing a traceable signal (Udy and Dennison, 1997). However, several algal and moss samples from throughout the watershed, including downstream of Federal lands, private lands fertilized in fall 1998, and potential septic or WWTP influences in Little River, did not indicate a strong enough gradient ($\delta^{15}N>3$ ‰) to identify sources or processes. Thus it is unlikely that natural-^{15}N alone can adequately be used to trace urea-N transport from forest fertilization through the forest floor and into aquatic biota. Nonetheless, additional reconnaissance of natural-^{15}N, alone or in conjunction with ^{18}O, and testing of the urea to be applied, is warranted because of the great advantage that this technique would provide if a distinct fertilizer signature were available.

There have been many studies of N movement in forest floors or into trees using fertilizers artificially enriched in ^{15}N (herein referred to as "labelled-^{15}N") to ensure a distinct tracer (Marshall and McMullan 1976; Nason et al., 1988; Preston et al., 1990; Fry et al., 1995; Downs et al., 1996; Jordan et al., 1997), but no studies to date have traced the movement of these isotopes from the forest floor to streams or aquatic biota. Although this technique is likely to be the most definitive way to trace the effect of urea-N into streams, it would also be expensive for even one of the fertilizer units in the BLM's proposed fertilization. A coarse cost estimate was made for an upper elevation drainage basin with minimal upstream influence from private land and a flowing stream in 1999 (above site 30, in section 32 of the USGS Red Butte 7-1/2 minute topographic map). Using basic costs from Fry et al. (1995), and scaling the effort to the 0.2-square-mile (52 ha) fertilization unit immediately upstream of site 30, with an application rate of 224 kg/ha, yields an estimate of $75,000, including purchase of the labelled urea and logistical costs of mixing the labelled urea with nonlabelled urea prior to application. Furthermore, application of labelled-^{15}N to a limited area could prove to be highly useful near the area of application, but its signature might be diluted below detection in

---

2 ^{15}N enrichment is measured in a relative sense compared to a known reference material. The delta value, expressed as $\delta^{15}N$ and with units of ‰ (parts per thousand), is determined as $\delta = (R_X/R_S-1)*1000$, where R is the ratio of the heavy to light isotope in the standard (S) and the sample (X). A positive δ value means that the isotopic ratio of the sample is higher than in the standard, and a negative δ value means that the sample is isotopically lighter than the standard.

downstream reaches most likely to respond biologically to increased nutrients. Thus, the use in this study of labelled urea to trace the movement of nitrogen into streams and ecological compartments, while an attractive method, could be too expensive for practical use at the drainage-basin scale. More detailed cost estimates of this method for specific locations would be warranted prior to final decisions about its use.

Possible Study Approaches

The level of investigation at Little River, and therefore the degree to which the study would address its objectives, will greatly depend on available resources. Table 9 indicates two possible levels of research and associated activities to address fertilizer impacts. Although these approaches are targeted to the Little River watershed and some of its specific complications, the concepts could apply broadly to other investigations of the effects of forest fertilization on aquatic systems.

A basic investigation would examine study streams for gross biological responses (table 9) before and after fertilization. The focus would be on the Wolf Creek subwatershed, although a few sites in the Little River would be sampled as well. The relative loading of nutrients to streams from fertilized and unfertilized areas would be determined. In order to separate confounding upstream influences in the Little River watershed, relations between catchment scale characteristics such as upstream forest-and land-management history, slope, riparian vegetation, geology, surface and subsurface hydrology, stream morphology and water-quality constituents would also be considered. Cumulative effects downstream would be evaluated at a relatively coarse level.

A variety of sampling activities would be utilized for the above approach including standard water-quality analyses, plus periphytic algal biomass and species data, prior to and following fertilization. Synoptic surveys would provide "snapshots" (Salvia et. al, 1999) of summer low-flow conditions before and after fertilization. Monthly sampling at a few sites during summer would provide data on algal biomass and succession, and on the variability of nutrients and algae. Using these data, gross summer-nutrient loading could be estimated, major sources of water and nutrients would be defined, and estimates could be made of fertilizer effects on algal growth. Major-ion data would help evaluate geological influences on water quality and quantity, and possibly indicate a catchment response to fertilizer through ion exchange. Recording monitors for temperature, pH, dissolved oxygen, and specific conductance would be used to define diel variability in those parameters, and the timing and magnitude of their seasonal maxima. The validity of the assumption that primary production is nitrogen limited would be tested in an assay using nutrient diffusing substrates. A reconnaissance of possible ground-water inputs would be done by sampling seeps and mass balancing streamflows. Potential signatures of different nutrient sources using natural-^{15}N levels in water and algal tissues would be assessed at a few locations. Selection of sites longitudinally within the Wolf-Creek subwatershed and the Little River will allow differentiation of runoff from upland fertilized stands compared with unfertilized stands in mixed-use forested areas, as well as generalized cumulative effects downstream. This analyses would be aided with broad characterization of upstream land uses from existing GIS data layers.

If there is a large biotic response to fertilization (objective 1, page 35), the basic approach above might successfully detect it. However, with the variability of forest management history in Wolf Creek, and of upstream nutrient sources to Little River, it is likely that subtle effects on biota or subtle cumulative changes in water quality would not be attributable specifically to any one cause. Nor would this effort generate information about the relative retention or loss of applied urea-N or its downstream transport, or define potentially important transport processes (through riparian or hyporheic zones, benthic recycling, or spiraling of nutrients). Relations among nutrients, riparian characteristics, aquatic habitat, or other water-quality parameters such as pH could also be tenuous.

Effects of fertilizer-N on higher trophic levels and questions about cumulative impacts could be better addressed with a more extensive level of study (table 9). This could include expanded efforts to evaluate (1) the status of water quality and nutrient sources in the Little River above and below

Table 9. Research components for different levels of investigation of effects of urea fertilization on water quality and stream ecology

[Research objectives are given on page 35. esp, especially; DO, dissolved oxygen; SC, specific conductance; GIS, Geographic Information System]

