Lisa Warr - NOAA Federal <lisa.s.warr@noaa.gov>

Fwd: EPA/NOAA proposed disapproval of Oregon's Coast Nonpoint Pollution Control Program

1 message

Joelle Gore - NOAA Federal <joelle.gore@noaa.gov> Fri, Mar 21, 2014 at 8:31 AM
To: Allison Castellan - NOAA Federal <allison.castellan@noaa.gov>, Lisa Warr - NOAA Federal <lisa.s.warr@noaa.gov>

---------- Forwarded message ----------
[redacted]
Date: Thu, Mar 20, 2014 at 11:31 PM
Subject: EPA/NOAA proposed disapproval of Oregon's Coast Nonpoint Pollution Control Program
To: joelle.gore@noaa.gov
[redacted]

March 20, 2014

Joelle Gore, Acting Chief
Coastal Programs Division (N/ORM3)
Office of Ocean and Coastal Resource Management
National Ocean Service, NOAA
1305 East-West Highway
Silver Spring MD 20910
joelle.gore@noaa.gov

Dear Ms. Gore,

These are comments regarding the EPA/NOAA proposed disapproval of Oregon's Coast Nonpoint Pollution Control Program.

We have lived on an original homestead in the Lake Creek, Siuslaw River drainage in the Oregon Coast Range since 1985. Lake Creek flows through our property on it's way to the Pacific Ocean. We have deployed Polar Organic Chemical Integrated Samplers in our local streams and detected herbicides that we know are used widely in this area for forestry site preparation and conifer release. We and others have had our urine tested and found Atrazine and 2,4-D. We own an organic farm and never use pesticides or chemical fertilizers.

In support of our belief that the State of Oregon does not protect nature as it should we would like to incorporate the following documents by reference:

Comments about Volatilization Drift to the Environmental Protection Agency Scientific Advisory Panel
http://www.gaiavisions.org/20140320-EPA-NOAA-NonpointPollution/20091130-Volatilization.pdf

Ecological Effects on Streams from Forest Fertilization
http://or.water.usgs.gov/pubs/WRIR01-4047/wri014047.pdf

Diminishing Returns: Salmon Decline and Pesticides
http://www.pcffa.org/salpest.pdf

1991 Wroncy v. Bureau of Land Management - Planned aerial fertilizer application
http://www.leagle.com/decision/19912323777FSupp1546_12097.xml/WRONCY%20v.%20BUREAU%20OF%20LAND%20MANAGEMENT

2003 Five Rivers Gypsy Moth Spray Program
http://www.ghdigital.com/jan/5R/index.html

2003 Willamette Industries Aerial Fertilizer Application
http://www.gaiavisions.org/20140320-EPA-NOAA-NonpointPollution/20030109-WillametteAerialFertilizer.pdf

Herbicide spray around Mapleton School, Mapleton OR
http://www.forestlanddwellers.org/Schools/Mapleton/Presentation-20070310/MapletonQ2-20070310.pdf

Herbicide spray around Triangle Lake School, Blachly OR
http://www.forestlanddwellers.org/Schools/TriangleLake/Presentation-20070319/TriangleLakeQ2-20070319.pdf

Herbicide spray in the Siuslaw watershed, coastal Oregon
http://www.forestlanddwellers.org/Maps/SiuslawWatershed/SiuslawPresentationQ2-20070404.pdf

Herbicide spray in the Siletz watershed, coastal Oregon
http://www.forestlanddwellers.org/Maps/SiletzWatershed/SiletzQ2-20070417.pdf

Photos of wetland area recently sprayed near headwaters of South Fork Alsea River
http://www.gaiavisions.org/20140320-EPA-NOAA-NonpointPollution/20140316-SouthForkAlsea.pdf

Oregon Department of Forestry Notification for South Fork Alsea spray operation
http://www.gaiavisions.org/notices/ODF/Paper/2014/2014-781-00194-Weyco-AlseaSouthFork.pdf

We need pesticide free buffers around schools
http://www.gaiavisions.org/20140320-EPA-NOAA-NonpointPollution/20090210-TestimonySB20-JW-final.pdf

Thank you for your consideration!

[redacted]

--
Joelle Gore
Stewardship Division, Acting Chief
OCRM/CSC
1305 East-West Highway
SSMC4, Room 11110
Silver Spring, MD 20910
Direct: 301-563-1177
Fax: 301-713-4370
www.coastalmanagement.noaa.gov

Comments about Volatilization Drift
to the Environmental Protection Agency Scientific Advisory Panel

Submitted by [redacted]

Accounts of Volatilization Drift and their negative impacts:

August 21, 2009: Oregon Forest Management Services applied Chopper manufactured by BASF, EPA No. 241-296 (imazapyr, active ingredient) plus Methylated Seed Oil foliar by back pack sprayers for Weyerhaeuser Company on steep clearcut forestland in the Coast Range of Oregon within Lane County adjacent to protected Coastal Coho Salmon streams (Congdon Creek and tributaries flowing into Lake Creek and then into the Siuslaw River).

Below is a picture of the Oregon Forest Management Services crew after they finished spraying the unit on August 21, 2009 about two air miles from my organic farm. (Photos by Gary Hale).

This type of application normally does not cause the amount of drift that an aerial application would, however, both kinds of applications do cause significant volatilization drift. Because of the steepness of the slopes treated and the herbicide/adjuvants used, there is noticeable vapor movement uphill with the warming air during the daytime, and downhill movement with the cooling air in the evening. The wind carries these vapors for miles, and the vaporization of these chemicals lasts for days, weeks, and even months.

The photo below shows how steep this unit is. The diurnal movements of air transport the vapors for a great distance from the sprayed units for a long time after the initial application of the pesticide or herbicide and adjuvant mixtures. Almost all the homes and farms are located in the bottom land in the valleys. The town of Horton was inhabited over 100 years ago. Our farm is the original homestead of Samuel Horton, one of the founding families of the town.

The following photo was taken of the sprayed unit after the herbicide was sprayed on the trees (mostly Big Leaf Maple) some of which were 15 or more feet tall. Spraying vegetation that tall with back pack sprayers would have increased the chance of drift during application.

Congdon Creek is the large fish-bearing stream below the unit that was sprayed. It is a prime spawning stream for Coho, Chinook and Stealhead. Congdon Creek flows into Lake Creek and then joins the Siuslaw River many miles downstream. The 1947 irrigation water right for our organic farm is around 3 miles downstream from the treated unit. The picture below is of Congdon Creek, taken from Majors Creek Bridge on the day of the spray. This part of the stream is prime spawning grounds for endangered salmon.

Not only did we receive drift from the original ground application, but we also received volatilization drift for weeks afterward. Then following rain, the contamination of our legal registered water right for irrigation water was evidenced by damage to the rows of crops watered by drip lines supplied with the river water.

The drift from vapors made it very difficult to work in my fields for any length of time because I quickly became ill (headaches, achiness, muscle aches, breathing problems and arrhythmia, etc.). My farm work fell behind schedule and I was never able to catch up for the season. Our farm cats, and dog also suffered from the vapors. My son was affected also. My husband was able to work inside with fewer effects because of a very expensive air filter we run in the house. But outside work remained difficult during this time.

After about one month, we went up to the public road (Bureau of Land Management road; Congdon Creek Road) below the spray unit to view the damage. All of the same symptoms of the vapors intensified again to the level they were present during the first few weeks after the unit was sprayed, so clearly the vapors were still present, and clearly the symptoms were a result of exposure to the vapors.

The photo below shows the same view of the sprayed unit over five weeks later (taken October 2, 2009). This unit was still fuming vapors which affected my health negatively, and were still capable of drifting off-target for a significant distance.

Vapor drift is significant and harmful to human health, animal health and the environment. Vapor drift is capable of being transported over long distances and lasts for days, weeks and months. Not only initial drift but also vapor drift must be taken into account by the Environmental Protection Agency while regulating pesticides and pesticide adjuvants.

Respectfully submitted by

[redacted]

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

The use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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

Acknowledgments v
Abstract 1
Introduction 1
Literature Review 2
    Forest Processes 2
    Water-Quality Effects on Streams from Forest Fertilization 10
        Immediate Nutrient Runoff 10
        Longer-Term Nutrient Losses to Streams 10
        Water-Quality Criteria 11
    Ecological Effects on Streams from Forest Fertilizations 12
    Stream Nutrient Dynamics 13
    Hyporheic Processing 15
        Conceptual Model of Ecological Processing of Fertilizer Nitrogen 16
        Suggested Approaches to Evaluate Ecological Effects of Forest Fertilization 17
Little River Watershed 20
    Physiographic Setting 20
    Water-Quality Issues in the Little River Watershed 22
    Land uses in the Little River Watershed and Potential Effects on Water Quality 23
    Water-Quality Conditions 24
        Methods 24
        Quality Assurance 29
        Environmental Data 30
Framework for Fertilization Study in Wolf Creek and Little River Watershed 35
    Possible Study Approaches 38
Summary 40
References Cited 41

TABLES

1. Summary of studies of forest fertilization effects on stream-water quality 4
2. Aquatic species of concern in the Little River watershed, Oregon 22
3. Sites sampled in the Little River watershed by the U.S. Geological Survey (USGS) in 1998 and 1999 25
4. Nutrient and field data in the Little River and tributaries from reconnaissance samplings during August and November, 1998 26
5. Nutrient and field data for sites in the Little River and tributaries, August 1999 27
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) 29
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 30
8. Algal biomass and chlorophyll a measured in the Little River watershed, August 1999 35
9. Research components for different levels of investigation of effects of urea fertilization on water quality and stream ecology 39

FIGURES

1. Map showing Little River watershed, Oregon, and proposed Bureau of Land Management fertilization units ... 3
2. Forest cycling pathways representing major processes and fates of nitrogen fertilizers ... 9
3. Schematic drawing of hypothetical transport pathways, dominant processes, and relative concentrations of nitrogen in response to urea fertilization in forested cathments of the western Cascades, Oregon, during fall/winter and late spring/summer ... 18
4. Map showing sampling locations in Little River Watershed, Oregon 1998 ... 21
5. Graph showing morning and afternoon pH and dissolved oxygen saturation in the Little River, July 28, 1998 ... 22

6.–9. Graphs showing:
6. Afternoon temperature in the main stem of the Little River, August 1999 ... 32
7. Afternoon pH in the main stem of the Little River, August 1999 ... 32
8. Major ion chemistry in Little River and Wolf Creek ... 33
9. Distribution of nutrient concentrations in the Little River and tributaries during August, 1999 ... 34

10. Concentrations of nitrate-nitrogen and total phosphorus, and daily maximum temperature and pH in the Little River Basin during August, 1998 ... 36

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

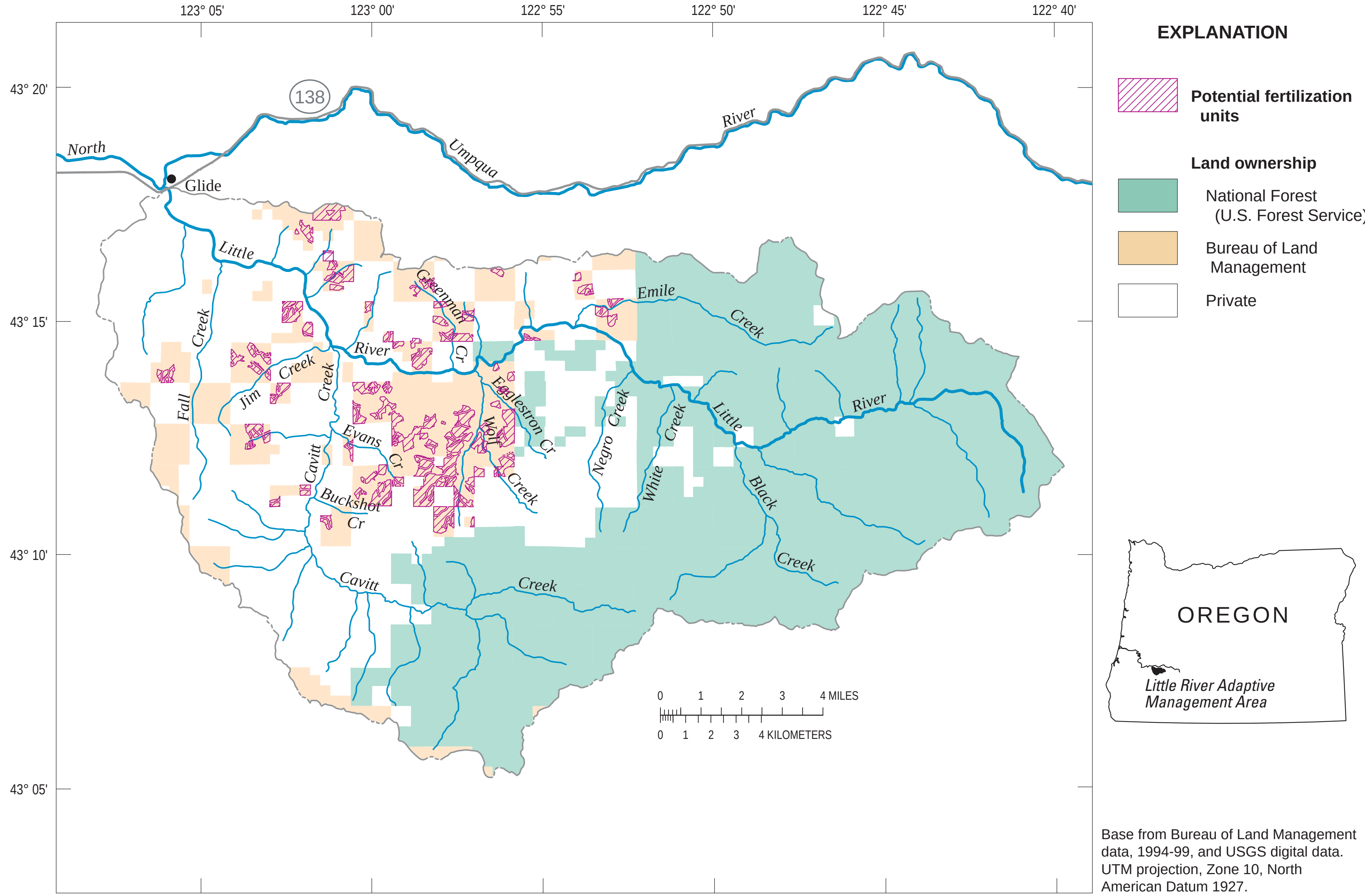


Figure 1. Map showing Little River watershed, Oregon, and proposed Bureau of Land Management fertilization units.

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 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: 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

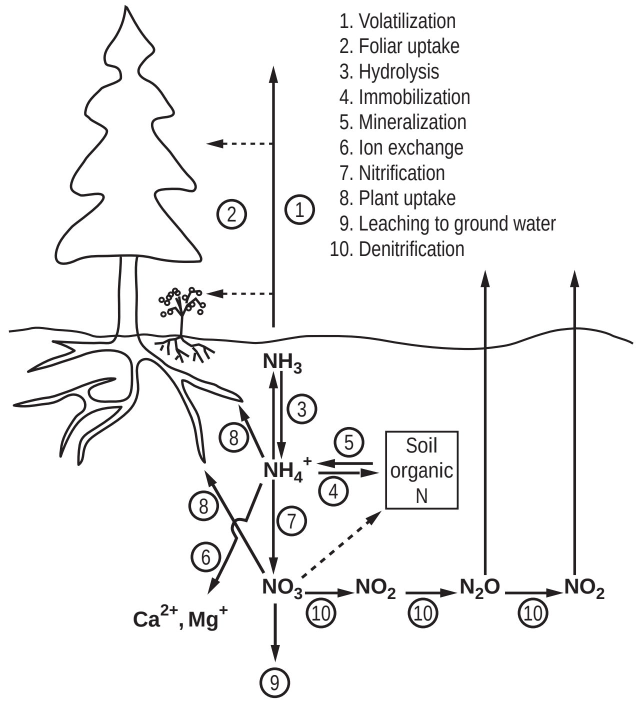


Figure 2. Forest cycling pathways representing major processes and fates of nitrogen fertilizers. (Modified from Nason and Myrold, 1992.)