Research level	Objectives addressed	Approach	Activities
Basic	Gross responses in nutrient concentrations and algal biomass Relations in Little River watershed among watershed characteristics and water quality (esp. pH, DO, nutrients inputs, and temperature)	Basic analysis of water quality and algal growth, primarily within Wolf Creek subbasin but including some sites in Little River Determine gross loading of nutrients to streams and major sources Determine possible differential nutrient sources or hydrologic flow paths using streamflow and geochemical data and reconnaissance of ground-water inputs Evaluate land use and relate to water quality. Determine gross cumulative effects on Little River	• Late summer synoptic surveys before and after fertilization. Include nutrients, major ions, daily maximum pH and DO, algal biomass and species composition. Evaluate stable isotopes as possible indicators of differences in N sources. Include a few sites in Little River to evaluate influences of upstream nutrient sources • Monthly sampling for nutrients, algal biomass, algal nutrient content (including algal slough/drift), and field parameters at select sites • Verification of limiting nutrient using nutrient diffusing substrates • Recording monitors for continuous measurement of flow and field parameters (pH/DO/SC/temp) during selected seasons before and after fertilization • Reconnaissance of ground water in basin, including nutrients, pH, and stable isotopes (natural-^{18}O or natural-^{15}N) • Characterize upstream land use with GIS data layers, including fertilization and harvest history in watershed
Extensive	Previous objectives, with additional detail, plus Cumulative downstream effects (spatial and temporal), of forest fertilization on • Water quality, and • Aquatic biological systems	Basic level, plus: More intensive evaluation of cumulative effects on Little River, including multiple upstream land uses Evaluation of nutrient retention, transport, and fate, including percentage of fertilizer N in different compartments Relation of fertilization effects on stream biota with riparian condition More intensive evaluation of ground water and hyporheic processes Evaluation of effects of fertilization on higher food webs Evaluation of longer term effects from fertilization Experimental tracer of urea with ^{15}N in selected fertilization units	Level 2, and: • Expanded network of sites in Little River and Wolf Creek, and use of tracers (Little River only) for septic waste • Possible use of isotopically labeled urea-N in selected areas • Sample for ^{15}N (naturally occurring or artificially labeled) in algae, sediments, and biota from hyporheic zones, streambeds • Nutrient sampling during and immediately after fertilization, and measuring effective fertilization rate on ground • Establish piezometer network near selected stream reaches to determine flow paths, transient storage, and nutrient transformation/retention in hyporheic zones • Measure primary production in treatment and reference streams • Macroinvertebrate (and possibly aquatic amphibian) sampling in conjunction with algal sampling • In-depth GIS analysis, including influences of confounding upstream land uses, measuring and mapping riparian conditions • Nutrient and algal sampling in second summer following fertilization

Wolf Creek, (2) nutrient transport and retention, (3) the relative importance of riparian buffers and vegetation in modifying stream response to fertilization, (4) ground-water inputs, including regional flow and local, hyporheic transformations, (5) effect of altered nutrient regimes on higher trophic levels, (6) long-term (> 1 year) contributions of fertilizer-N to streams, and (7) possible use of urea with labelled-^{15}N.

Study elements for this comprehensive approach would involve sampling at more sites in Wolf Creek and in the Little River in order to evaluate possible septic or sewage inputs to Little River and reference sites outside of Wolf Creek subwatershed. It would also include an assessment of major ions, and possibly ^{18}O to evaluate source water based on potential geochemical differences among sites. To investigate the potential for long-term changes in nutrient regimes, sampling would be extended into fall and possibly an additional summer of the second year following fertilization. In addition to mass balances on nutrients and major ions, inputs from possible septic or sewage sources to the Little River could be traced by using either natural-^{15}N, caffeine, or other methods. Retention of nutrients in streams would be assessed by analyzing benthic and drifting algae for tissue-nutrient content and calculating the amount of N and P retained or transported in biomass. Community-level effects on algae would be determined by multivariate analysis of periphyton species data and by measuring metabolic rates prior to and following fertilization. Higher trophic levels would be investigated by sampling of macroinvertebrates in conjunction with algae in summers prior to and following fertilization to investigate effects on secondary consumers. If reconnaissance data indicate that natural-^{15}N will prove useful in following fertilizer N movement, or if urea with labelled-^{15}N is used, macroinvertebrates would also be sampled for ^{15}N levels. GIS data layers, including mapping of riparian areas and (if possible) private-timber-land management and fertilization history, would be generated. These data would be used to help define the importance of various influences on water quality at downstream stations. Depending on available resources this study would also include installation of test wells near selected stream sites to investigate ground-water exchange with streams and localized nutrient dynamics in hyporheic zones.

Together, these approaches would help to more definitively determine changes in algal communities among sites affected and not affected by fertilization, secondary effects on macroinvertebrates, and relations between nutrient concentrations (in water and algal tissues) and water-quality parameters. They would also provide a better opportunity to observe cumulative effects in downstream reaches, including Little River, and differentiate them from effects of other land uses. Ideally, models would eventually be constructed to evaluate the effects of fertilizer N inputs on stream biota and water quality; however, modeling periphytic systems is still relatively imprecise. Spreadsheet models of ground-water-N input and stream dynamics have been developed for intensively studied streams (Peterson et al., 2001), but they do not model primary production or its resulting affects on DO and pH. The Oregon Department of Environmental Quality (2000) recently developed a model for use in setting TMDLs in periphytic streams. This model predicts DO and pH as a function of nutrient concentration, and may work reasonably well for streams where point sources have been reduced, but has not yet been tested for systems with diffuse nutrient sources.

SUMMARY

Forests in the Pacific Northwest and elsewhere have long been fertilized to increase timber productivity, with over 120,000 acres per year being fertilized in the Pacific Northwest in the late 1980's. Recent (1990–98) fertilization levels in Oregon have averaged approximately 95,000 acres annually. A review of literature on water-quality effects from fertilization of forests with nitrogen indicates that applied nitrogen does indeed run off to streams, in amounts ranging from less than 1% to as much as 27% of applied nitrogen. The amount of applied nitrogen lost to streams depends on many factors, including the amount and form of fertilizer applied, timing of application (usually fall), weather during and after application, degree to which the application was able to avoid direct input to streams, width of riparian buffers, nitrogen status of soils in the watershed, hydrologic processes in the watershed (including ground-water residence time), and history of forestry or other land-use practices in the watershed. Invariably there have been high-concentration pulses of nitrogen, usually as urea (or total Kjeldahl nitrogen) and NH_3-N (ammonia-nitrogen), during runoff immediately following applications, with subsequent decreases in concentrations. Subsequent increases in NO_3-N (nitrate-nitrogen) concentrations can be more prolonged, often for the duration of the winter and spring. Summer NO_3-N concentrations are frequently low, often resembling background, but usually have been elevated during the following fall in streams draining treated watersheds.

Despite these increases, water-quality criteria for nutrients have almost always been met, except in rare instances such as where soils were already nitrogen saturated. However, water-quality criteria for nutrients are targeted towards human health (for NO_3-N) or aquatic toxicity (for NH_3-N), and are not set at levels relevant to ecologic processes in most forested aquatic ecosystems. Biological processes following fertilization have rarely been studied, and most were completed prior to the development of key concepts of nutrient processing and ecological dynamics in streams. In several cases, techniques were not sensitive to potential processes in the streams studied. Meanwhile, many forest streams continue to indicate breakdown of ecological systems, from eutrophication to potential food-web alterations and loss of sensitive species. Thus, key questions about the ecological effects of forest practices remain unresolved. For these reasons, new approaches to evaluation of forest management practices, such as fertilization, are necessary.

In Cascade streams of the Pacific Northwest, productivity in mountainous streams, like forests, is typically nitrogen limited. Increases in nitrogen inputs to streams can potentially increase primary production, and possibly alter successional patterns, community dynamics, and trophic structure of benthic communities. Nutrient inputs have long been linked to occurrences of nuisance algal growth in many streams, with secondary effects on water quality (DO and pH) from algal metabolism. These situations are increasingly frequent in forested systems.

Pathways for nitrogen input to streams from upland disturbances include direct runoff, groundwater inputs, and hyporheic flow. Instream pathways for nitrogen processing, besides classical transport, include hyporheic retention and processing by microbial communities, uptake by benthic algae, and downstream transport by boundary layer recycling or transport of sloughed, particulate forms of algae. All of these processes can be extremely efficient and represent significant portions of the nitrogen budget of a stream. Yet most are ignored by standard approaches to water sample collection. Thus, the actual amount of nitrogen entering streams and contributing to ecological processes from upland sources (such as fertilization) may have been underestimated in some previous studies.

The Little River watershed, in southwestern Oregon, has been designated as one of 10 Adaptive Management Areas (AMA's) under the President's Northwest Forest Plan. Forest land ownership in the watershed is predominantly Federal but private timberland also constitutes much of the watershed and is interspersed among many Federal tracts. Currently, water quality in the Little River during summers does not meet State standards for temperature or pH in some locations, and in many locations nuisance algal conditions are common. Nutrient concentrations are typically low and streams are generally nitrogen limited.

To accompany a proposed operational fertilization of Federal (Bureau of Land Management) timberlands in portions of the watershed, a multi-level framework for investigation of water quality and ecological processes is suggested. The studies would focus primarily on biological endpoints but also would include hydrologic components and nutrient-data collection to help understand ecological processes. The different levels of study would help, to varying degrees, define the effects, if any, of fertilizer-nutrient inputs on aquatic ecosystems and processes, relations between nutrient inputs, watershed characteristics, and water quality, and finally, downstream cumulative effects on both water quality and aquatic-biological systems.

REFERENCES CITED

Aber, J.D., Nadelhoffer, K.J., Steudler, P., and Melillo, J.M., 1989, Nitrogen saturation in northern forest ecosystems: Bioscience, v. 39, p. 378–386.

Adams, P.W. and Stack, W.R., 1989, Streamwater quality after logging in southwest Oregon: Portland, Oregon, Project completion report supplement number PNW 87–400, U.S. Forest Service, Pacific Northwest Research Station, 19 p.

American Public Health Association, 1998, Standard Methods for the Examination of Water and Wastewater (20th ed.): Washington, D.C., American Public Health Association, variously paged.

Anderson, C. W., and Carpenter, K. D., 1998, Water-quality and algal conditions in the North Umpqua River Basin, Oregon, 1992–95, and implications for resource management: U.S. Geological Survey

Water-Resources Investigations Report 98–4125, 78 p., 1 pl.

Beyers, R.J., 1970, A pH-carbon dioxide method for measuring aquatic primary productivity: Bulletin of the Georgia Academy of Science, v. 28, p. 55–68.

Biggs, B.J.F., Stevenson, R.J., and Lowe, R.L., 1998, A habitat matrix conceptual model for stream periphyton: Archiv fur Hydrobiologie, v. 143, no. 1, p. 21–56.

Binkley, D., and Reid, P., 1985, Long-term increases of nitrogen availability from fertilization of Douglas-fir: Canadian Journal of Forest Research, v. 15, p. 723–724.

Binkley, 1986, Forest nutrition management: New York, John Wiley and Sons, 300 p.

Binkley, D., and Brown, T.C., 1993a, Forest practices as nonpoint sources of pollution in North America: Water Resources Bulletin, v. 29, p. 729–740.

Binkley, D., and Brown, T., 1993b, Management impacts on water quality of forests and rangelands: United States Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station RM–239, 114 p.

Binkley, D., Burnham, H., and Allen, H.L., 1999, Water quality impacts of forest fertilization with nitrogen and phosphorus: Forest Ecology and Management, v. 121, p. 191.

Bisson, P.A., 1982, Annual fertilization and water quality—Final report: Weyerhauser Company Technical Report 050–5411–02, Centralia, Washington, 32 p.

Bisson, P.A., 1988, 1988 Weyerhauser forest fertilization/water quality monitoring studies—Forks Creek, Ryderwood, Silver Lake: Weyerhauser Company Technology Center, Tacoma, Washington, 26 p.

Bisson, P.A., Ice, G.G., Perrin, C.J., and Bilby, R.E., 1992, Effects of forest fertilization on water quality and aquatic resources in the Douglas-fir region, *in* Chappell, H., N., Weetman, G., F., and Miller, R., E., eds., Forest fertilization—Sustaining and improving nutrition and growth of western forests: Seattle, Washington, University of Washington, Institute of Forest Resources Contribution No. 73, p. 179–193.

Borchardt, M.A., 1996, Nutrients, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 183–227.

Bothwell, M.L., 1988, Growth rate responses of lotic periphyton diatoms to experimental phosphorus enrichment—The influence of temperature and light: Canadian Journal of Fisheries and Aquatic Sciences, v. 45, p. 261–270.

Bothwell, M.L., 1989, Phosphorus-limited growth dynamics of lotic periphyton diatom communities: Areal biomass and cellular growth rate responses: Canadian Journal of Fisheries and Aquatic Sciences, v. 46, p. 1293–1301.

Bothwell, M.L., 1992, Eutrophication of rivers by nutrients in treated kraft pulp mill effluent: Water Pollution Research Journal of Canada, v. 27, no. 3, p. 447–472.

Bott, T.L., Brock, J.T., Baattrup-Pedersen, A., Chambers, P.A., Dodds, W.K., Himbeault, K.T., Lawrence, J.R., Planas, D., Snyder, E., and Wolfaardt, G.M., 1997, An evaluation of techniques for measuring periphyton metabolism in chambers: Canadian Journal of Fisheries and Aquatic Sciences, v. 54, p. 715–725.

Boulton, A.J., Findlay, S., Marmonier, P., Stanley, E.H., and Valett, H.M., 1998, The functional significance of the hyporheic zone in streams and rivers: Annual Reviews of Ecology and Systematics, v. 29, p. 59–81.

Brown, G.W., Gahler, A.R., and Marston, R.B., 1973, Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range: Water Resources Research, v. 9, no. 5, p. 1450–1453.

Brown, T.C., and Binkley, D., 1994, Effect of management on water quality in North American Forests: U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station RM–248.