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;

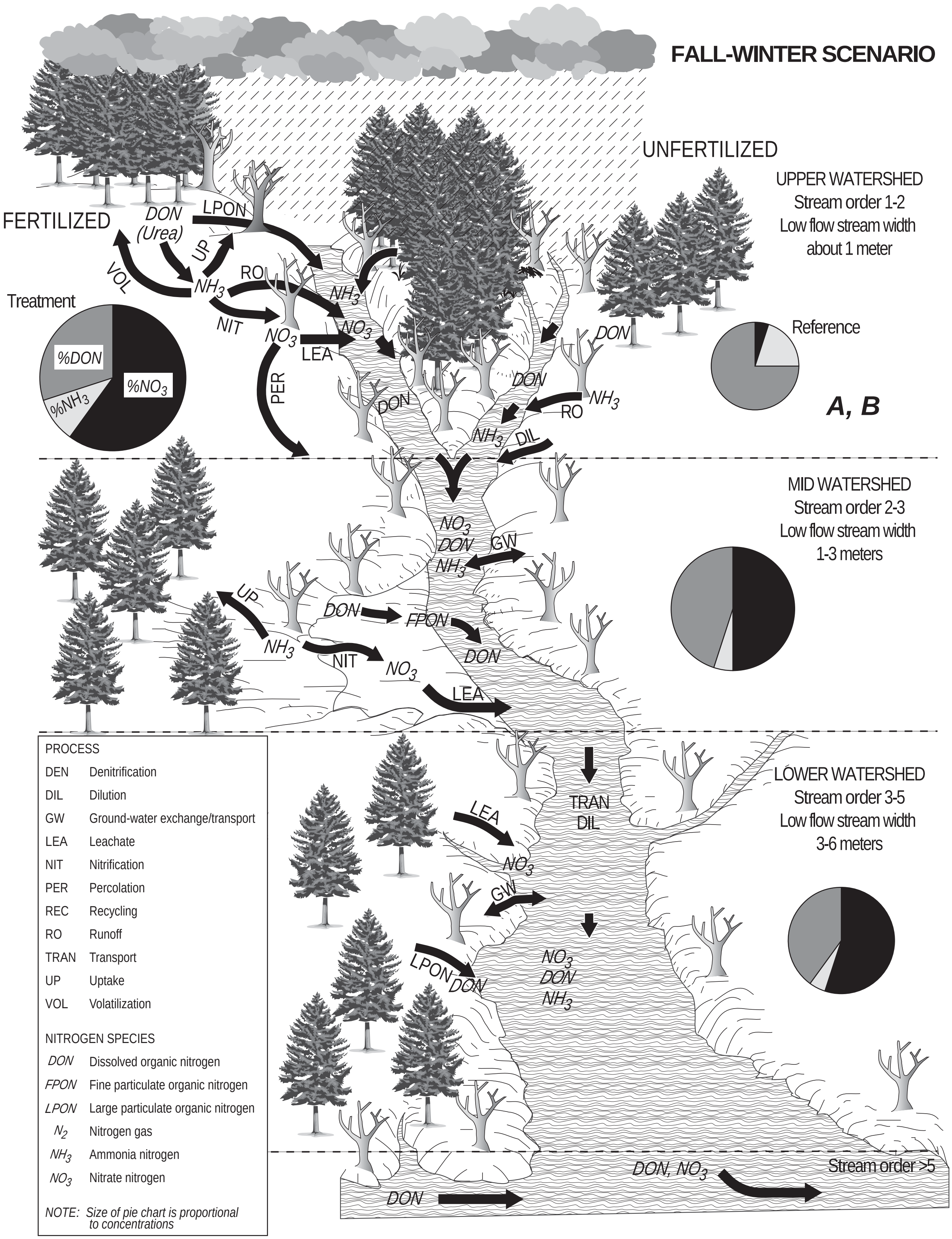
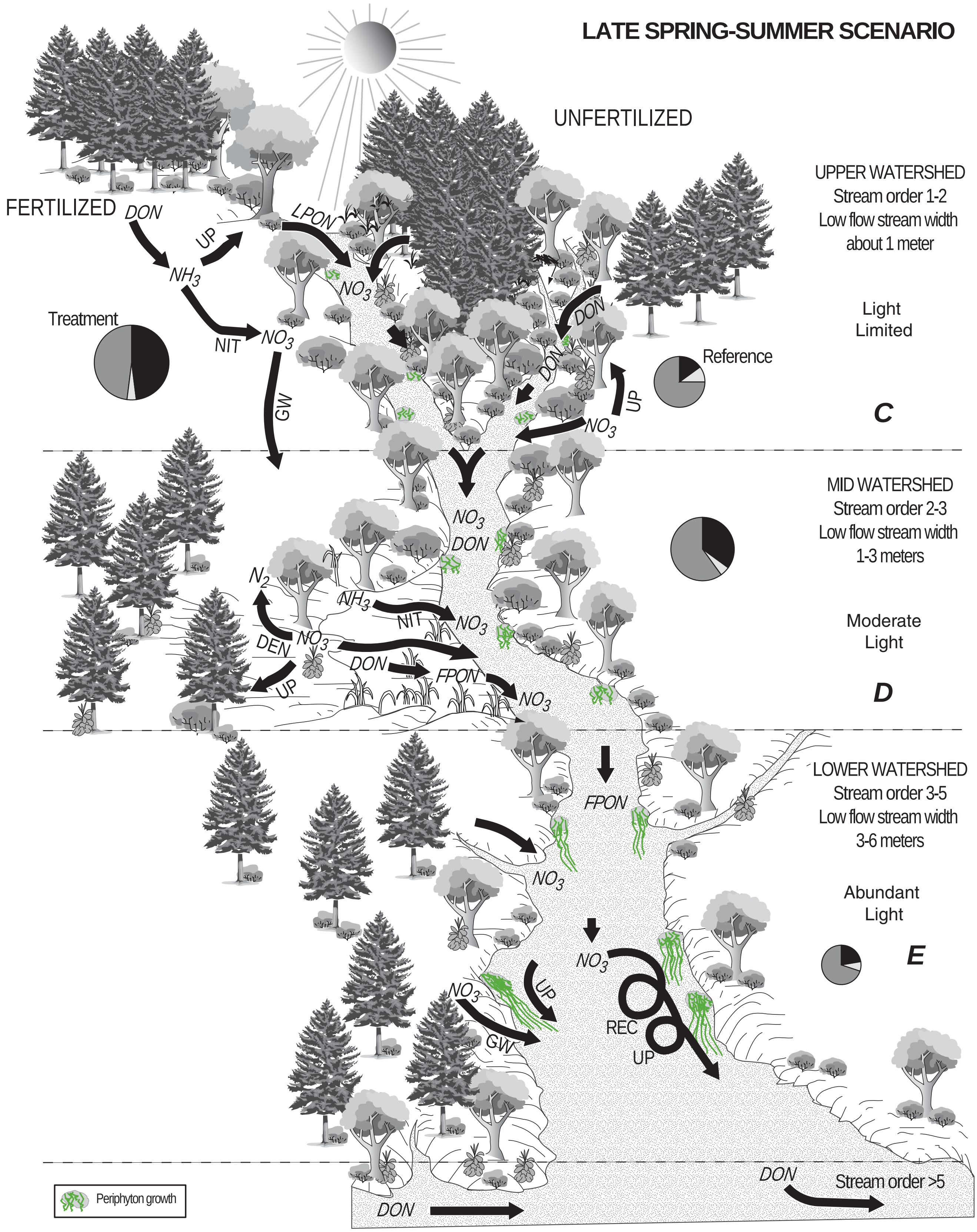


Figure 3. Hypothetical transport pathways, dominant processes, and relative concentrations of nitrogen in response to urea explained in the text.)



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

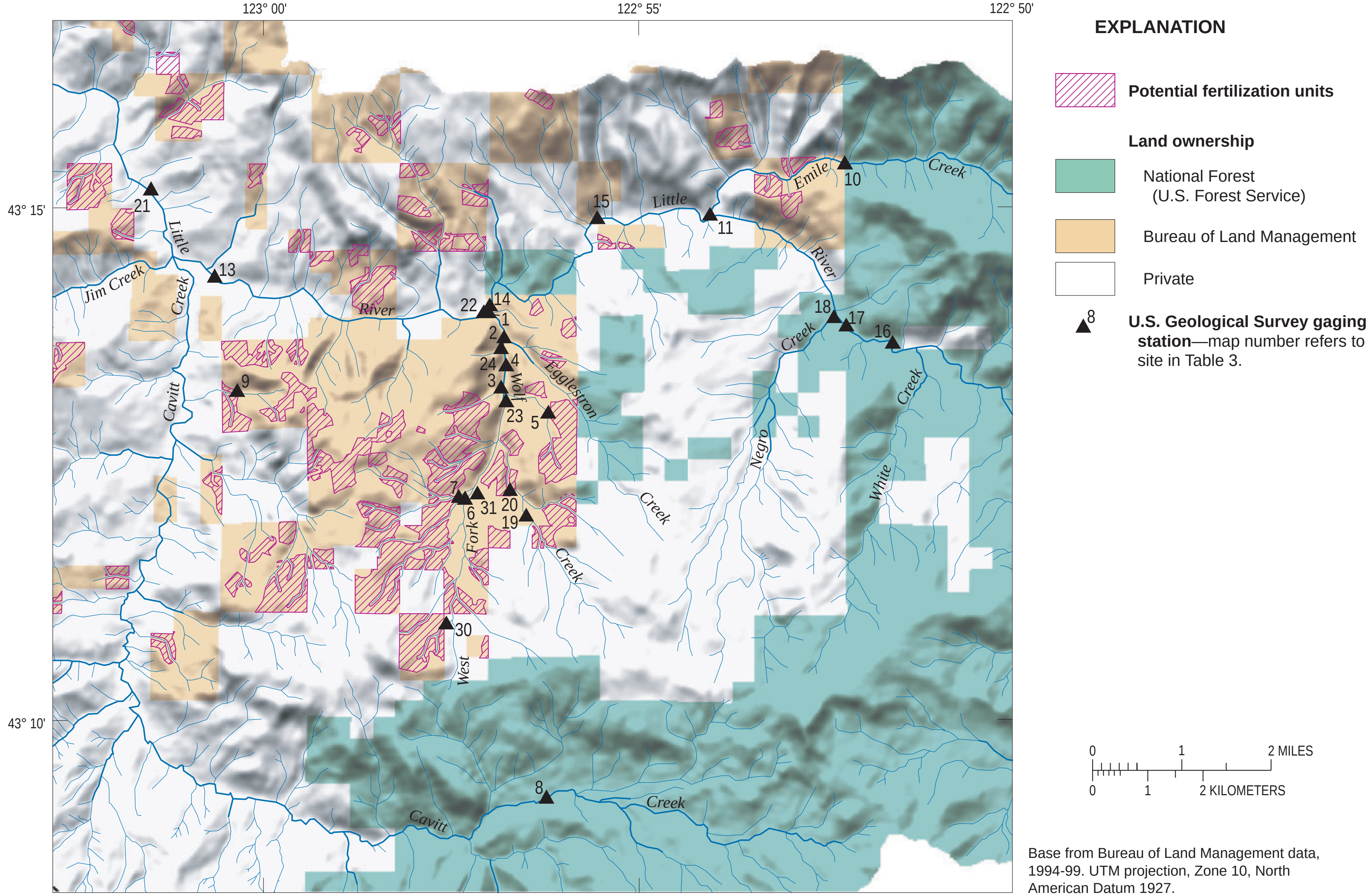


Figure 4. Sampling locations in Little River Watershed, Oregon 1998.

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).

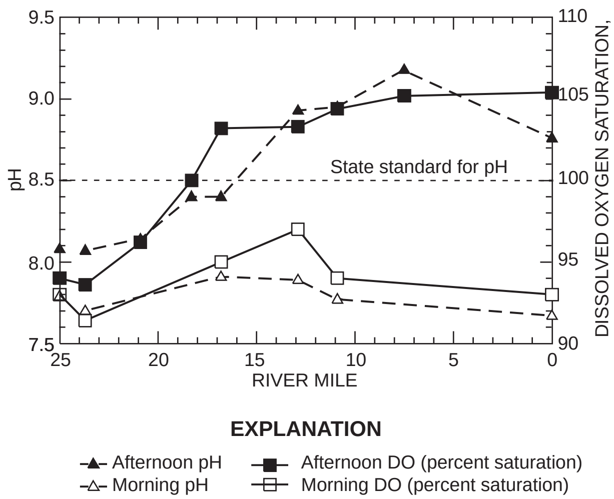


Figure 5. Morning and afternoon pH and dissolved oxygen saturation in the Little River, July 28, 1998. (Source: Powell, 1998.)

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 (×°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.

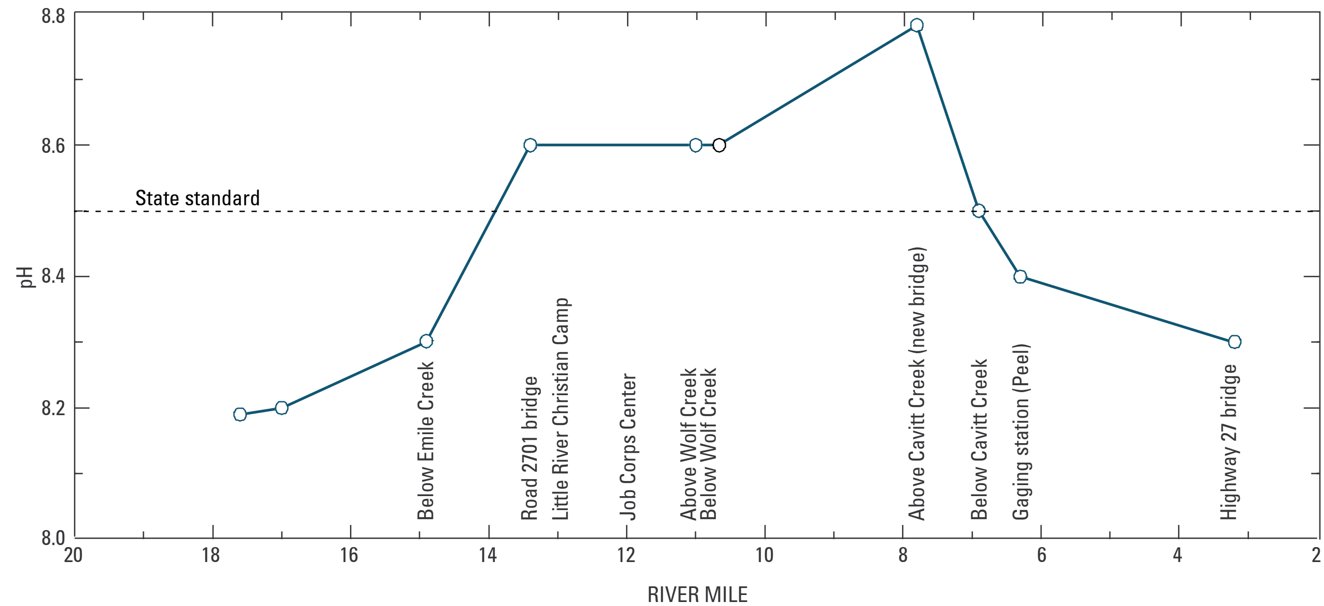


Figure 6. Afternoon pH in the main stem of the Little River, August 1999.