Carter, R., 1992, Diagnosis and interpretation of forest stand nutrition status, *in* Chappell, H.N., Weetman, G.F., and Miller, R.E., eds., Forest fertilization—Sustaining and improving nutrition and growth of western forests: Seattle, Washington, University of Washington, Institute of Forest Resources Contribution No. 73, p. 90–97.

Cattaneo, A., and Amireault, M.C., 1992, How artificial are artificial substrata for periphyton?: Journal of the North American Benthological Society, v. 11, no. 2, p. 244–256.

Chafiq, M., Gibert, J., and Claret, C., 1999, Interactions among sediments, organic matter, and microbial activity in the hyporheic zone of an intermittent stream: Canadian Journal of Fisheries and Aquatic Sciences, v. 56, p. 487–495.

Chappell, H.N., Weetman, G.F., and Miller, R.E., eds., 1992, Forest fertilization—Sustaining and improving nutrition and growth of western forests: Seattle, Washington, University of Washington, College of Forest Resources, Institute of Forest Resources, Contribution No. 73, 302 p.

Chestnut, T.M., and McDowell, W.H., 2000, C and N dynamics in the riparian and hyporheic zones of a tropical stream, Luquillo Mountains, Puerto Rico: Journal of the North American Benthological Society, v. 19, no. 2, p. 199–214.

Cirmo, C.P., and McDonnell, J.J., 1997, Linking the hydrologic and biogeochemical controls of nitrogen transport in near-stream zones of temperate forested catchments—A review: Journal of Hydrology, v. 199, p. 88–120.

Clark, G.M., Mueller, D.K., and Mast, M.A., 2000, Nutrient concentrations and yields in undeveloped stream basins of the United States: Journal of the American Water Resources Association, v. 36, no. 4, p. 849–860.

Cline, C., 1973, The effects of forest fertilization on the Tahuya River, Kitsap Peninsula, Washington: Olympia, Washington Department of Ecology, Report no. 74–2.

Cole, D.W., and Gessel, S. P., 1965, Movement of elements through a forest soil as influenced by tree removal and fertilizer additions, *in* Youngberg, C.T., ed., Forest-soil relationships in North America: Corvallis, Oregon, Oregon State University Press, p. 95–104.

Cole, D.W., 1979, Mineral cycling in forest ecosystems of the Pacific Northwest, *in* Gessel, S.P., Kenady, R.M., and Atkinson, W.A., ed., Proceedings of the Forest Fertilization Conference: Union, Washington, University of Washington, College of Forest Resources, Institute of Forest Resources, p. 29–36.

Correll, D.L., 1998, The role of phosphorus in the eutrophication of receiving waters—A review: Journal of Environmental Quality, v. 27, p. 261–266.

Craig, J.R., and Wollum, A.G.I., 1982, Ammonia volatilization and soil nitrogen changes after urea and ammonium nitrate fertilization of *Pinus taeda* L.: Journal of the Soil Science Society of America, v. 46, p. 409–414.

Dahm, C.N., Grimm, N.B., Marmonier, P., Valett, H.M., and Vervier, P., 1998, Nutrient dynamics at the interface between surface waters and ground waters: Freshwater Biology, v. 40, p. 427–451.

Denicola, D.M., 1996, Periphyton responses to temperature at different ecological levels, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 149–183.

Dillon, P.J., and Kirchner, W.B., 1975, The effects of geology and land use on the export of phosphorus from watersheds: Water Research, v. 9, p. 135–148.

Dodds, W.K., Jones, J.R., and Welch, E.B., 1998, Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus: Water Research, v. 32, no. 5, p. 1455–1462.

Dodds, W.K., Smith, V.H., and Zander, B., 1997, Developing nutrient targets to control benthic chlorophyll levels in streams: Water Research, v. 31, no. 7, p. 1738–1750.

Downs, M.R., Nadelhoffer, K.J., Melillo, J.M., and Aber, J.D., 1996, Immobilization of a ^{15}N-labeled nitrate addition by decomposing forest litter: Oecologia, v. 105, p. 141–150.

Edwards, P.J., Kochenderfer, J.N., and Seegrist, D.W., 1991, Effects of forest fertilization on stream water chemistry in the Appalachians: Water Resources Bulletin, v. 27, n. 2, p. 265–274.

Fairchild, G.W., Lowe, R.L., and Richardson, W.B., 1985, Algal periphyton growth on nutrient-diffusing substrates—An *in situ* bioassay: Ecology, v. 66, no. 2, p. 465–472.

Fenn, M.E., Poth, M.A., Aber, J.D., Baron, J.S., Bormann, B.T., Johnson, D.W., Lemly, A.D., McNulty, S.G., Ryan, D.F., and Stottlemyer, R., 1998, Nitrogen excesses in North American Ecosystems: predisposing factors, ecosystem responses, and management strategies: Ecological Applications, v. 8, no. 3, p. 706–733.

Fishman, M.J., 1993, Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory—Determination of inorganic and organic constituents in water and fluvial sediments: U.S. Geological Survey Open File Report 93–125, 217 p.

Floener, L., and Bothe, H., 1980, Nitrogen fixation in Rhopalodia gibba, a diatom containing blue-greenish inclusions symbiotically, *in* Schwemmler, W., and Schenk, H.E.A., eds., Endocytobiology, endosymbiosis, and cell biology: Berlin, Walter de Gruyter and Company, p. 541–552.

Fredriksen, R.L., Moore, D.G., and Norris, L.A., 1975, The impact of timber harvest, fertilization, and herbicide treatment on streamwater quality in western Oregon and Washington, *in* Bernier, B., and Winget, C., eds., Forest Soils and Forest Land Management– Proceedings of the Fourth North

American Forest Soils Conference: Quebec, Canada, Laval University Press, p. 283–313.

Fry, B., Jones, D.E., Kling, G.W., McKane, R.B., Nadelhoffer, K.J., and Peterson, B.J., 1995, Adding ^{15}N tracers to Ecosystem Experiments, *in* Wada, E., ed., Stable isotopes in the biosphere: Kyoto, Kyoto University Press, p. 171–191.

Gessel, S.P., Kenady, R.M., and Atkinson, W.A. (eds.), 1979, Proceedings of the Forest Fertilization Conference: Seattle, Washington, University of Washington, College of Forest Resources, Institute of Forest Resources, Contribution no. 40, 275 p.

Göthe, L., Söderberg, H., Sjölander, E., and Nohrstedt, H.-O., 1993, Effects on water chemistry, benthic invertebrates, and brown trout following forest fertilization in Central Sweden: Scandinavian Journal of Forest Research, v. 8, p. 81–93.

Gregory, S.V., Lamberti, G.A., Erman, D.C., Koski, K.V., Murphy, M.L., and Sedell, J.R., 1987, Influence of forest practices on aquatic production, *in* Salo, E.O., and Cundy, T.W., eds., Streamside management—Forestry and fishery interactions: Seattle, Washington, College of Forest Resources, University of Washington, p. 233–255.

Groman, W.A., 1972, Forest fertilization—A state-of-the-art review and description of environmental effects: Northwest Water Lab, U.S. Environmental Protection Agency Technical Series EPA–R2–016.

Guasch, H., Armengol, J., Marti, E., and Sabater, S., 1998, Diurnal variation in dissolved oxygen and carbon dioxide in two low-order streams: Water Research, v. 32, no. 4, p. 1067–1074.

Haase, D., and Rose, R., eds., 1997, Symposium proceedings—Forest seedling nutrition from the nursery to the field: Corvallis, Oregon State University, Forest Sciences Laboratory, 161 p.

Harriman, R., 1978, Nutrient leaching from fertilized forest watersheds in Scotland: Journal of Applied Ecology, v. 15, p. 933–942.

Harvey, C.J., Peterson, B.J., Bowden, W.B., Hershey, A.E., Miller, M.M., Deegan, L.A., and Finlay, J.C., 1998, Biological responses to fertilization of Oksrukuyik Creek, a tundra stream: Journal of the North American Benthological Society, v. 17, no. 2, p. 190–209.

Harvey, J.W., and Bencala, K.E., 1993, The effect of streambed topography on surface-subsurface water exchange in mountain catchments: Water Resources Research, v. 29, no. 1, p. 89–98.

Hedin, L.O., Armesto, J.J., and Johnson, A.H., 1995, Patterns of nutrient loss from unpolluted, old-growth temperate forests—Evaluation of biogeochemical theory: Ecology, v. 76, no. 2, p. 493–509.

Hellebust, J.A., and Lewin, J., 1977, Heterotrophic nutrition, *in* Werner, D., ed., The biology of diatoms: Oxford, Scientific Publications, p. 169–197.

Helvey, J.D., Kochenderfer, J.N., and Edwards, P.J., 1989, Effects of forest fertilization on selected ion concentrations in central Appalachian streams, *in* Proceedings of the Central Hardwood Conference: Southern Illinois University, Carbondale, Illinois, North Central Forest Experiment Station, U.S. Department of Agriculture, Forest Service, p. 278–282.

Hetherington, E.D., 1985, Streamflow nitrogen loss following forest fertilization in a southern Vancouver Island watershed: Canadian Journal of Fisheries and Aquatic Sciences, v. 15, p. 34–41.

Hillebrand, H., and Sommer, U., 1999, The nutrient stoichiometry of benthic microalgal growth—Redfield proportions are optimal: Limnology and Oceanography, v. 44, no. 2, p. 440–446.

Holmes, R.M., Aminot, A., Kerouel, R., Hooker, B.A., and Peterson, B.J., 1999, A simple and precise method for measuring ammonium in marine and freshwater ecosystems: Canadian Journal of Fisheries and Aquatic Sciences, v. 56, p. 1801–1808.

Johnson, D.W., 1992, Nitrogen retention in forest soils: Journal of Environmental Quality, v. 21, no. 1, p. 1–12.

Jordan, M.J., Nadelhoffer, K.J., and Fry, B., 1997, Nitrogen cycling in forest and grass ecosystems irrigated with ^{15}N-enriched wastewater: Ecological Applications, v. 7, no. 3, p. 864–881.

Kahl, J.S., Norton, S.A., Fernandez, I.J., Nadelhoffer, K.J., Driscoll, C.T., and Aber, J.D., 1993, Experimental inducement of nitrogen saturation at the watershed scale: Environmental Science and Technology, v. 27, p. 565–568.

Kendall, C., and McDonnell, J.J., 1998, Isotope tracers in catchment hydrology: Amsterdam, Elsevier, 839 p.

Kim, B.A., Jackman, A.P., and Triska, F.J., 1992, Modeling biotic uptake by periphyton and transient hyporheic storage of nitrate in a natural stream: Water Resources Research, v. 28, no. 10, p. 2743–2752.

Klinka, K., 1991, Approach to evaluating site characteristics for forest fertilization, *in* Lousier, J.D., Brix, R., Brockley, R., Carter, R., and Marshall, V.G., eds., 1991, Improving forest fertilization decision-making in British Columbia—Proceedings of a forest fertilization workshop, March 1988: Victoria, B.C., Crown Publications, Inc., p. 153–161.

Klock, G.O., 1971, Streamflow nitrogen loss following forest erosion control fertilization: U.S. Forest Service Pacific Northwest Forest and Range Experiment Station PNW–169.

Lajtha, K, and Michener, R. H., 1994, Stable isotopes in ecology and environmental science: London, Blackwell Scientific Publications, 336 p.

Lousier, J.D., Brix, R., Brockley, R., Carter, R., and Marshall, V.G., eds., 1991, Improving forest fertilization decision-making in British Columbia—Proceedings of a forest fertilization workshop, March 1988: Victoria, B.C., Crown Publications, Inc., 313 p.

Lowe, R.L., and Pan, Y., 1996, Benthic algal communities as biological monitors, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 609–639.

Lundberg, C., 1996, Effects of grazing and nitrogen enrichment on the taxonomic structure of periphyton assemblages in lotic ecosystems: Corvallis, Oregon State University, Master's thesis, 120 p.

Malueg, K.W., Powers, C.F., and Krawczyk, D.F., 1972, Effects of aerial forest fertilization with urea pellets on nitrogen levels in a mountain stream: Northwest Science v. 46, n. 1, p. 52–58.

Marshall, V.G., 1991, Mechanisms of response to fertilization I. Fate of nitrogenous fertilizers, *in* Lousier, J.D., Brix, R., Brockley, R., Carter, R., and Marshall, V.G., eds., 1991, Improving forest fertilization decision-making in British Columbia—Proceedings of a forest fertilization workshop, March 1988: Victoria, B.C., Crown Publications, Inc., p. 51–75.

——— 1986, Environmental fate of nitrogenous fertilizers, *in* Brockley, R.P., ed., Proceedings of the Interior Forest fertilization workshop: Victoria, B.C., Ministry of Forest Lands, p. 149–168.

Marshall, V.G., and McMullan, E.E., 1976, Balance sheet of recovered ^{15}N-labelled urea in a pot trial with *Psuedotsuga menziesii*: Canadian Journal of Soil Science, v. 56, p. 311–314.

Marzolf, E.R., Mulholland, P.J., and Steinman, A.D., 1994, Improvements to the diurnal upstream-downstream dissolved oxygen change technique for determining whole-stream metabolism in small streams: Canadian Journal of Fisheries and Aquatic Sciences, v. 51, no. 1591–1599.