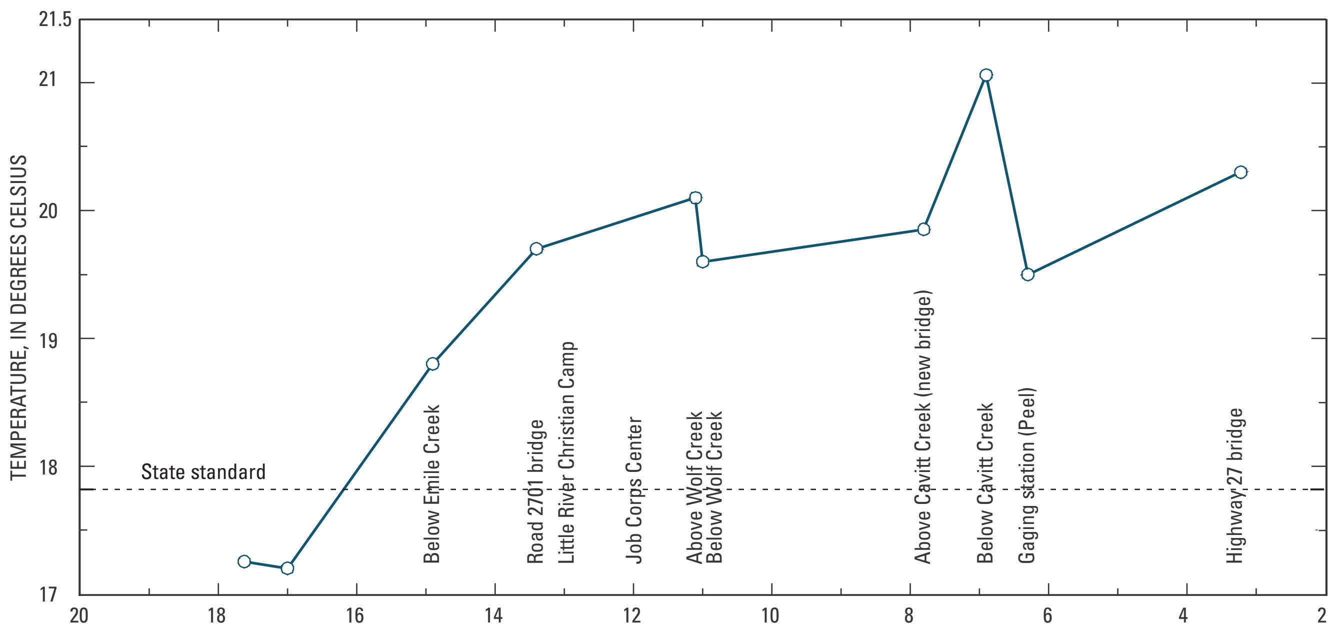


Figure 7. Afternoon temperature in the main stem of the Little River, August 1999.

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).

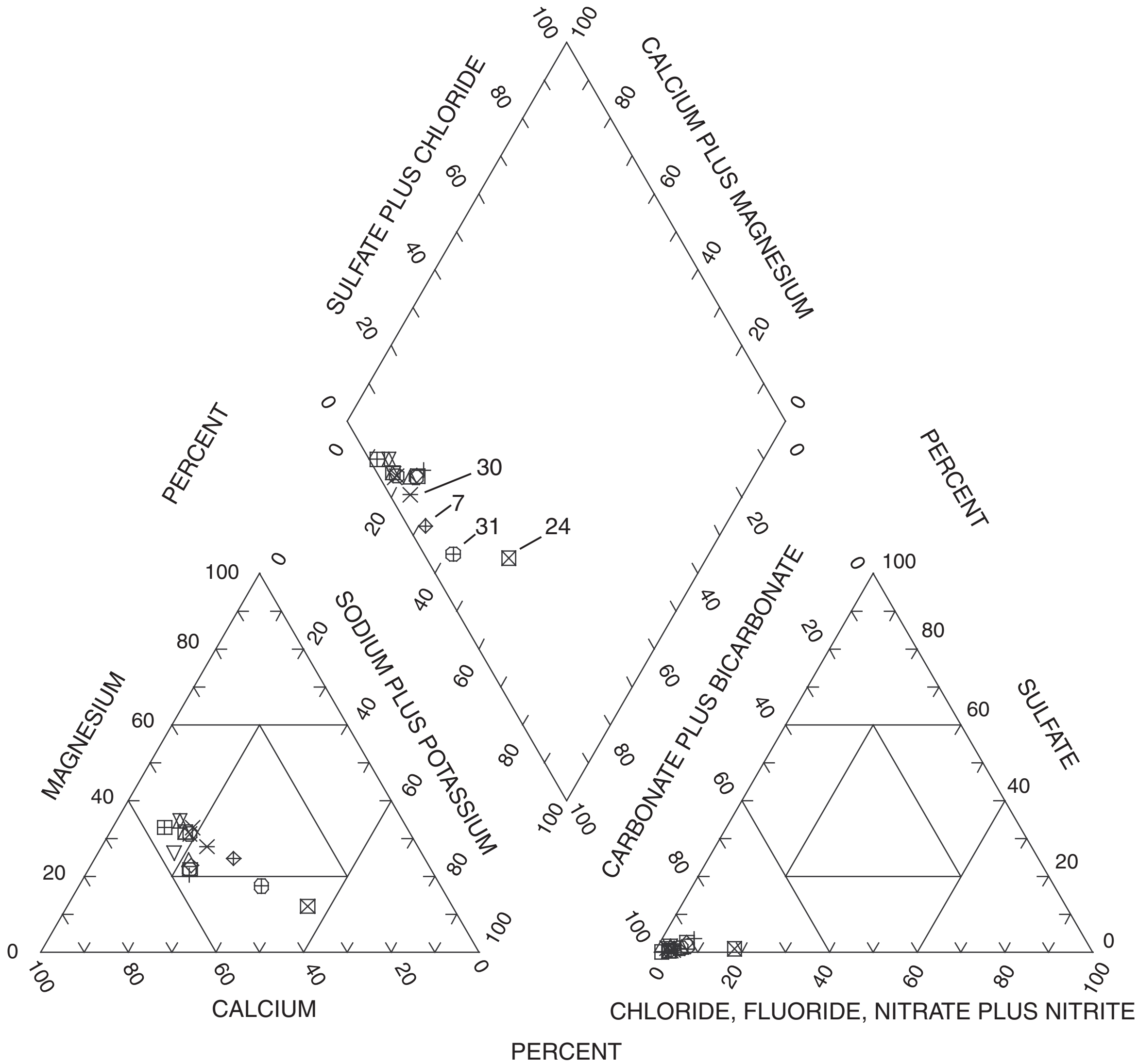


Figure 8. Major ion chemistry in Little River and Wolf Creek. (Sites that are the most chemically distinct are the reference tributary to Wolf Creek above Egglestron Creek [site 24], an unnamed tributary east of the West Fork of Wolf Creek on road 16.1 [site 30], an unnamed tributary west of the West 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

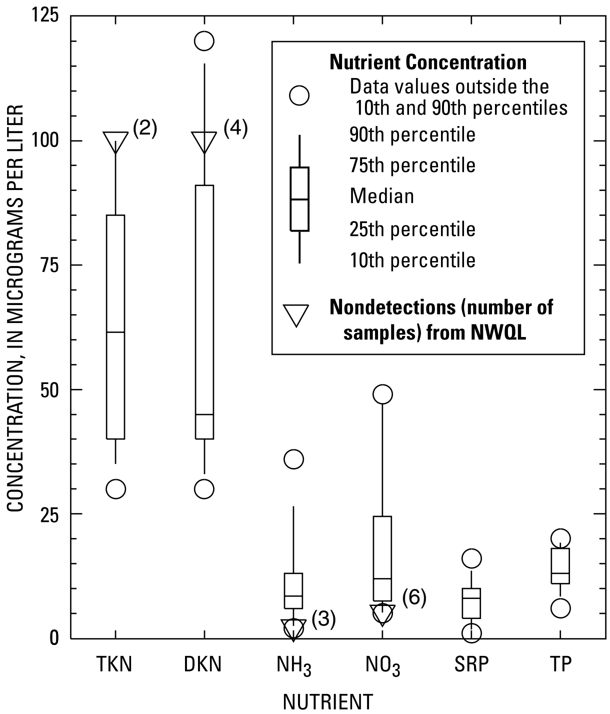


Figure 9. Distribution of nutrient concentrations in the Little River and tributaries during August, 1999. (All data are from the U.S. Geological Survey's National Water Quality Laboratory (NWQL) in Denver, Colorado, except dissolved Kjeldahl nitrogen (DKN) and total Kjeldahl nitrogen (TKN), which include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

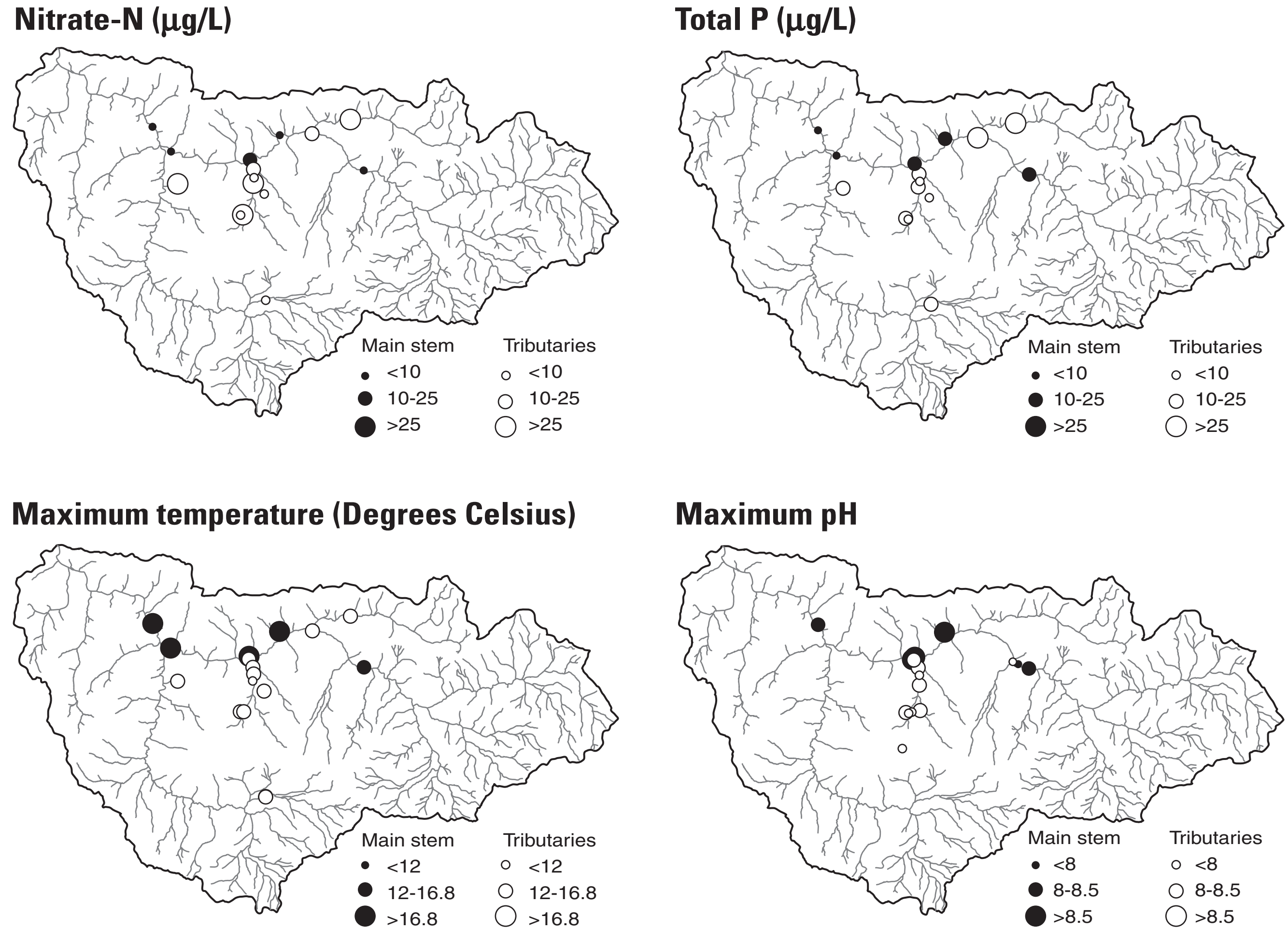


Figure 10. Concentrations of nitrate-nitrogen and total phosphorus, and daily maximum temperature and pH in the Little River Basin during August, 1998. (Sampling sites are given in table 3 and figure 5, and water quality data are provided in tables 4 and 5.)

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 $\delta = (R_X/R_S\text{-}1)*1000$, 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 multilevel 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.

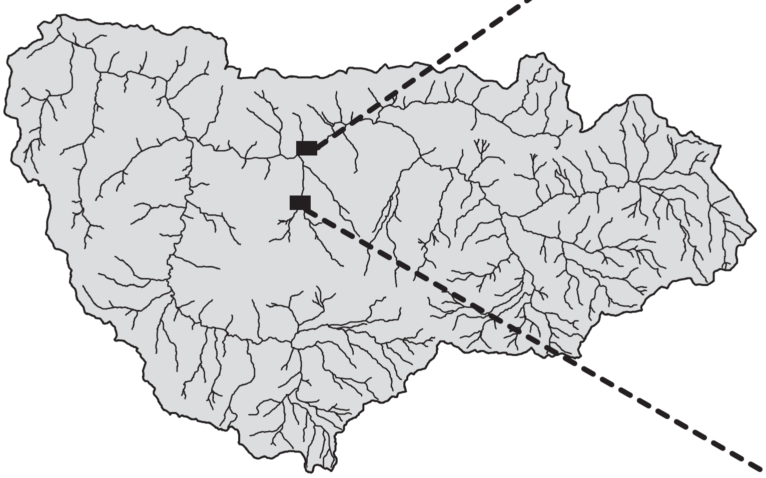
Woessner, W.A., 2000, Stream and fluvial plain ground water interactions—Rescaling hydrogeologic thought: Ground Water, v. 38, no. 3, p. 423–429.

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)*.

Diminishing Returns:

Oregon Pesticide Education Network

Salmon Decline and Pesticides

A publication of the
Oregon Pesticide Education Network (OPEN)

- Oregon Environmental Council
- Northwest Coalition for Alternatives to Pesticides
- OSPIRG Foundation
- Institute for Fisheries Resources

We gratefully acknowledge the support of:

- The Pew Charitable Trusts
- The Bullitt Foundation
- Turner Foundation
- The Tides Center
- Rockefeller Family Fund
- True North Foundation
- The Educational Foundation of America

as well as numerous individuals who have supported the work of the Oregon Pesticide Education Network.

We sincerely appreciate the special contributions that the following individuals made to this report: Jeff Allen, Lev Anderson, Robie Honeyman-Colvin, Caroline Cox, Georgia Ewing, Lita Furby, Norma Grier, Neva Hassanein, Jacob Kann, Maureen Kirk, Becky Long, Kay Rumsey, Glen Spain, and Laura Weiss.

We also value the following individuals and agencies who contributed photos and graphics that are reproduced here with permission: Nancy Allen of Corvallis Environmental Center, K. Born, Andy Duncan and Lynn Ketchum of OSU Extension and Experiment Station Communications, Institute for Fisheries Resources, Kathryn Kostow, Tom and Pat Leeson, Oregon Dept. of Environmental Quality, Oregon Dept. of Fish and Wildlife, Public Interest GRFX, Oregon Sea Grant, Ana Soto, Steve Stone of National Marine Fisheries Service, and the US Fish and Wildlife Service.

The opinions expressed in this report do not necessarily reflect the views of the supporting foundations and individuals.

Copies of this report can be found at:
http://www.pond.net/~fish1ifr/salpest.pdf

Diminishing Returns: Salmon Decline and Pesticides

By Richard D. Ewing, PhD
February 1999

TOM AND PAT LEESON

About the Author:

Richard D. Ewing is a native Oregonian who graduated from Reed College in 1962, and received his Ph.D. in cellular and molecular biology from the University of Miami at Coral Gables in 1968. After conducting research at Oak Ridge National Laboratory and at Oregon State University, Dr. Ewing joined the research team at the Oregon Department of Fish and Wildlife where he worked as a physiologist and hatchery specialist from 1975-1992. He left ODFW to form Biotech Research and Consulting, Inc., a Corvallis-based consulting company that specializes in hatchery operations and chemical analyses relating to fisheries. Dr. Ewing has published widely in professsional journals and written many reports for the Oregon Department of Fish and Wildlife.