Marzolf, E.R., Mulholland, P.J., and Steinman, A.D., 1998, Reply—Improvements to the diurnal upstream-downstream dissolved oxygen change technique for determining whole-stream metabolism in small streams: Canadian Journal of Fisheries and Aquatic Sciences, v. 55, p. 1786–1787.

Matzner, F., Khanna, P.K., Meiwes, M.J., and Ulrich, B., 1983, Effects of fertilization on the fluxes of chemical elements through different forest ecosystems: Plant and Soil, v. 74, no. 343–358.

McCammon, B., 1994, Recommended watershed terminology: Watershed Management Council Newsletter, v. 6, no. 2, http://www.watershed.org/wmc/news/www_docs/fall_94.html, accessed January 29, 2001.

McFarland, W.D., 1983, A description of aquifer units in western Oregon: U.S. Geological Survey Open-File Report 82–165, 35 p.

McIntire, D.C., Gregory, S.V., Steinman, A.D., and Lamberti, G.A., 1996, Modeling benthic algal communities—An example from stream ecology, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 670–704.

Meehan, W.R., Lotspeich, F.B., and Mueller, E.W., 1975, Effects of forest fertilization on two southeast Alaska streams: Journal of Environmental Quality, v. 4, p. 50–55.

Miegroet, H.V., and Cole, D.W., 1988, Influence of nitrogen-fixing Alder on acidification and cation leaching in a forest soil, *in* Cole, D.W., and Gessel, S.P., eds., Forest site evaluation and long-term productivity: Seattle, University of Washington Press, Institute of Forest Resources, Contribution no. 63, p. 113–124.

Miegroet, H.V., Johnson, D.W., and Cole, D.W., 1990, Soil nitrification as affected by N fertility and changes in forest floor C/N ration in four forest soils: Canadian Journal of Forest Research, v. 20, p. 1012–1019.

Mika, P.G., Moore, J.A., Brockley, R.P., and Powers, R.F., 1992, Fertilization response by Interior Forests: When, where, and how much?, *in* Chappell, H., N., Weetman, G.F., and Miller, R.E., eds., Forest fertilization—Sustaining and improving nutrition and growth of western forests: Seattle, Washington, University of Washington, Institute of Forest Resources Contribution No. 73, p. 127–142.

Miller, H.G., 1986, Long term effects of application of nitrogen fertilizers on forest sites, *in* Proceedings of the Proceedings of the International Union of Forestry Research Organizations (IUFRO): Rome, p. 496–506.

Miller, R.E., and Fight, R.D., 1979, Fertilizing Douglas-fir forests: U.S. Forest Service Pacific Northwest Forest and Range Experiment Station PNW–83, 29 p.

Mitchell, A.K., Barclay, H.J., Brix, H., Pollard, D.F.W., Benton, R., and deJong, R., 1996, Biomass and nutrient element dynamics in Douglas-fir—Effects of thinning and nitrogen fertilization over 18 years: Canadian Journal of Forest Research, v. 26, p. 376–388.

Moffatt, R.L., Wellman, R.E., and Gordon, J.M., 1990, Statistical summaries of streamflow data in Oregon, Volume 1—Monthly and annual streamflow, and flow duration values: U.S. Geological Survey Open-file Report 90–118, 413 p.

Moldan, F., and Wright, R.F., 1998, Changes in runoff chemistry after five years of N addition to a forested catchment at Gardsjohn, Sweden: Forest Ecology and Management, v. 101, no. 1–3, p. 187–197.

Moore, D.G., 1971, Fertilization and water quality, *in* Western reforestation—Proceedings of the Annual Meeting of the Western Reforestation Coordinating Committee: Western Forest Conservation Association, p. 28–31.

Moore, D.G., 1975, Impact of forest fertilization on water quality in the Douglas-fir region—A summary of monitoring studies: New York, Proceedings of the 1974 National Convention, Society of American Foresters, p. 209–219.

Mulholland, P.J., 1992, Regulation of nutrient concentrations in a temperate forest stream—Roles of upland, riparian, and instream processes: Limnology and Oceanography, v. 37, no. 7, p. 1512–1526.

Mulholland, P.J., 1996, Role in nutrient cycling in streams, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 609–639.

Mulholland, P.J., Marzolf, E.R., Webster, J.R., Hart, D.R., and Hendricks, S.P., 1997, Evidence that hyporheic zones increase heterotrophic metabolism and phosphorus uptake in forest streams: Limnology and Oceanography, v. 42, no. 3, p. 443–451.

Mulholland, P.J., Marzolf, E.R., Webster, J.R., Hart, D.R., and Hendricks, S.P., 1999, Evidence that hyporheic zones increase heterotrophic metabolism and phosphorus uptake in forest streams—Errata: Limnology and Oceanography, v. 42, no. 3, p. 443–451.

Mulholland, P.J., and Rosemond, A.D., 1992, Periphyton response to longitudinal nutrient depletion in a woodland stream—Evidence of upstream-downstream linkage: Journal of the North American Benthological Society, v. 11, no. 4, p. 405–419.

Naegeli, M.W., and Uehlinger, U., 1997, Contribution of the hyporheic zone to ecosystem metabolism in a prealpine gravel-bed river: Journal of the North American Benthological Society, v. 16, no. 4, p. 794–804.

Nason, G.E., and Myrold, D.D., 1992, Nitrogen Fertilizers—Fates and environmental effects in forests, *in* Chappell, H.N., Weetman, G.F., and Miller, R.E., eds., Forest fertilization—Sustaining and improving nutrition and growth of western forests: Seattle, Washington, University of Washington, Institute of Forest Resources Contribution No. 73, p. 67–81.

Nason, G.E., Pluth, D.J., and McGill, W.B., 1988, Volatilization and foliar recapture of ammonia following spring and fall application of nitrogen-15 urea to a Douglas-fir ecosystem: Soil Science Society of America Journal, v. 52, p. 821–828.

National Council of the Paper Industry for Air and Stream Improvement (NCASI), 1999, Water quality effects of forest fertilization: Research Triangle Park, North Carolina, Technical Bulletin No. 782, 53 p.

Newbold, J.D., Elwood, J.W., O'Neill, R.V., and Van Winkle, W., 1981, Measuring nutrient spiralling in streams: Canadian Journal of Fisheries and Aquatic Sciences, v. 38, p. 860–863.

Norris, L.A., Lorz, H.W., and Gregory, S.V., 1991, Forest chemicals, *in* Meehan, W.R., ed., Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats: Bethesda, Maryland, American Fisheries Society, p. 207–296.

Norton, S.A., Kahl, J.S., Fernandez, I.J., Rustad, L.E., Scofield, J.P., and Haines, T.A., 1994, Response of the West Bear Brook Watershed, Maine, USA, to the addition of $(NH_4)_2SO_4$: 3-year results: Forest Ecology and Management, v. 68, p. 61–73.