Table of Contents

Executive Summary 3

Introduction 7

Dynamics of Pesticides in the Environment 10

- Pesticides are Ubiquitous in the Western Watersheds 10
- Pesticides Persist in the Environment and Degrade into Toxic Products 11
- Pesticides Move throughout Watersheds 12
- Pesticide Concentrations Can Become Magnified in Tissues 14

Fish Kills and Acute Toxicity of Pesticides 17

Behavioral Effects of Pesticides at Sublethal Concentrations 19

- Pesticides Impair Swimming Performance 19
- Pesticides Can Increase Predation on Juvenile Salmonids 21
- Temperature Selection is Changed by Pesticides 22
- Schooling Behavior is Reduced by Pesticides 22
- Pesticides Can Interfere with Seaward Migration 23
- Pesticides Interfere with Seawater Adaptation 23
- Pollutants Affect Adult Returns 25

Compromised Immune Systems 27

Endocrine Disruptors 29

- Pesticides Can Mimic Estrogens 30
- Pesticides May Have Other Endocrine Effects 34

Indirect Effects of Pesticides on Salmon 36

Conclusions 39

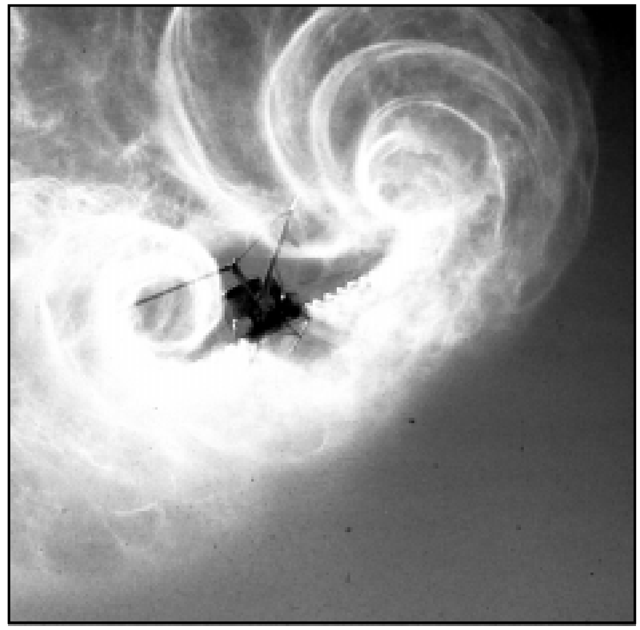
Recommendations 43

References 45

Glossary, Abbreviations and Scientific Names 52

Executive Summary

Pacific salmon are in serious trouble. The enormous runs of migratory salmon of the past have slowly diminished to the trickle of adult spawning salmon that presently inhabit western rivers. Although salmon recovery efforts are underway, scientists, policy makers, and interest groups have thus far given insufficient attention to the role that pesticide contamination of our watersheds may play in salmon decline. Accordingly, the purpose of *Diminishing Returns: Salmon Decline and Pesticides* is to review scientific literature on the effects of sublethal concentrations of pesticides on salmonids (see full report for documentation and references). The report places special emphasis on how pesticides can alter the biology of fishes in subtle ways that decrease their chances for reproduction and survival.

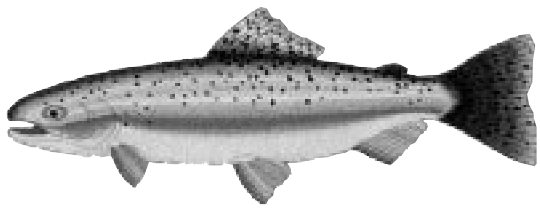


Dynamics of Pesticides in the Environment

Pesticides include a broad class of chemical and biological agents that are purposefully introduced into the environment to kill or damage organisms, including insecticides, herbicides, and fungicides. Once applied, pesticides move into streams and rivers throughout watersheds and may pose problems far from the site of application. Movement often occurs through the medium of water, thereby exposing all aquatic organisms during this transport. Where water quality monitoring has been done, a great variety of pesticides are typically found in salmon habitat. Federal and state agencies have established few criteria or standards for the protection of aquatic life from short-term (acute) and long-term (chronic) exposure to pesticides.

Pesticides do not necessarily disappear with time. They transform into other compounds that may be less toxic, of equal toxicity, or of greater toxicity than the original compound. The toxicity of these breakdown products is not well understood, and in general how they affect aquatic life has not been studied. All the while, fish and other aquatic organisms must continue to cope daily with pesticides (and their breakdown products), some of which are no longer used but remain in watersheds.

Although pesticides are diluted by transport in rivers and streams, a number of mechanisms concentrate the chemicals, often to toxic levels. In a process known as bioaccumulation, pesticides absorbed into plant and animal tissues may become concentrated and reach levels many times higher than those in surrounding water.



Fish Kills and Acute Toxicity of Pesticides to Salmon

Pesticides are capable of killing salmon and other aquatic life directly and within a short period of time. For example, in 1996 the herbicide acrolein was responsible for the death of approximately 92,000 steelhead, 114 juvenile coho salmon, 19 resident rainbow trout, and thousands of nongame fish in Bear Creek, a tributary of the Rogue River. Deaths of threatened and endangered species from accidental contamination of waterways are of grave concern. The loss of each individual in a sensitive population makes recovery efforts that much more difficult. Fortunately, these deaths are relatively infrequent.

Behavioral Effects of Pesticides at Sublethal Concentrations

In contrast to dramatic fish kills, the effects of sublethal concentrations of pesticides are more subtle and go largely unseen and unregulated. Sublethal concentrations of pesticides do not cause immediate death, but can interfere with the biology of the organism in other ways and can ultimately impact the survival of the species. Laboratory studies show that sublethal concentrations of pesticides can affect many aspects of salmon biology, including a number of behavioral effects:

- Long-term exposure to certain pesticides can increase stress in juvenile salmonids and thereby render them more susceptible to predation.

- Certain pesticides can alter swimming ability, which in turn can reduce the ability to feed, to avoid predators, to defend territories, and to maintain position in the river system.

- Many pesticides interrupt schooling behavior, a critical tactic for avoiding predation during salmon migration. Disruption of schooling behavior is thought by some researchers to be a classic method for examining sublethal effects of pesticides because the effect is so common.

- Several pesticides (and other pollutants) have been shown to cause fish to seek suboptimal water temperatures, thus subjecting them to increased dangers of disease and predation.

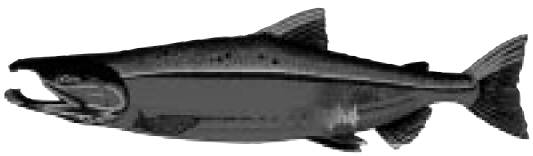
- Some herbicides have been shown to inhibit normal migration to the sea, resulting in severe disruption of the life cycle. There is a dearth of research looking at this effect for common insecticides.

- Several studies suggest that certain pesticides can impair salmonid's ability to transition from freshwater to seawater. There is a need for further research in this area, placing particular emphasis on the critical period of transition that takes place in the estuary.

- Adult salmon adjust their migration patterns to avoid polluted areas, resulting in delayed spawning.

Compromised Immune Systems

In addition to changes in behavior, exposure to relatively low concentrations of pesticides can disrupt the immune system of salmon. Evidence for these effects in salmonids is not as extensive as for disruption of behavior, but the data available suggest that pesticides can have serious negative impacts on the immune system. Such disruption results in the onset of disease and even death.



Endocrine Disruptors

Fish and other organisms are especially vulnerable to endocrine-disrupting effects during the early stages of development. Pesticides at low concentrations may act as mimics or blockers of sex hormones, causing abnormal sexual development, feminization of males, abnormal sex ratios, and unusual mating behavior. The unique plasticity of sex differentiation in fish suggests that these animals may be very susceptible to disruption of sexual characteristics by pollutants. Pesticides can also interfere with other hormonal processes, such as thyroid functioning and bone development.

Indirect Effects of Pesticides on Salmon

Pesticides can indirectly affect fish by interfering with their food supply or altering the aquatic habitat, even when the concentrations are too low to affect the fish directly. Such indirect effects greatly reduce the abundance of food organisms which in turn reduces the growth and probability of survival of the fish. In addition, removal of aquatic vegetation can decrease habitat suitability and increase the salmon's susceptibility to predation. These indirect effects are subtle, but evidence suggests that in complex ecosystems indirect effects can be even more important than direct effects.

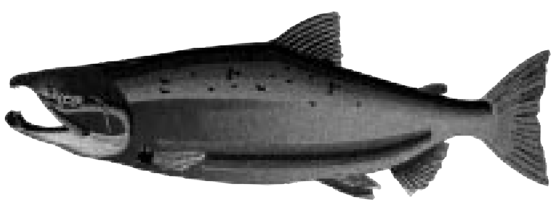
Recommendations

From the evidence available at present, there is a plausible basis for considering pesticides to be one of the causes of declining salmon populations in the Pacific Northwest. Based on this review, we offer several policy recommendations and identify areas for further research:

1. Address the impacts of pesticides on salmon when developing and implementing recovery plans for threatened and endangered species. To date, efforts to recover salmon have devoted insufficient attention to pesticides as a contributing factor in salmon decline. We must act now using available information to formulate management strategies that will minimize the potential danger from sublethal concentrations of pesticides.

2. Conduct ecoepidemiological studies in critical salmonid habitat. Most of the effects of pesticides referred to in this report have been determined in experimental laboratories. In the field, however, environmental conditions are not controlled, and many factors interact to confuse the determination of direct relationships. Ecoepidemiological investigations in the Great Lakes have established the relationship between chlorinated hydrocarbons and the decrease in lake trout populations. An ecoepidemiological approach for salmon in the Pacific Northwest would be particularly valuable because it is designed to attribute causality to events occurring in real-world situations.

3. Create comprehensive pesticide tracking systems in the Pacific Northwest. To better understand the relationship between pesticides and salmon decline, we must have accurate, site-specific data on the patterns of pesticide use in the watersheds of the Northwest. State and provincial governments need to collect data on which pesticides are used where, when, and in what amounts. Such data can then be combined with watershed-specific information on indicators of salmon health. (Currently, California is the only state with Pacific salmon habitat where such information is collected.) Pesticide use information will also enable efficient instream monitoring for pesticide contamination.



4. Establish instream monitoring programs in critical salmon habitats. A systematic monitoring program for pesticides and their breakdown products needs to be undertaken. Not all pesticides can be tested for in all locations, but current testing is woefully inadequate for understanding the role of pesticides in salmon decline. In conjunction with pesticide use data, these analyses can be targeted to the compounds of most concern. Such targeting can greatly improve the cost-effectiveness of monitoring.

5. Err on the side of caution when setting water quality standards for pesticides. There are few established criteria for the protection of aquatic life from pesticides. Moreover, evidence reviewed here shows that sublethal effects on salmonids have not been fully appreciated, that juvenile salmonids succumb more easily to toxins in the water, that laboratory studies do not reflect the natural life cycle of the fish, and that little is known about how pesticides affect aquatic ecosystems. These factors must be considered when setting standards, and a precautionary approach must be adopted.

6. Prevent pesticide contamination of salmonid habitat by reducing pesticide use. Once contaminated, water is difficult if not impossible to clean up. Therefore, pest management approaches that do not depend on pesticide use in agricultural and non-agricultural settings should be encouraged and further developed. There is ample evidence that ecologically sound and economically viable methods can be successfully implemented. The adoption of such alternatives can be encouraged through technical assistance, financial incentives and disincentives, demonstration programs, and information exchange opportunities.

7. Adopt state and provincial programs in the Pacific Northwest to phase out pesticides that persist and bioaccumulate in the environment. Numerous pesticides, including some that are no longer used and many that are currently used, are known to persist in the environment and to bioaccumulate in aquatic systems. Washington State's Department of Ecology is now considering a plan to end the release of such toxins, including certain pesticides, into the environment. To ensure salmon recovery, all state and provincial governments in the Pacific Northwest should adopt similar programs.

Introduction

Salmon are a keystone species for Northwest ecosystems. Entire foodchains base their existence on the proliferation of salmon and trout that surge from the sea into streams and rivers, work their way upstream to their natal spawning areas, lay their eggs, and die, providing a source of nutrients for the very organisms that later feed the salmon's progeny.

Salmon and trout make up the family Salmonidae, a group of fish characterized by an adipose fin located near the tail. These fish require clean, cold fresh water for much of their life history. Because such habitat is low in nutrients, salmon and many of the trout developed anadromy, a behavior pattern in which the juveniles migrate to the sea. In the sea, the juvenile fish use rich ocean productivity for feeding and growth, then return to the streams where they were born to lay their eggs.

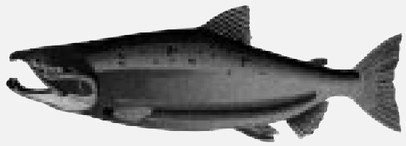
Rivers and streams have become great conduits through which pesticides, either intact or as breakdown products, flow to the sea. Salmon now live throughout their life cycle with these residues as part of their daily environment.

Today, migratory salmon and trout are fighting for their survival. The enormous runs of migratory salmonids in the 1860s have slowly diminished to the trickle of adult spawning salmon that presently inhabit western rivers. The causes for the decline are many, some of which are well documented. Yet, in their recovery efforts for threatened populations, scientists, policy makers, and interest groups have tended to overlook the role of pesticides that flow daily through salmon habitat, potentially changing the biology of the fish in subtle ways that decrease their chances for survival and reproduction.

A great deal of work is being done to catalog and restore the physical aspects of Northwest streams (e.g., restoring riparian vegetation and gravel beds, planting buffer strips along streams). Insufficient attention, however, has been devoted to the use and presence of pesticides in the watersheds and the role this water quality degradation plays in salmon decline. Accordingly, the purpose of this report is to present information from scientific literature that points to the unseen danger posed by the existence of pesticides in salmonid habitats.

Pesticides include a broad class of chemical and biological agents that are purposefully applied to the environment to kill or damage organisms (National Research Council 1993). These agents are used in a wide range of occupations for a variety of purposes. Foresters use herbicides to keep broadleaf plants from competing with conifer seedlings, and insecticides to deal with numerous insect pests that damage forests. Farmers use insecticides to protect their crops and keep insects away from their livestock, and herbicides to remove unwanted weeds from fields and waterways. State and local agencies use herbicides to remove brush from roadsides. Fishermen use defouling compounds like tributyltin to keep organisms from settling on the hulls of their boats. The ordinary homeowner uses a wide variety of pesticides: herbicides on the lawn, insecticides on pets, and fungicides in house paint.

Oregon’s ocean-going salmon

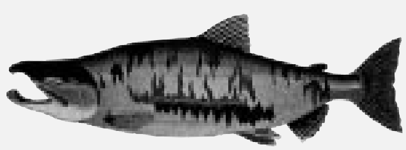


Chinook Salmon (ocean-rearing)

- Distribution includes coast and Columbia Basin mainstem rivers.
- Juveniles migrate to the ocean the first fall after they hatch, rearing briefly in estuaries.
- They rear over a broad ocean area, ranging from northern California to the Gulf of Alaska.
- Adults, typically 3 to 5 years old, return to fresh water in the spring, summer or fall.
- Spring and summer migrants prefer deep, cool pools where they hold several months before fall spawning.
- Adults spawn in large concentrations on mainstem gravel bars; may use both upper and lower mainstems.

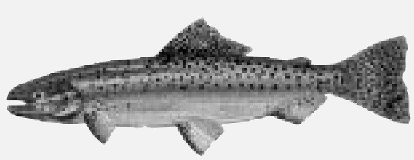
Chinook (stream-rearing)

- In Oregon, they are only in upper Columbia Basin tributaries.
- Juveniles migrate to the ocean as 1-year-olds, in the spring.
- Little is known about the ocean distribution of Oregon’s stream-rearing chinook.
- Adults return to fresh water in the spring, when 3 to 5 years old, and require deep, cool pools to hold for several months over the summer before fall spawning.
- They spawn in concentrations on gravel bars in upper tributaries.



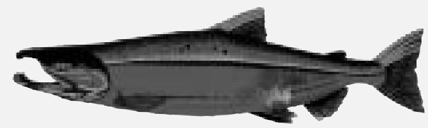
Chum Salmon

- Shortest freshwater residence of all salmon. Adults stay only about a week prior to spawning; juveniles migrate to the ocean hours after hatching.
- Juveniles rear briefly in estuaries.
- Most Oregon chums migrate to the Gulf of Alaska for ocean rearing.
- Adults spawn at 3 to 5 years of age.
- Spawning occurs in lower mainstems, concentrated on large gravel bars.
- Adults are unable to pass even minor barriers.



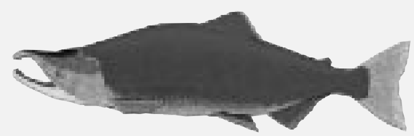
Coastal Cutthroat

- Some coastal cutthroats migrate to the ocean. But others may migrate only to the estuary or river mainstems, or they may not migrate at all.
- Those that do go to the ocean migrate out in the spring, stay only a few months close to shore, then return in the fall.
- The ones that migrate may rear in fresh water for several years before going to the ocean.
- They spawn in the winter and early spring, using small pockets of gravel. They may spawn more than once. The spawning age of cutthroats seems to vary over their distribution area.
- Cutthroat prefer the smallest, highest tributaries in a basin.



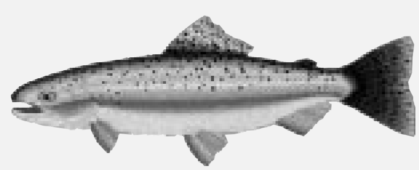
Coho Salmon

- Juveniles rear throughout watersheds and tend to live in pools in the summer.
- Juveniles migrate to the ocean at 1 year, in the spring.
- Most Oregon coho rear just off our coast.
- Adults return to fresh water in the fall and spawn in late fall and winter.
- Adults tend to spawn in concentrations on gravel bars in upper watersheds.
- Most adults spawn when they are 3 years old.



Sockeye/Kokanee Salmon

- There is both an ocean-going form (called sockeye), and a resident form (called kokanee).
- Juveniles rear in a lake, spending 1 to 2 years in fresh water before migrating to the ocean in the spring.
- Columbia Basin sockeye migrate to the Gulf of Alaska for ocean rearing.
- Adults typically spend 2 years in the ocean.
- Loss of Oregon sockeye resulted from blocked access to lakes. Kokanee are thriving in some lakes.



Steelhead

- There are two subspecies of steelhead in Oregon. Each also has a resident form. Coastal steelhead are closely related to rainbow trout. Inland steelhead are closely related to redband trout.
- Most juveniles rear in fresh water for 1 or 2 years and migrate to the ocean in the spring.
- Most steelhead spend 2 years in the ocean. Their distribution is poorly known but appears to be further off-shore than other salmon.
- Most inland steelhead return to fresh water in the summer while most (but not all) coastal steelhead return in the winter.
- Summer-run steelhead require cold, deep pools where they hold until spawning. All steelhead spawn in the winter and may spawn more than once.

ILLUSTRATION AND INFORMATION COURTESY: KATHRYN KOSTOW, OREGON DEPARTMENT OF FISH AND WILDLIFE

Once applied, pesticides can move away from the point where they were used. As a result, rivers and streams have become great conduits through which pesticides, either intact or as breakdown products, flow to the sea. Salmon now live throughout their life cycle with these residues as part of their daily environment.

This report reviews our existing scientific knowledge of the effects of pesticides on salmonids, placing a special emphasis on sublethal effects of pesticides. Sublethal effects are those that result from exposure to a pesticide in an amount that is not high enough to cause death, but can damage an organism in other ways, including physiological and behavioral changes that can ultimately impact the survival of the species. These effects can occur throughout the entire life history of salmonids, from hatching of eggs, entry of juveniles into the ocean, and return of adults for spawning. With the data currently available, it is possible to identify key areas that need immediate attention.

The present report emphasizes research performed on salmonids. Most of the studies included here, however, have been done with rainbow trout, which are found throughout the United States, are relatively easy to grow, and provide a reasonable standard for examining physiological and behavioral changes. Fewer studies were done with anadromous (seagoing) salmon. Where this research exists, the information is emphasized. Studies from other species of fish are introduced when certain points need to be made and information is not available from salmonid research.

The data suggest there is a plausible basis for considering pesticides as a causative factor in salmon population declines.

Dynamics of Pesticides in the Environment

Pesticides are Ubiquitous in Western Watersheds

In 1991, the U. S. Congress provided funds for the U. S. Geological Survey to conduct a National Water-Quality Assessment (NAWQA) on major river systems in the United States. The study units chosen encompass sources of drinking water for about 70 percent of the U. S. population. In the Northwest, study units included the Puget Sound and the central Columbia River plateau of Washington, the Willamette River Basin of Oregon, the Sacramento and San Joaquin River systems of California, and the Snake River Basin of Idaho. Table 1 presents the numbers of pesticides detected in streams and the numbers of pesticide detections that exceed criteria for aquatic life.

A major problem with the interpretation of the data presented in Table 1 is that concentrations of pesticides compatible with aquatic life are not well defined. The U.S. Environmental Protection Agency (EPA) has identified aquatic life criteria for the protection of aquatic organisms from short-term (acute) or long-term (chronic) exposure for very few pesticides. Of the 118 pesticides typically looked for in water quality studies, only 20 (17%) have been

Table 1. Pesticide detections in the western United States as measured by the U. S. Geological Survey NAWQA program.

State, region	# Pesticides Examined	# Pesticides Detected	# Exceeding Aquatic Life Criteria[1]	# For Which Aquatic Life Criteria Are Available
Oregon				
Willamette Basin[2]	86	36	4	22
Washington				
Puget Sound Basin[3]	NA	23	4	NA
Central Columbia River Plateau[4]	84	45	5	18
California				
San Joaquin Basin[5]	83	49	7	16
Idaho				
Snake River Basin[6]	80	36	2	17

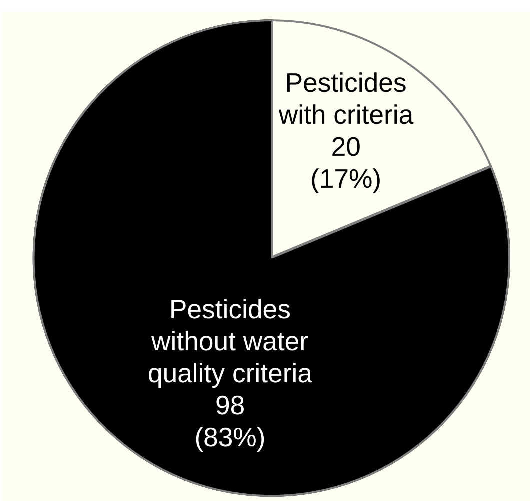
NA, not available.

[1] Aquatic life criteria set by the National Academy of Sciences and National Academy of Engineering or by the Canadian Council of Resources and Environment Ministers.
[2] Data from Anderson et al. (1997).
[3] Data from Bortleson and Davis (1997).
[4] Data from Wagner et al. (1996).
[5] Data from Dubrovsky et al. (1998).
[6] Data from Clark et al. (1998).

assigned aquatic life criteria (Larson et al. 1997). Of the 96 herbicides, 55 insecticides, and 30 fungicides that currently have the highest agricultural use in the United States, EPA has established aquatic life criteria for only 6 insecticides (Larson et al. 1997). EPA has not established any aquatic life criteria for the herbicides and fungicides most commonly used today.

The National Academy of Sciences (NAS) and National Academy of Engineers (NAE) set aquatic life criteria for a number of commonly used pesticides in 1973, but these are outdated. Their derivation was based on acute toxicity data but did not take into account bioaccumulation, sublethal effects, or synergistic effects.

Figure 1
Number of pesticides with aquatic life criteria.



(Based on 118 pesticides typically looked for in water quality studies)

Source: Larson et al. (1997)

The EPA and NAS/NAE criteria are commonly used as indicators of the degree of water pollution, but, as we shall see, may be far higher than the levels at which damage to fish can occur.

From the NAWQA studies, one can see that pesticide use is prevalent in our watersheds, and the water in our river basins contains dozens of pesticides. Exactly what happens to the hundreds of tons of pesticides put on the land each year?

Pesticides Persist in the Environment and Degrade into Toxic Products

Most pesticides undergo chemical transformations after application (Day 1991). The transformations may result from either physical processes, such as oxidation or photolysis, or biological metabolism by microbes. The breakdown products may be less toxic, of equal toxicity, or more toxic than the original compound. In many cases, the breakdown products of pesticides are not completely understood and their toxicities to aquatic life have not been studied. In surveys of pesticides in streams, the parent compound is typically looked for. When it is not found, it may be concluded that the pesticide has been cleared from the system. However, the breakdown products may still be present and constitute a danger to the organisms living there.

For example, Roundup, or glyphosate, has been publicized as an environmentally friendly herbicide that breaks down shortly after application. However, experiments have shown that glyphosate may persist in the environment for as long as 3 years (Torstensson et al. 1989). Its metabolite, AMPA, may persist even longer (World Health Organization 1994). Glyphosate is typical of many pesticides in that its breakdown is dependent upon the environmental conditions in which it is used and that the toxicity of its breakdown products is equal to or greater than the toxicity of glyphosate itself.

The rate at which a pesticide breaks down varies widely, depending upon the conditions of application. Degradation depends largely upon temperature. Pesticides such as glyphosate may oxidize in as little as 3 days in Texas or as long as a year in Canada. Conditions that favor microorganisms also tend to promote pesticide degradation (Barbash and Resek 1996), although in some cases, the presence of humic material in the soil stabilizes the pesticide (Chapman et al. 1981; Barbash and Resek 1996). Light and water are also important in the degradation of pesticides (Barbash and Resek 1996). Pesticides break down more quickly under bright sunshine than under cloud cover. Breakdown occurs more quickly under moist conditions than under dry conditions. Oxygen concentration is also important for metabolism of pesticides. For example, pesticides are resistant to breakdown in the anoxic, highly reducing muds of estuaries where there is an absence of oxygen (Barbash and Resek 1996).

Pesticides may remain in the environment much longer than expected or claimed, and the breakdown products may also be toxic to organisms.

In short, the timing of degradation of a pesticide is highly variable and dependent upon environmental conditions. Pesticides may remain in the environment much longer than expected or claimed, and the breakdown products may also be toxic to organisms. Fish and aquatic organisms must cope daily with a variety of pesticides or metabolites that may have been used years before. Many of the pesticides which were banned long ago still appear in water quality surveys. One aspect of pesticide degradation is quite clear. Pesticides and their metabolites do not magically disappear from the environment.

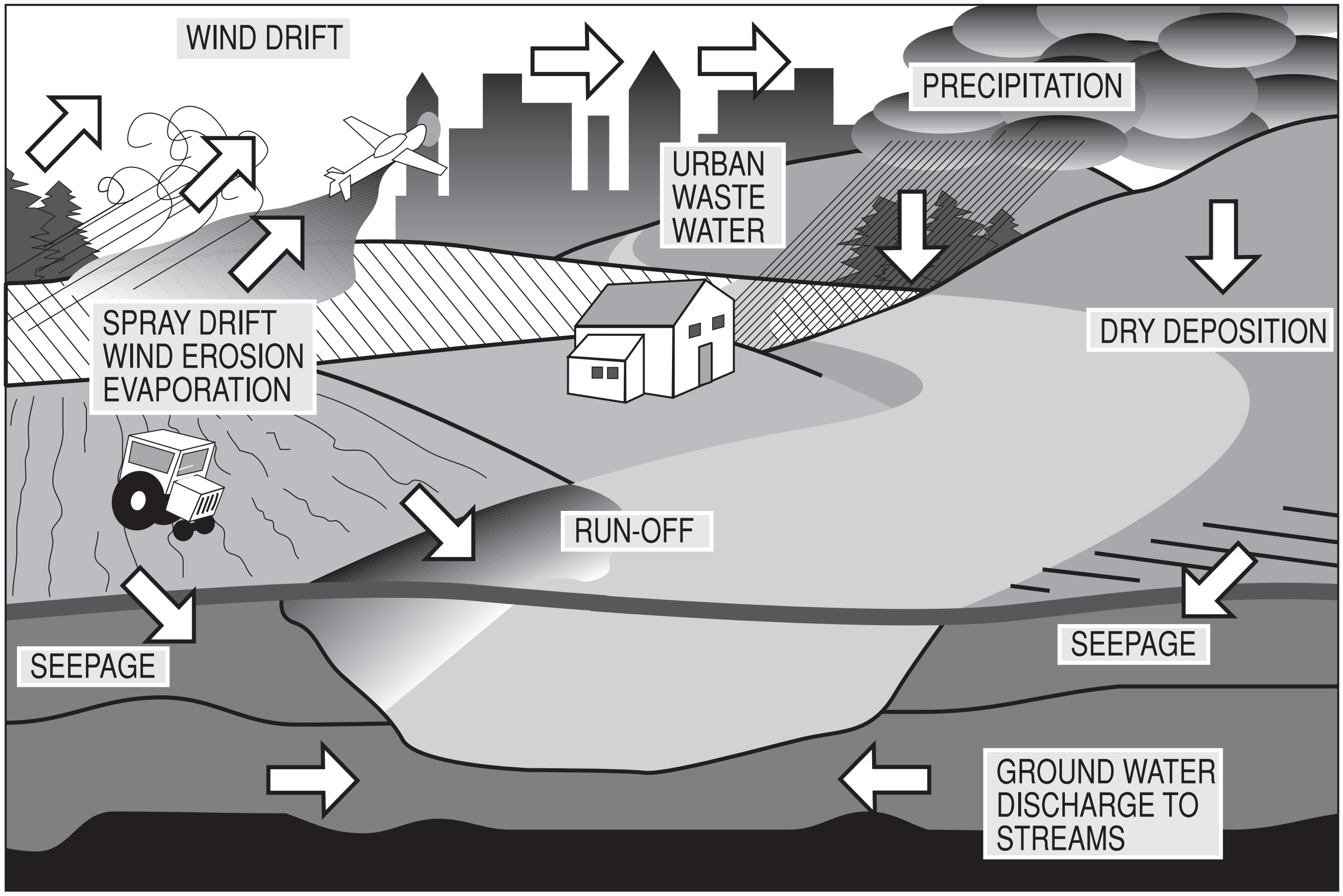
Pesticides Move throughout Watersheds

Water promotes the transport of pesticides from their site of application (see Figure 2). Streams are often the recipients of pesticide residues following rainfall. But the rapid downstream transport of chemicals by streams and rivers does not mean that pesticides have no effects on a variety of organisms living in the area of application. Pesticides in streams typically reach high levels for short periods of time after application, then decrease to very low or undetectable levels. During the brief period of high concentration, damage to

NEVA HASSANEIN

Pesticide residues move readily into streams and rivers following rainfall.

Figure 2. Routes through which pesticides move in the water cycle.



organisms in the stream may occur. Fish and wildlife at early stages of development are particularly vulnerable. Brief exposures to toxic materials at early stages in development may arrest future development of complete organ systems (Guillette et al. 1995). During early development, the sexual characteristics of the animal are formed, the immune system develops competence, and a variety of hormones regulate the formation of bone structure and organ systems. Disruption of any of these complex systems at critical times can lead to permanent impairment.

Pesticides in streams are rapidly diluted by riverflow so that soon after entry into a stream they can be detected only at very low levels or no longer be detected at all. While the concentrations in streams may be relatively harmless during rain and high water flow, under certain conditions pesticides are concentrated rather than diluted. Depending on the nature of the pesticide, they may attach to sediment particles, accumulate in the tissues of various organisms, or become buried in the sediment (Barbash and Resek 1996). Eventually, all probably reach the sea, where they become a problem of relatively unknown magnitude. If pesticides are applied in huge amounts each year for dozens of years, the dilution effect begins to disappear as the chemical concentrations increase in their particular final resting place. Thus, spraying of hillsides with a certain pesticide may not affect the animals in the immediate vicinity but may have disastrous effects on other populations downstream from the site of application.