Ochtere-Boateng, J., 1979, Reaction of nitrogen fertilizers in forest soils, *in* Gessel, S.P., Kenady, R.M., and Atkinson, W.A. (eds.), Proceedings of the Forest Fertilization Conference, Union, Washington, University of Washington, College of Forest Resources, Institute of Forest Resources, Contribution No. 40, p. 37–47.

Oregon Department of Environmental Quality, 1999, Final 1998 303(d) database: http://waterquality.deq.state.or.us/WQLData/SubBasinList98.asp, accessed August 26, 1999.

Oregon Department of Environmental Quality, 2000, Upper Grande Ronde TMDL, Appendix B—Periphyton Analysis: http://waterquality.deq.state.or.us/wq/TMDLs/TMDLs.htm, accessed April 10, 2000.

Oregon Department of Forestry, 1999, Annual reports 1990–98: http://www.odf.state.or.us/annual_reports/ANNUAL_%20REPORTS.htm, accessed January 5, 2000.

Owenby, J.R., and Ezell, D.S., 1992, Monthly station normals of temperature, precipitation, and heating and cooling degree days, 1961–1990, Oregon: Asheville, North Carolina, National Oceanic and Atmospheric Administration, 16 p.

Pan, Y., Stevenson, R.J., Hill, B.H., Herlihy, A.T., and Collins, G.B., 1996, Using diatoms as indicators of ecological conditions in lotic systems—A regional assessment: Journal of the North American Benthological Society, v. 15, no. 4, p. 481–495.

Parkhill, K.L., and Gulliver, J.S., 1998, Application of photorespiration concepts to whole stream productivity: Hydrobiologia, v. 389, p. 7–19.

Patton, C.J., and Truitt, E.P., 2000, Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory—Determination of ammonium plus organic nitrogen by a Kjeldahl digestion method and an automated photometric finish that includes digest cleanup by gas diffusion: U.S. Geological Survey Open-File Report 00–170, 31 p.

Perrin, C.J., Shortreed, K.S., and Stockner, J.G., 1984, An integration of forest and lake fertilization—Transport and transformations of fertilizer elements: Canadian Journal of Fisheries and Aquatic Sciences, v. 41, p. 253–262.

Peterson, B.J., Wolheim, W.M., Mulholland, P.J., Webster, J.R., Meyer, J.L., Tank, J.L., Marti, E., Bowden, W.B., Valett, H.M., Hershey, A.E., McDowell, W.H., Dodds, W.K., Hamilton, S.K., Gregory, S.V., and Morrall, D.D., 2001, Control of nitrogen export from watersheds by headwater streams: Science, v. 292, p. 86–90.

Peterson, C.A., 1996, Response of benthic algal communities to natural physical disturbance, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 375–402.

Powell, M.A., 1995, Report on pH in the Jackson Creek and Little River drainage basins of the Umpqua National Forest: Roseburg, Oregon, U.S. Forest Service, Umpqua National Forest, Draft report to the U.S. Forest Service, [variously paged].

——— 1998, Umpqua National Forest Water Quality Study, 1998: Roseburg, Oregon, U.S. U.S.Forest Service, 13 p., plus maps, tables, and figures.

Prescott, C.E., Kischuk, B.E., and Weetman, G.F., 1995, Long-term effects of repeated N fertilization and straw application in a jack pine forest; 3, Nitrogen availability in the forest floor: Canadian Journal of Forest Research, v. 25, p. 1991–1996.

Preston, C.M., Marshall, V.G., and McCullough, K., 1990, Fate of ^{15}N-labelled fertilizer applied on snow at two forest sites in British Columbia: Canadian Journal of Forest Research, v. 20, p. 1583–1592.

Pringle, C.M., and Triska, F.J., 1996, Effects of nutrient enrichment on periphyton, *in* Hauer, F.R., and Lamberti, G.A., eds., Methods in Stream Ecology: San Diego, Academic Press, p. 607–623.

Rantz, S.E., and others, 1982, Measurement and computation of streamflow: U.S. Geological Survey Water-Supply Paper 2175, v. 1 and 2., 631 p.

Russel, D.A., 1979, Forms of nitrogen fertilizers, *in* Gessel, S.P., Kenady, R.M., and Atkinson, W.A. (eds.), 1979, Proceedings of the Forest Fertilization Conference, Union, Washington: University of Washington, College of Forest Resources, Institute of Forest Resources, Contribution No. 40, p. 179–186.

Salvia, M., Iffly, J.F., Borght, P.V., Sary, M., and Hoffman, L., 1999, Application of the “snapshot” methodology to a basin-wide analysis of phosphorus and nitrogen at stable low flow: Hydrobiologia, v. 410, p. 97–102.

Seaber, P., Kapinos, F.P., and Knapp, G., 1987, Hydrologic unit maps: U.S. Geological Survey Water-Supply Paper 2294, 63 p.

Sollins, P., Grier, C.C., McCorison, F.M., Cromack, K., and Fogel, R., 1980, The internal element cycles of an old-growth Douglas-fir ecosystem in western Oregon: Ecological Monographs, v. 50, no. 3, p. 261–285.

Sollins, P., and McCorison, F.M., 1981, Nitrogen and carbon solution chemistry of an old-growth coniferous forest watershed before and after cutting: Water Resources Research, v. 17, no. 5, p. 1409–1418.

Stanford, J.A., and Ward, J.V., 1993, An ecosystem perspective of alluvial rivers—Connectivity and the hyporheic corridor: Journal of the North American Benthological Society, v. 12, no. 1, p. 48–60.

Stay, F.S, Malueg, K.W., Austin, R.E., Crouse, M.R., Katko, A., and Dominguez, S.E., 1978, Ecological effects of forest fertilization with urea on small western Cascade streams of Oregon, U.S.A.: Verhandlungen Internationale Vereinigung für Theoretische und Angewandte Limnologie, v. 20, p. 1347–1358.

Stay, F.S., Katko, A., Malueg, K.W., Crouse, M.R., Dominguez, S.E., and Austin, R.E., 1979, Ecological effects of forest fertilization on major biological components of small Cascade streams, Oregon: Environmental Protection Agency EPA–600/3–79–099, 18 p.

Steinman, A.D., 1996, Effects of grazers on freshwater benthic algae, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 341–374.

Stevenson, R.J., 1996a, An introduction to algal ecology in freshwater benthic habitats, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 3–30.

Stevenson, R.J., 1996b, The stimulation and drag of current, *in* Stevenson, J.R., Bothwell, M.L., and Lowe, R.L., eds., Algal ecology—Freshwater benthic ecosystems: San Diego, California, Academic Press, Inc., p. 321–340.