Most pesticides reach the water bound to soil particles (Barbash and Resek 1996). Erosion, which in the United States is estimated to transport four billion tons of soil a year into waterways, is a major contributor of pesticides to rivers and river beds. Thus, poor land-use practices not only contribute buildup of silt to the spawning areas in streams which results in suffocation of eggs. Erosion may also deliver pesticides to the rivers at a time when the fish, as embryos, are most sensitive to their deleterious effects.

Pesticides can be taken up from the water and accumulated in tissues of aquatic organisms, often becoming magnified thousands of times higher in the organism than in the surrounding water.

Pesticide Concentrations Can Become Magnified in Tissues

Under certain circumstances, pesticides can be taken up from the water and accumulated in tissues of aquatic organisms, often becoming magnified thousands of times higher in the organism than in the surrounding water. Usually biomagnification is a function of the fat solubility of the pesticide (Norris et al. 1991). Biomagnification is thus dependent upon the metabolic state of the animal, such that fatter animals tend to accumulate more lipid soluble pesticides. This reaches an extreme in the lipid-rich eggs and embryos.

CORVALLIS ENVIRONMENTAL CENTER

Erosion, which in the United States is estimated to transport four billion tons of soil a year into waterways, is a major contributor of pesticides to rivers and river beds.

Eggs of aquatic organisms usually have enough fats in the form of yolk to meet early energy requirements during development. Lipid soluble pesticides accumulate rapidly in these fat deposits and are not easily removed. There they may compete with hormones provided maternally for use by the embryos (Bern 1990). The presence of environmental contaminants in the yolk that act as endocrine mimics thus may have a powerful influence on embryonic development.

Not only are pesticides taken up directly from the environment, fishes can also absorb them from food organisms. A study of DDT accumulation in brook trout (Macek and Korn 1970) found that ten times more of the available DDT and its metabolites were absorbed from food than from the water. This can be magnified still further as the pesticide travels through food chains. Metcalf et al. (1971) demonstrated this biomagnification effect convincingly with radioactive DDT in a model ecosystem. C^{14}-DDT was applied to *Sorghum* as the source for the pesticide. The *Sorghum* was then

Table 2. Bioaccumulation of various pesticides, pesticide additives, and pesticide contaminants in fishes.

Pesticide	Category	Bioaccumulation Factor	Species	Reference
Carbaryl (Sevin)	Insecticide	30	golden ide	1
Chlordecone	Insecticide	1100-2200	fathead minnow	2
Chlorothalonil	Fungicide	840	rainbow trout	3
Chlorpyrifos (Dursban)	Insecticide	1374	rainbow trout	4
Cypermethrin	Insecticide	700-1000	rainbow trout	5
Diazinon	Insecticide	60	carp	2
Dichlobenil (Casoron)	Herbicide	40	golden ide	1
Fenvalerate	Insecticide	40-200	salmon	6
Flucythrinate	Insecticide	3000-5000	fathead minnow	6
Hexachlorobenzene	Contaminant of chlorothalonil, picloram, and pentachlorophenol	5500	rainbow trout	7
Nonyl phenol	Surfactant	1300	stickleback	8
Pentachlorophenol	Insecticide	251-5370	rainbow trout	2
Permethrin	Insecticide	1700-3300	fathead minnow	6
		73	Atlantic salmon	9
Pentachlorophenol	Fungicide	100	rainbow trout	10
2,3,7,8-TCCD	Contaminant of Dacthal, and 2,4-D	28,000	rainbow trout	11

1. Freitag et al. 1985.
2. Howard 1991.
3. World Health Organization 1996.
4. Racka 1993.
5. Hill 1985.
6. Smith and Stratton 1986.
7. Veith et al. 1979.
8. Ahei et al. 1993.
9. McLeese et al. 1980.
10. Hattula et al. 1981.
11. Mehrle et al. 1987.

introduced into a system with several components that provided a food web. *Sorghum* was eaten by a salt marsh caterpillar. Excreta from the caterpillar was consumed by diatoms, which were subsequently eaten by nine species of plankton. The plankton was eaten by mosquito larvae which were subsequently eaten by *Gambusia,* the mosquito fish. After one month, 54% of the radioactivity was found in the fish, of which most was DDE, a breakdown product of DDT. The concentration of DDT was 84,000 times greater in the fish than in the water, while the concentration of DDE was 110,000 times greater in the fish than in the water. Biomagnification of concentrations of DDT and its metabolites has been well established (Woodwell et al. 1967; Risebrough et al. 1967), which was a major factor leading to its removal from the pesticide market in the United States.

Modern pesticides are usually more water soluble and do not accumulate in high concentrations in fat deposits. However, they do show bioaccumulation (Table 2). While these accumulations are not as dramatic as those of DDT, they show that the concentration of pesticide in the water may not be relevant in determining whether levels of pesticide will cause biological effects. These accumulations are usually measured in whole animals. Localized concentrations of pesticides at the cellular level may be extremely high, but this is an area that has not been widely explored.

Without complete analysis of the pesticides and their breakdown products in aquatic organisms in a natural state, we are unable to determine the exact concentrations to which they are subjected. If aquatic life has been exposed for long periods of time, the concentrations within the tissues may be much higher than that of the surrounding water.

Bioaccumulation and biomagnification properties of pesticides in tissues represents a problem in our assessment of their toxicity. Without complete analysis of the pesticides and their breakdown products in aquatic organisms in a natural state, we are unable to determine the exact concentrations to which they are subjected. If aquatic life has been exposed for long periods of time, the concentrations within the tissues may be much higher than that of the surrounding water. In addition, exposure to many types of pesticides may lead to interactions between them that increase their toxicity. Little work has been done on the interactions between different pesticides, but available evidence suggests that these interactions can increase toxic effects (Koenig 1977; Cook et al. 1997).

Fish Kills and Acute Toxicity of Pesticides

Pesticides are capable of killing salmonids and other aquatic organisms quickly. These short-term, acute toxicities to fish have been studied extensively for most chemicals used in forestry, agriculture, manufacturing, and the home. Much of this information is provided by the chemical manufacturer in order to meet requirements by the Environmental Protection Agency and various state agencies.

Most of these acute toxicity studies report lethal amounts as LC_{50}s, the concentrations of chemicals that kill 50% of the test animals within 48 or 96 hours, or LD_{50}s, the doses of chemicals in milligrams per kilogram body weight of the test animal which kill 50% of the animals within 48 or 96 hours. Organisms used for these tests range from algae to mice and rats. Rainbow trout are a common subject, as are bluegills, fathead minnows, mosquitofish (*Gambusia*), and zebrafish (*Brachydanio rerio*). A number of LC_{50}s for various fish are available in the literature (e.g., Norris et al. 1991; Anderson et al. 1997) or on the internet (e.g.,http://ace.orst.edu/info/extoxnet/pips/ghindex.html). In the interest of space, they will not be reproduced here.

The herbicide acrolein killed approximately 92,000 steelhead, 114 juvenile coho, 19 resident rainbow trout, and thousands of nongame fish in the Rogue River Basin.

Acute toxicity studies are usually performed on subadult or adult fish. Few of these studies have used eggs or fry, even though studies from the Great Lakes have shown that exposure during this stage of life can lead to profound results. Cook et al. (1997) found that the toxicities of congeners of TCDD injected into eggs of lake trout were additive and effective at much lower dosages than with juvenile fish. In general, embryonic stages are the most sensitive to environmental pollutants (Guillette et al. 1995). This is an area of research in western salmonids which has been overlooked in the past and needs immediate consideration.

When pesticides in water supplies exceed their lethal concentrations, the results are immediate. Large numbers of fish are killed, and these are reported to a variety of federal and state agencies. The spill is cleaned up when possible and the responsible parties are fined. An example of this regulatory action comes from southern Oregon. The Rogue River has received a number of inadvertant spills of pesticides, particularly acrolein, an herbicide used for removal of aquatic vegetation. This herbicide is very toxic to fish. Lorz et al. (1979) reported a 1977 release of treated irrigation water containing Magnicide H, a gaseous form of acrolein, into the Rogue River within 24 hours of treatment instead of the recommended holding time of 6 days. A 10 mile section of the river was affected. ODFW officials estimated that 238,000 fish were killed, including 42,000 salmonids with an estimated value of $284,000.

On May 9, 1996, another large fish kill occurred in a four-mile stretch of Bear

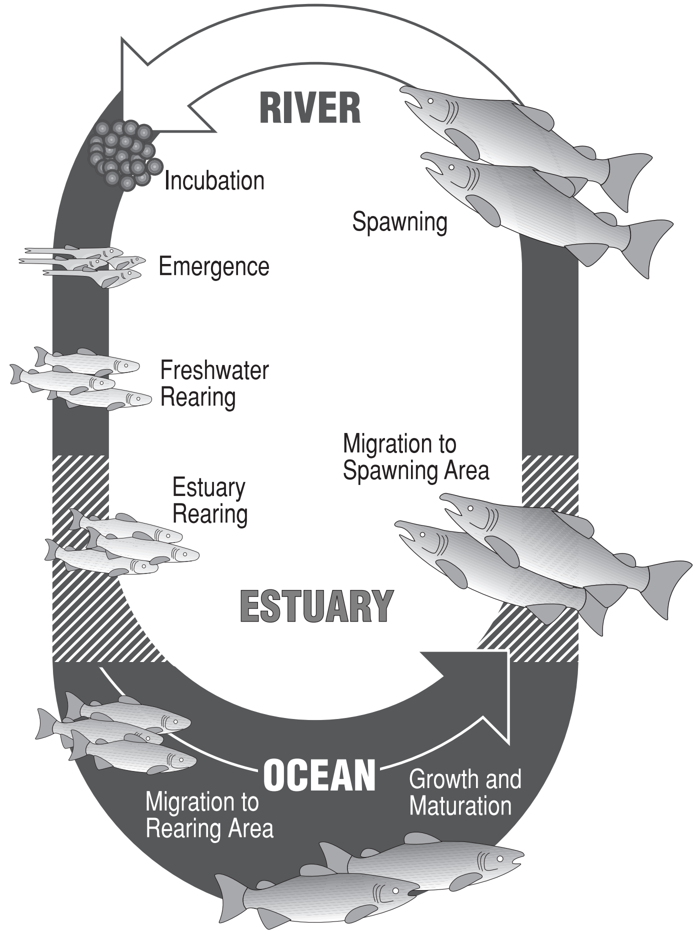
Creek, a tributary of the Rogue River, Oregon. A head gate was found open on an irrigation canal which had been treated with acrolein. The acrolein was used to remove aquatic vegetation that grew in the canal and interfered with its operation. Approximately 92,000 steelhead, 114 juvenile coho salmon, 19 resident rainbow trout, and thousands of nongame fish were killed within a short period of time after exposure. Talent Irrigation District was fined $356,000 for the loss of steelhead by Oregon Department of Fish and Wildlife. They were also fined $50,000 by the Oregon Department of Environmental Quality and an additional fine of $407 by the Oregon Department of Agriculture for allowing the pesticide to enter the stream (Evenson 1998).

Deaths of threatened and endangered species from inadvertant contamination of waterways are of grave concern because the loss of each individual in a sensitive population makes recovery efforts that much more difficult.

Deaths of threatened and endangered species from inadvertant contamination of waterways are of grave concern because the loss of each individual in a sensitive population makes recovery efforts that much more difficult. Fortunately, these events are relatively infrequent. By contrast, pesticide contamination at sublethal levels are probably an even greater danger to salmonid populations because the contamination is poorly regulated, the mortalities go unseen, and the consequences are unknown.

Behavioral Effects of Pesticides at Sublethal Concentrations

Figure 3
Salmon life cycle.



Salmon and many trout have developed anadromy, a behavior pattern in which juveniles migrate to the sea. In the sea, juvenile fish feed and grow before returning to the stream where they were born to lay their eggs.

Behavior of salmonids is specifically directed at the continuance of the species. Each aspect of behavior has been highly selected over the generations to contribute to survival. (See Figure 3.) Interruptions of these behavior patterns will therefore reduce survival and diminish populations.

Pesticides at sublethal concentrations have been shown to disrupt many of the behavior patterns of both juvenile and adult salmonids and probably have deleterious impacts on their survival. Examples of these disruptions are presented below, although their impact on survival of fish populations is unknown at this time.

Pesticides Impair Swimming Performance

The ability to feed, to avoid predators, to defend territories, and to maintain position in the river system are all dependent upon the swimming ability of the fish. A decrease in swimming performance reduces the ability of the fish to survive and compete with others. Swimming performance is commonly measured in swim tubes where the fish is forced to swim against a gradually increasing current. The current velocity at which the fish loses its position is referred to as critical swimming speed. The time of swimming at a particular velocity before it loses its position is referred to as swimming stamina. Both are used as indications of swimming performance.

Exposure to sublethal concentrations of pesticides often causes a loss in swimming performance. The fungicide TCMTB, which is used to prevent fungal staining of logs, causes deleterious effects on the swimming ability of salmonids. Chinook salmon and rainbow trout juveniles exposed to 5-20 ppb TCMTB for 48 hours, then removed to clean water for 12 hours, showed a dose-dependent reduction in critical swimming speeds (Nikl and Farrell 1993). A further study with coho salmon at the same concentrations indicated that the reduction in critical swimming speed was dependent on both the concentration of TCMTB and the time of exposure to the chemical (Nikl and Farrell 1993). Damage to gill structure also increased with concentration and time of exposure. The researchers speculated that gill damage prevented respiratory exchange and that the decrease in oxygen availability was at least partially responsible for the decrease in swimming speed.

Colquhoun et al. (1984) tested the swimming stamina of brown trout exposed to the insecticide naled for 24 hours at a concentration of 84 ppb. The stamina of the exposed fish was reduced 57% compared to controls. Paul and Simonin (1996) felt that this exposure time was unrealistic when compared to the actual exposure times from aerial application of pesticides over streams. They held brook trout in water containing 23 and 46 ppb naled for 6 hours and found no change in swimming stamina. However, they found that both an insecticide formulation containing resmethrin and Scourge, a synergised formulation with resmethrin and piperonyl butoxide, did affect swimming stamina. Brook trout that were held in solutions of 3.2 ppb synergised and non-synergised resmethrin for 6 hours had significantly reduced stamina. Little et al. (1990) examined behavior of rainbow trout exposed for 96 hours to sublethal concentrations of five agricultural chemicals: carbaryl, chlordane, 2,4-D amine, methyl parathion, and pentachlorophenol. All chemicals inhibited spontaneous swimming activity and swimming stamina.

Rainbow trout exposed to tributyltin oxide, an anti-fouling agent used in boat paint, lost their ability to orient themselves in a current (Chliamovitch and Kuhn 1977). Rainbow trout exposed to Aqua-Kleen, 2,4-D butoxyethanol ester, also lost orientation to currents with increasing concentrations (Dodson and Mayfield 1979a). At near lethal concentrations, the orientation became variable. The most evident behavior change was a lethargy that increased with greater concentrations of the herbicide.

CORVALLIS ENVIRONMENTAL CENTER

Studies show that low concentrations of certain pesticides impair the swimming performance of rainbow trout.

Half of juvenile Atlantic salmon subjected to 1 ppm of the insecticide fenitrothion for 15-16 hours lost the ability to maintain territories for a period of six days after exposure (Symons 1973). Exposure of rainbow trout to 46 ppb of Vision, a glyphosate herbicide, for one month caused increased aggressive behavior (Morgan and Kiceniuk 1992). Rainbow trout exposed to the synthetic pyrethroids, fenvalerate and permethrin, tended to swim at the water surface, coughed repeatedly (attempting to clear the gill surface) and increased gill movements, suggesting respiratory distress (Holcombe et al. 1982). Those exposed to the organophosphate pesticides, Dursban (chlorpyrifos) and disulfoton, became darker and showed indications of muscle spasms and skeletal deformations. Both pesticides are cholinesterase inhibitors and the reactions were attributed to blocked neural transmission. Trout exposed to the chlorinated hydrocarbon, Kelthane (dicofol), became dark, lethargic, and showed signs of respiratory distress, suggesting both respiratory and neurotransmission problems (Holcombe et al. 1982).

These experiments show that pesticides at very low concentrations can interfere with swimming performance in salmonids and that these effects can last for some time after exposure to the chemicals.

These experiments show that pesticides at very low concentrations can interfere with swimming performance in salmonids and that these effects can last for some time after exposure to the chemicals. The evidence comes from laboratory experiments and would be very difficult to replicate in the field. If similar effects occurred in field situations, however, interference with swimming performance would result in increased death from predation.

Pesticides Can Increase Predation on Juvenile Salmonids

The end result of any decrease in swimming ability will be an increase in predation upon the juvenile salmonids. If a young salmonid is slowed in its escape responses by ion imbalances, metabolic problems, or disease, it becomes prey for a number of hungry predators. Its demise is not immediately evident without sophisticated predator tests. Several studies have demonstrated that pesticides can indeed increase predation.

For example, Kruzinski and Birtwell (1994) found that juvenile coho salmon exposed to sublethal concentrations of the fungicide TCMTB showed fright and escape responses similar to controls in a simulated estuarine environment. After five days, however, TCMTB-treated fish had been preferentially consumed by introduced rockfish 5.5 times more frequently than controls. In another study, after a 24-hr exposure to 1.0 ppm fenitrothion, Atlantic salmon parr were more vulnerable to predation by brook trout than controls (Hatfield and Anderson 1972). Exposure to DDT had no effect on the predation. Little et al. (1990) found that carbaryl and pentachlorophenol increased vulnerability of rainbow trout to predation.

Predator avoidance tests with coho (Olla et al. 1992) and chinook salmon (Olla et al. 1995) indicated that stresses causing increased plasma cortisol levels resulted in susceptibility to predation, but that predator avoidance in coho salmon returned within 90 minutes (Olla et al. 1992) or 4 hours (Olla et al. 1995). In chinook salmon, predator avoidance was recovered in 24 hours after exposure to stress. While acute stresses seem to render the juveniles susceptible to predation, the effects of chronic stresses from pesticide exposure are not well known. Exposure of juvenile coho salmon to sublethal levels of two triclopyr herbicides for four hours did not result in significant changes in secondary stress effects, such as respiration, plasma glucose and lactate, and hematocrit (Janz et al. 1991). No plasma cortisol levels were measured, but these usually correlate well with plasma glucose concentrations. However, exposure to sublethal concentrations of the butoxyethanol ester of 2,4-D caused enlargement of the interrenal gland, an indication that cortisol production was increased (McBride et al. 1981). If chronic exposure to pesticides does cause increase stress responses in salmonids, the studies of Olla et al. (1992, 1995) suggest that predation will be much greater in exposed fish.

Temperature Selection is Changed by Pesticides

Most organisms respond to the presence of pathogens through temperature selection. Cold-blooded animals such as lizards or fish typically seek warmer environments to increase metabolic repair when subjected to toxicants or infections. In salmonids, studies showed that the temperature selected depended on the concentration of the pesticides to which the fish were exposed. Juvenile Atlantic salmon exposed to low doses of DDT selected lower temperatures than controls (Ogilvie and Anderson 1965; Peterson 1973). At higher doses of DDT, exposed fish selected higher temperatures than controls. This effect was accentuated in fish acclimated to warm temperatures (17^0 C) compared to fish acclimated to cool temperatures (8^0 C). Fish acclimated to 17^0 C and exposed to 10 ppb or higher of DDT became hyperactive when introduced into cold water. Brook trout exposed to isomers of DDT and derivatives showed similar changes in temperature selection (Miller and Ogilvie 1975; Gardner 1973). The alteration in temperature selection by subyearling Atlantic salmon exposed to sublethal concentrations of DDT persisted for at least a month after the fish were transferred to clean water (Ogilvie and Miller 1976). Exposure to the organochlorine insecticide aldrin at a concentration of 100-150 ppb also caused Atlantic salmon juveniles to select lower water temperatures (Peterson 1973).

These results suggest that salmonids exposed to some pesticides select temperatures according to the concentration of pesticide. At low concentrations, fish try to lower their body temperature to minimize the effects on physiological processes. As the concentration increases, they respond by seeking warmer water to stimulate detoxification. These effects can last for a considerable time after the pesticide is removed, causing the fish to seek abnormal water temperatures and thus subjecting them to increased dangers of disease and predation.

Disruption of schooling behavior by many pesticides suggests that these compounds may increase predation upon juvenile salmonids and lead to population losses that would be very difficult to detect by conventional fisheries techniques.

Schooling Behavior is Reduced by Pesticides

Schooling behavior is a tactic to reduce predation. Predators are presented with a large mass of fish from which it is difficult to focus on a single individual (Cushing and Harden-Jones 1968). Seaward migration of juvenile salmonids is accompanied by schooling responses, probably to reduce losses to the population during the migration.

Many pesticides disrupt schooling behavior. Drummond et al. (1986) found that loss of schooling behavior in fathead minnows was the most sensitive indicator of stress for 133 of 139 organic chemicals tested, including a number of pesticides. Exposure to sublethal concentrations of the fungicide TCMTB interferes with schooling behavior in chinook salmon (Kruzinski and Birtwell 1994). Exposure to methoxyclor caused a progressive increase in inter-fish distance in brook trout (Kruzinski 1972). Holcombe et al. (1982) found that schooling behavior was disrupted in fathead minnows exposed to sublethal concentrations of Kelthane, Dursban, disulfoton, fenvalerate, and permethrin.

Disruption of schooling behavior by many pesticides suggests that these compounds may increase predation upon juvenile salmonids and lead to population losses that would be very difficult to detect by conventional fisheries techniques. The prevalence of this effect by organic chemicals (Drummond et al. 1986) suggests that this sublethal effect may be more widespread than we might suspect.

Pesticides Can Interfere With Seaward Migration

Juvenile salmon rearing in streams reach a stage where their color changes from brown to silver. This is the stage of parr-smolt transformation where a variety of morphological, physiological, and behavioral changes occur. In addition to the silvery color, the smolts lose their territorial behavior, begin to school, and to migrate downstream to the sea. The mechanisms of this process have been widely debated and have been the subject of many local and international workshops.

The effects of pesticides on seaward migration have not been extensively stud-

INSTITUTE FOR FISHERIES RESOURCES

Estuaries are a critical transition zone where salmon adapt to saltwater.

ied because of the difficulty of measurement of migration tendency. Lorz et al. (1979) studied the effects of three herbicides on seaward migration in coho salmon. Diquat at concentrations from 0.5 ppm to 3.0 ppm inhibited seaward migration in a dose-dependent manner. Dinoseb inhibited migration only at the high (but sublethal) concentrations of 40 and 60 ppm. Tordon 101 (a mixture of picloram and 2,4-D) caused a slight inhibition of migration at the highest levels studied (1.2 and 1.8 ppm), but the results were not significantly different from controls. Although the herbicides were not lethal to the fish at the concentrations used, the levels were very high (ppm) compared to other concentrations of pesticides (ppb) described in this section. This may be due to the low toxicity of most herbicides, compared to other classes of pesticides. More work on seaward migration should be done with other pesticides, especially the organochlorine and organophosphate insecticides commonly found in western streams.

Pesticides Interfere with Seawater Adaptation

The end result of seaward migration is the entry of juvenile salmon into an estuary and the tranformation of the fish's osmoregulatory mechanisms from a fresh water to a saltwater environment. These osmoregulatory processes permit the fish to maintain required water and ion concentrations in various organ systems. The stresses involved with this transition can be surmised from the large mortality (usually about 90% or greater for hatchery smolts) which results from seawater entry (Pearcy 1992). Death can result from several causes. The juvenile fish may simply not make the transition to seawater and die from osmoregulatory problems. The juvenile fish may be slowed by the osmotic stresses and be eaten by birds or predatory fish. The immune system of the fish may be compromised by the osmotic stresses, permitting the onset of disease and subsequent predation by piscivorous birds or fish.

Because the transition from fresh water to seawater is a critical period in the survival of the juvenile salmon, one would expect pesticides to have some of their greatest effects at this stage.

Most salmonids spend a period of transition in the estuary where they seek the salinities they prefer. Many utilize the less saline estuarine surface water which stratifies over deeper more saline water (Iwata et al. 1982; Birtwell and Kruzinski 1989).

Few studies have looked at the salinity preference of fish exposed to sublethal levels of pesticides. One would expect that the stress from pesticide exposure might cause the fish to select salinities comparable to that of their blood to relieve the extra burden of osmotic stress. Coho salmon in the stratified salinity tank set up by Kruzinski and Birtwell (1994) sought the freshwater wedge if they were previously exposed to TCMTB. Mosquitofish, however, sought a higher salinity if exposed to DDT (Hansen 1972). No change in salinity preference was induced by malathion (Hansen 1972).

In a study on the effects of herbicides on juvenile coho salmon, three of the thirteen herbicides studied, Amitrole-T, diquat, and paraquat, caused dose-dependent mortality in seawater tests (Lorz et al. 1979). An interesting phenomenon occurred in fish exposed to Tordon 101 (a mixture of 2,4-D and picloram). In December, fish were tested at eight different concentrations. The two lowest concentrations, 0.25 and 0.5 ppm, caused increased mortalities, while at higher concentrations up to 20 ppm all fish survived. When this was repeated in March, fish exposed to 20 ppm Tordon 101 all died in seawater. The researchers could not explain the unexpected mortalities at the low concentrations but suggested that pre-smolted coho salmon in December were less sensitive to environmental pesticides than those in smolted condition in March (Lorz et al. 1979). Mitchell et al. (1987) found that a 10-day exposure to 2.78 ppm of the herbicide Roundup (a glyphosate formulation) did not affect seawater adaptation or growth.

Because the transition from fresh water to seawater is a critical period in the survival of the juvenile salmon, one would expect pesticides to have some of their greatest effects at this stage. However, studies showing relationships between pesticides and osmoregulatory problems leading to mortality in salmonids are few. Most of the studies that have been done have focused on marine species or euryhaline species which do not undergo the radical transformation of osmoregulation experienced by salmonids and other anadromous species. Studies on the effects of pesticides, especially organochlorine and organophosphate pesticides, on salmonids are needed in this area.


OREGON SEA GRANT

An adult chinook salmon that is ready to spawn.

Pollutants Affect Adult Returns

The result of interference with life history mechanisms of the juveniles should be the decrease in numbers of returning adults. Because the sublethal effects of pesticides on fish survival have not been fully appreciated, there are few studies that have correlated the return of adult salmon with the extent of pesticide use in the river basin. In New Brunswick, however, researchers have found that decreased returns of Atlantic salmon have been correlated with the use of pesticides in the area. Specific concern has focused on the compounds known as nonyl phenols, which are used not only as wetting agents in pesticides, but also in the production of paper, textiles, and petroleum products. Nonyl phenols are thought to act both by endocrine disruption and by interference with the normal adaptation of salmon smolts to seawater. While the evidence is convincing that the salmon runs have been affected by spraying, the experimental evidence has not yet been published in the scientific literature. A paper describing these effects has been submitted for publication (Fairchild 1999).

Migration patterns and timing of adult salmon may be altered by pollutants as they re-enter fresh water during their spawning runs. Elson et al. (1972) showed that radiotagged Atlantic salmon entering the unpolluted Tabusintac estuary in New Brunswick moved quickly up the estuary and into the river system. In contrast, salmon entering the Miramichi estuary, a heavily industrialized estuary, spent time swimming back and forth in an attempt to avoid heavily polluted water. They were never found in a channel on the northwest side of Beaubears Island, where a marsh at the lower end of the island served as a reservoir for effluent from a wood-preservative plant. This effluent seeped into the river at high tides. Salmon clung to the cleanest side of the river as they swam upstream past the industrial area. Clearly, the fish were slowed in their spawning migration as they searched for the cleanest parts of the river.

Respiration of adult coho salmon was examined as they attempted to pass a polluted estuary within the city limits of Seattle, Washington (Smith et al. 1972). They found that respiration decreased and blood lactate increased during swimming in the polluted area. They suggested that the salmon either avoided the areas of pollution, resulting in delayed passage times, or swam through the polluted areas, resulting in an oxygen debt and decreased energy reserves. Decreased energy reserves could result in smaller eggs and insufficient energy for passage into upstream spawning areas.

In summary, these studies on the disruption of behavior of salmonids by pesticides suggest that the presence of pesticides can have profound effects on the survival of the juveniles. Every aspect of their biology can be affected. The studies reported here consider mainly the freshwater portion of the salmonid life history. Exposure to pesticides continues in the estuaries and in the sea. For example, a major food source for juvenile salmon in the estuary is the amphipod *Corophium,* which spends the day buried in sediments which may be the final repository for many of the pesticides applied far upstream. It is likely that the amphipods accumulate pesticides and transfer them to the salmonids when eaten. Few studies have documented the behavior of juvenile salmonids once they reach the estuary. Even fewer have examined the effect of pollutants and pesticides on their behavior and survival.

Compromised Immune Systems

Fish in polluted waters are subject to more frequent and more severe outbreaks of disease because many synthetic chemicals tend to suppress the immune system (Moller 1985; Repetto and Baliga 1996). Immune systems protect fish from serious diseases caused by external bacteria, parasites, and viruses. The complexities of these systems are regulated and coordinated by a variety of compounds that act similar to hormones but that do not quite fit into the classical definition (Bern 1990). Most texts on molecular endocrinology include interleukins, cytokines, epithelial growth factors, and other compounds that influence the immune system as hormones (Bolander, 1994). These immunohormones can be compromised in various ways by environmental pollutants (Anderson 1996; Repetto and Baliga 1996).

Fish in polluted waters are subject to more frequent and more severe outbreaks of disease because many synthetic chemicals tend to suppress the immune system.

Although considerable work has been done on the immune system of fish, our understanding is still far from that we have gained about the intricacies of the human and rat immune mechanisms. As in mammals, fish have both cellular and humoral (antibody) responses to foreign agents, and exert both the B cell and T cell responses to these stimuli. It is generally agreed that size rather than age of fish determines the immunological maturity (Tatner 1996). In rainbow trout, a gradual increase in immune competence occurs during early growth, with non-specific immunity developing first, followed by cell-mediated and then humoral immunity. Full humoral immunity, the ability to produce circulating antibodies, develops at about 1 g in size.

A number of natural and anthropogenic factors suppress the function of the immune system. Stresses from habitat changes, poor water quality, lack of food, and pollutants all result in immunosupression and an increase in disease (Pickering 1993).

Information on immunosuppression in salmonids due to pesticides is scanty, although other fish have received more

CORVALLIS ENVIRONMENTAL CENTER

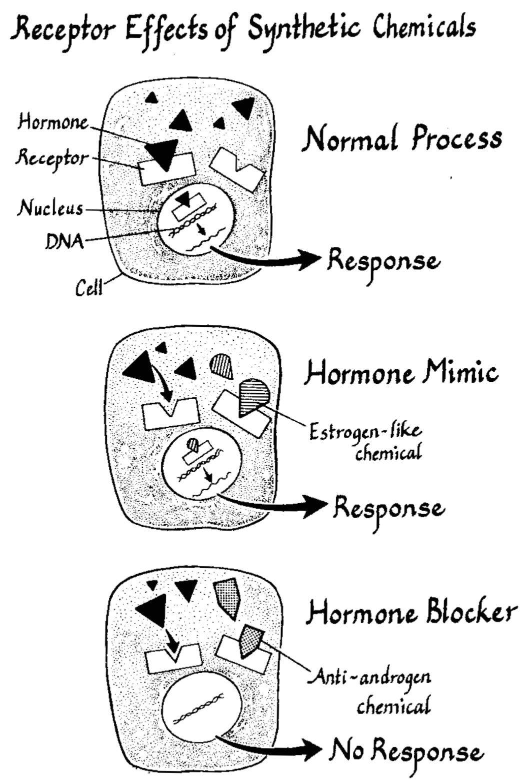
attention (Anderson 1996; Iwama and Nakanishi 1996). The insecticide endrin reduced the activities of phagocytes and antibody producing cells in rainbow trout (Bennett and Wolke 1987). Organotin compounds caused immune suppression in rainbow trout (O'Halloran et al. 1998). Although tributyltin is considered more toxic to fish than its breakdown product, dibutyltin, the latter was the more potent immunotoxin. Phenol and the PCB Arochlor 1254 were found to reduce antibody producing cells in rainbow trout (Anderson et al. 1984) and coho salmon (Cleland et al. 1989). TCDD caused mitogenic suppression in rainbow trout (Spitsbergen et al. 1986) and increased susceptibility to infectious hematopoetic necrosis virus (Spitsbergen et al. 1988).