Stoddard, J.L., 1994, Long-term changes in watershed retention of nitrogen: its causes and consequences., *in* Baker, L.A., ed., Environmental chemistry of lakes and reservoirs—Advances in Chemistry: Washington, DC, American Chemical Society, p. 223–284.

Storey, R.G., Fulthorpe, R.R., and Williams, D.D., 1999, Perspectives and predictions on the microbial ecology of the hyporheic zone: Freshwater Biology, v. 41, p. 119–130.

Tamm, C. O., Holmen, H., Popovic, B., and Wiklander, G., 1974, Leaching of plant nutrients from soils as a consequence of forestry operations: Ambio, v. 3, no.6., p. 211–221

Tank, J.L., and Webster, J.R., 1998, Interaction of substrate and nutrient availability on wood biofilm processes in streams: Ecology, v. 79, no. 6, p. 2168–2179.

Tanner, D.Q., and Anderson, C.W., 1996, Assessment of water quality, nutrients, algal productivity, and management alternatives for low-flow conditions, South Umpqua River Basin, Oregon, 1990–92: U.S. Geological Survey Water-Resources Investigations Report 96–4082, 71 p.

Teal, J.M., and Kanwisher, J., 1966, The use of pCO_2 for the calculation of biological production, with examples from waters off Massachusetts: Journal of Marine Research, v. 24, no. 1, p. 4–14.

Tiedemann, A.R., Helvey, J.D., and Anderson, T.D., 1978, Stream chemistry and watershed nutrient economy following wildfire and fertilization in eastern Washington: Journal of Environmental Quality, v. 7, no. 4, p. 580–588.

Tiedemann, A.R., Quigley, T.M., and Anderson, T.D., 1988, Effects of timber harvest on stream chemistry and dissolved nutrient losses in northeast Oregon: Forest Science, v. 34, no. 2, p. 344–358.

Triska, F.J., Kennedy, V.A., Avanzino, R.J., and Reilly, N.B., 1983, Effect of simulated canopy cover on regulation of nitrate uptake and primary production by natural periphyton communities, *in* Fontain, T.D., and Bartell, S.M., eds., Dynamics of lotic ecosystems: Ann Arbor, Michigan, Ann Arbor Science Publishers., p. 129–159.

Triska, F.J., Sedell, J.R., Cromack, K.J., Gregory, S.V., and McCorison, F.M., 1984, Nitrogen budget for a small coniferous forest stream: Ecological Monographs, v. 54, no. 1, p. 119–140.

Udy, J.W., and Dennison, W.C., 1997, Growth and physiological responses of three seagrass species to elevated sediment nutrients in Moreton Bay, Australia: Journal of Experimental Marine Biology and Ecology, v. 217, no. 1997, p. 253–277.

U.S. Environmental Protection Agency, 1986, Quality criteria for water, 1986: The Gold Book, EPA 440/5–86–001

——— 2000a, National strategy for the development of regional nutrient criteria: EPA 822–B–00–002, http://www.epa.gov/ostwater/standards/nutsi.html, accessed October 21, 1999.

——— 2000b, Ambient water quality criteria recommendations—Information supporting the development of State and Tribal nutrient criteria—Rivers and streams in Nutrient Ecoregion II: EPA 822–B–00–015, variously paged, http://www.epa.gov/water science/criteria/nutrients/ecoregions/rivers, accessed February 6, 2001.

U.S. Forest Service and Bureau of Land Management, 1994, Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the ranges of the Northern Spotted Owl: Portland, Oregon, U.S. Forest Service and Bureau of Land Management, April 1994, 74 p.

——— 1995, Little River watershed analysis, version 1.1: Roseburg, Oregon, U.S. Forest Service, North Umpqua Ranger District, [variously paged].

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.A., Schlesinger, W.H., and Tillman, G.D., 1997, Human alteration of the global nitrogen cycle—Causes and consequences: Issues in Ecology, v. 1, 16 p.

Welch, E.B., ed., 1992, Ecological effects of wastewater—Applied limnology and pollutant effects, 2nd ed.: London, Chapman and Hall, 425 p.

Wetzel, R.G., 1983, Limnology, 2d ed.: Philadelphia, Saunders College Publishing. 858 p.

Wilkison, D.H., Blevins, D.W., and Silva, S.R., 2000, Use of isotopically labelled fertilizer to trace nitrogen fertilizer contributions to surface, soil, and ground water: Journal of Environmental Hydrology, v. 8, p. 2–16.

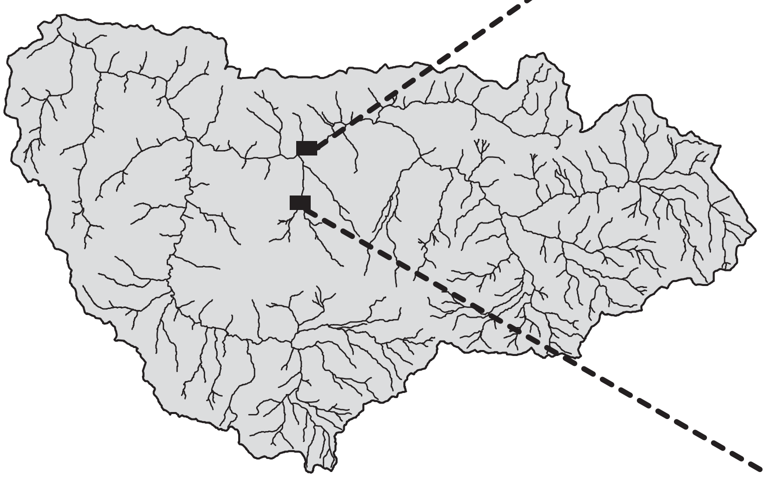
Woessner, W.A., 2000, Stream and fluvial plain ground water interactions—Rescaling hydrogeologic thought: Ground Water, v. 38, no. 3, p. 423–429.

Wondzell, S.M., and Swanson, F.J., 1996, Seasonal and storm dynamics of the hyporheic zone of a 4th order mountain stream—II—Nitrogen cycling: Journal of the North American Benthological Society, v. 15, no. 1, p. 20–34.

U.S. Department of the Interior
U.S. Geological Survey

Ecological Effects on Streams from Forest Fertilization—Literature Review and Conceptual Framework for Future Study in the Western Cascades

Water-Resources Investigations Report 01–4047



Prepared in cooperation with
the BUREAU OF LAND MANAGEMENT

USGS
science for a changing world

Cover photographs:

Upper: Little River above Wolf Creek *(photograph by John C. Risley, U.S. Geological Survey)*.
Lower: West Fork Wolf Creek *(photograph by Chauncey W. Anderson, U.S. Geological Survey)*.