The timing of exposure to pollutants may be an important factor for the health of the immune system in anadromous salmonids. Maule et al. (1987) found that immune suppression occurred to a greater extent during smolting of coho salmon. This may be due to the higher levels of the stress hormone, cortisol, found during smolting in salmonids (Specker and Schreck 1982; Thorpe et al. 1987). Cortisol is known to suppress the immune system of salmonids (Maule et al. 1987; Pickering and Pottinger 1989; Maule and Schreck 1990). It is likely that the effects of exposure to immunosuppressant chemicals during the period of seaward migration may be compounded by the high levels of plasma cortisol in the fish at that time. Many pesticides cause some degree of immunosuppression in mammalian systems (Repetto and Baliga 1996). It seems likely that similar effects may be found in the fish immune systems. Before the necessary research provides answers, it seems prudent to err on the side of caution and assume that similar immunosuppression by pesticides can be found in salmonids.

Endocrine Disruptors

In the late 70s, researchers Mike Howell and Ann Black found that their seine hauls from a small coastal stream in the panhandle of Florida contained mosquitofish with a strange characteristic. They were all males (Howell et al. 1980). On further inspection, another oddity was noted. Many of the apparently male fish had a gravid spot on their gonopodium, which normally indicates a female with internally developing young. Further analysis in the laboratory indicated that many of the apparent males were in fact masculinized females. A few months later, a second population of masculinized females was discovered in the Fenholloway River of coastal Florida (Bortone and Davis 1994), a stream approximately 300 kilometers east of the first stream. This collection included not only mosquito fish but also the least killifish and the sailfin molly. In both streams, the masculinized females were found downstream of paper mills. Some component of the effluent from paper mills caused the females to acquire masculine traits.

This research was one of many studies conducted over the past twenty years that indicated that certain synthetic chemicals, including many pesticides, can alter the sexual endocrine (or hormone) system of wildlife (Colborn and Clement 1992). The effects of these pseudo-hormones can vary considerably. In the above example, female fish became masculinized. Other pollutants act as artificial female sex hormones or estrogens. In still others, sexual expression is not affected. Instead, the chemicals interfere with bone formation during early development and result in stunted or malformed adults. As scientists gain greater insights into the complex workings of endocrine systems, they have come to appreciate the delicate balance of hormones needed to direct the early development of an embryo and the profound effects that endocrine mimics may produce.

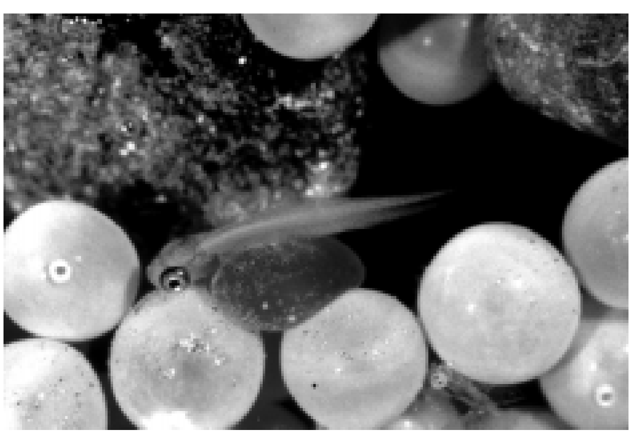


Endocrine-disrupting chemicals can interfere with normal functioning of the body's hormone system in a variety of ways. By mimicking or blocking hormones, the synthetic chemicals disrupt normal cellular communication, altering or preventing the appropriate biological response. Some endocrine disruptors can also interfere with chemical messengers involved with the immune system, hormone synthesis, neurotransmitters, and growth processes.

K. BORN

Endocrine systems in an organism control production of hormones, which in turn control many of its functions. An endocrine system is a complex network of positive effectors and feedback loops that carry messages from one part of the body to another to keep the body functioning properly. Initial input to the system comes from the environment in the form of temperature, light, or other sensory stimuli. These stimuli are recognized and integrated in the brain. Through a number of pathways, brain centers control the synthesis of primary hormones in the pituitary, a small bud of nerves and tissues at the base of the brain. In the reproductive cycle, the pituitary produces gonadotropins which act on the gonads to produce steroid hormones: androgens predominating for males and estrogens predominating for females. Steroids and other hormones travel through the blood to seek specific receptors in the cytoplasm of responsive cells. Each sex contains both male and female receptors in its cells so that the relative concentration of androgens and estrogens are the determining factor for sexual differentiation. When the steroid binds to the receptor protein, the steroid-receptor complex finds its way to the nucleus where it binds to DNA response elements to stimulate or regulate the production of steroid specific proteins.

Certain pesticides and pollutants can mimic the effects of estrogens, causing feminization of male animals.



OREGON DEPT. OF FISH AND WILDLIFE

During early development, fish are especially vulnerable to pesticides that can mimic natural hormones.

In such a complex system, pesticides and their derivatives can interact at many levels in the brain-pituitary-target organ axis with a variety of effects. These effects can manifest themselves through behavioral, reproductive, immunological, or physiological changes (Bern 1990). Our knowledge of the hormonal relationships to many of these functions is not well enough defined at present to exactly determine what effects pollutants may have on these functions. However, our knowledge of the female endocrine system and the effects of pesticides on its function has grown rapidly in recent years, and numerous studies have defined some of the unwanted effects.

Pesticides Can Mimic Estrogens

The sex of fishes can be determined relatively late in development. For example, applications of testosterone during the egg and fry stages of salmonids can convert the young fish completely to males or cause them to become sterile, depending on the concentration. Applications of estradiol, the female sex hormone, can cause production of 100% female salmon (Hunter and Donaldson 1983). In chinook salmon, Piferrer et al. (1994) suggested that aromatase, an enzyme that converts androgens to estrogens, acts as a regulatory switch for the process of sex differentiation. They showed that a treatment of young salmon with an aromatase inhibitor for only two hours could transform genetic females into functional males. These transexual males had testes indistinguishable from normal males, underwent all stages of sperm maturation, and were later used to fertilize females to produce all female offspring. Concentrations of androgens and estrogens circulating through the blood provide a stimulus for induction of levels of aromatase activity (Callard and Callard 1987). With a system of sex differentiation in fishes that is labile (i.e., likely to change), the potential for major impacts from estrogen mimics may be quite large.

Certain pesticides and pollutants can mimic the effects of estrogens, causing feminization of male animals. Actions of pesticides that result in feminization in fish are thought to occur in three ways:

1) Estrogen mimics compete with estradiol, the female sex hormone, for its receptor protein. The complex that is formed enters the nucleus, binds the DNA response elements, and stimulates the responses expected from normal female sex hormones. This has little effect in females but stimulates inappropriate responses in males, which have only very low concentrations of circulating estrogens. The result is feminization of the males.

2) Antiandrogenic compounds compete with androgens, or male sex hormones, for the androgen receptor protein. Once bound, the complex enters the nucleus but is not able to bind to DNA response elements. Consequently, no androgenic response is stimulated. In the absence of an androgenic response, the fish assumes that it is a female and the result is feminization.

3) Compounds may interfere with neurotransmitters or signal transduction mechanisms in the brain. The interference results in an upset of the regulatory mechanisms that distinguish males from females and sends the wrong messages to the pituitary. The result may be the release of gonadotropins that stimulate feminization of the juvenile fish.

OREGON DEPT. OF FISH AND WILDLIFE

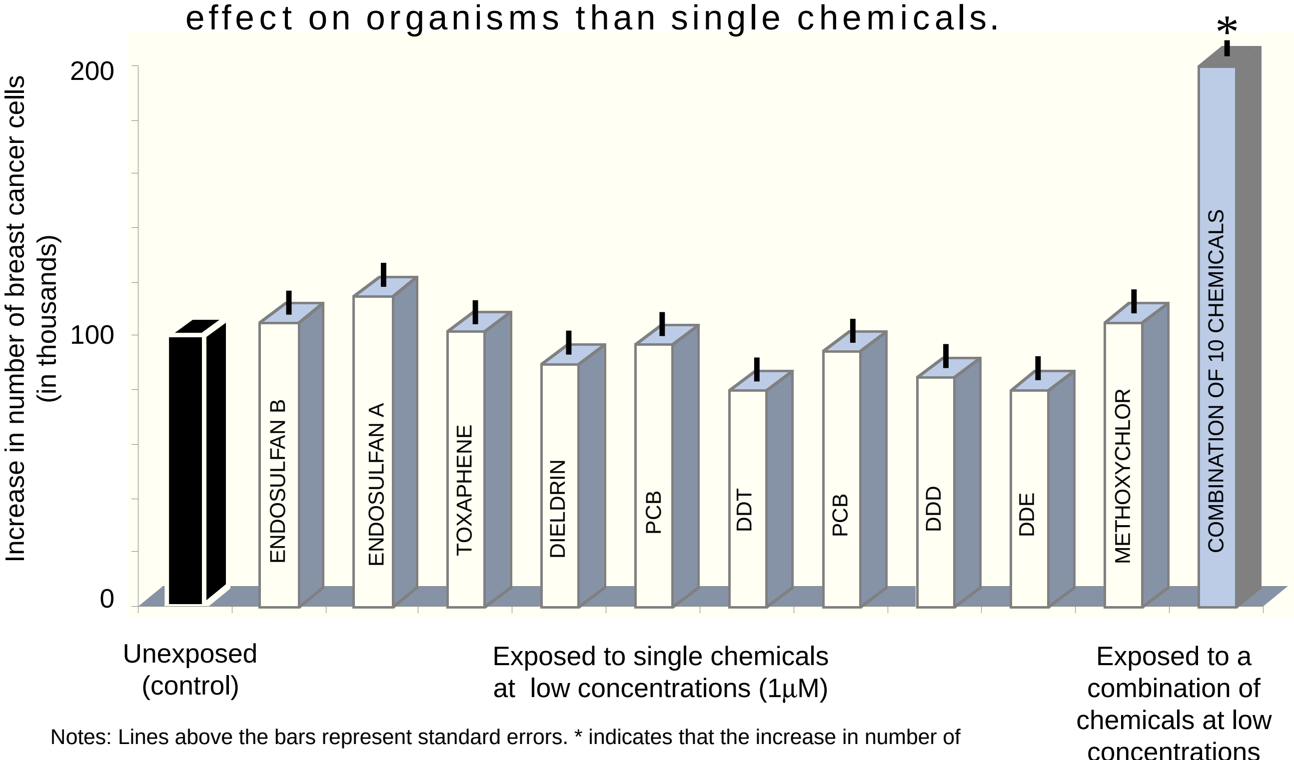
Pesticide exposure at the fry stage can cause feminization of male fish or even sterility.

The best known examples of feminization by competition of pesticides for estrogen receptors have occurred with alligators in the Southeast (Guillette et al. 1995), sea gulls (Fry and Toone 1981) and Caspian terns (Fox 1992). Increasing numbers of chemicals are found to have estrogenic activity in fish (Jobling and Sumpter 1993; Jobling et al. 1995). Unknown chemical contaminants from sewage effluents have been shown to produce estrogenic responses in male carp (Folmar et al. 1996). DDT and its long-lived breakdown product, o,p′-DDE, have been shown to bind estrogen receptors to provide weakly estrogenic responses, whereas p,p′-DDE does not bind estrogen receptors except at high concentrations (Donohoe and Curtis 1996). p,p′-DDE has a higher affinity for androgen receptors (Kelce et al. 1995), suggesting that the binding may be nonspecific for a number of steroid receptors. Lindane, chlordecone (Donohoe and Curtis 1996; Thomas and Khan 1997), alkylphenols (Jobling et al. 1995; White et al. 1994), endosulfan, dieldrin, and toxaphene (McLachlan 1993; Soto et al. 1994) have all been shown to provide estrogenic responses in fish and other organisms through binding to estrogen receptors.

Environmental compounds in combination may act together with the estrogen receptors to provide responses far greater than that of single compounds.

Most of these pesticides bind only weakly to the estrogen receptor. For example, the binding of the insecticides endosulfan, dieldrin, toxaphene, and chlordane to the estrogen receptor was very weak: less than 1/10,000 that of the natural binding compound, estradiol (McLachlan,1993; Soto et al. 1994). For this reason, these compounds were thought for a long time to produce few environmental effects. However, environmental compounds in combination may act together

Figure 4. Combinations of chemicals have a greater estrogenic effect on organisms than single chemicals.



Notes: Lines above the bars represent standard errors. * indicates that the increase in number of cells is significantly greater than unexposed cells. Proliferation of breast cancer cells is a standard test for estrogenic effects.

Source: Soto, A. M., K. L. Chung, and C. Sonnenschein. 1994. The pesticides endosulfan, toxaphene, and dieldrin have estrogenic effects on human estrogen-sensitive cells. *Environ. Health Perspect.* 102:380-383.

with the estrogen receptors to provide responses far greater than that of single compounds (see Figure 4).

Feminization of males may also be brought about by antiandrogenic compounds. Vinclozolin, a fungicide used widely for grapes, ornamental plants, snap peas, and turfgrass (Kelce et 1994), causes feminization of males. The fungicide binds to androgen receptors and enters the nucleus, but blocks normal androgenic responses. The vinclozolin-androgen receptor complex cannot recognize the appropriate DNA response elements to stimulate production of essential male proteins (Wong et al. 1995). DDE, the persistent break-down product of DDT, also binds to androgen receptors with similar results (Kelce et al. 1995). Antiandrogenic responses in fish are not well understood at present. Part of this difficulty is the problem of demonstrating androgen binding activity in fish tissues (Monosson et al 1997). The relationship of the active androgen in fish, 11-ketotestosterone, to receptors for testosterone in the nucleus is not clear (Pasmanik and Callard 1985; Callard and Callard 1987), although a receptor for 11-ketotestosterone in the ovaries of coho salmon has been reported (Fitzpatrick et al. 1994).

The peculiar lability of sex differentiation in fishes suggests that endocrine-disrupting chemicals in the environment may profoundly alter the sex ratios and breeding capabilities of our native fish.

CORVALLIS ENVIRONMENTAL CENTER

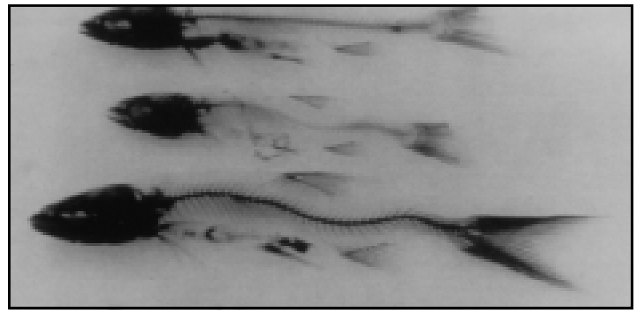
The third category of hormonal disruption which may result in feminization by environmental pollutants arises from the interference with regulatory functions in the brain and pituitary. This interference can affect neurotransmitter functions, sensory systems, control of serum hormone levels and therefore hormonal feedback. We know little about the role of pesticides in this area, although there is ample information that shows that brain function is a necessary part of providing the correct hormones for sexual differentiation. Serotonin has been shown to cause release of gonadotropins from the pituitary in croakers (Khan and Thomas 1992) and goldfish (Somoza et al. 1988) so compounds that interfere with serotonin levels may affect the release of gonadotropins. Arochlor 1254 has been shown to cause decreases in serotonin and dopamine levels in the brains of fish (Kahn and Thomas 1996), due either to increased degradation or decreased synthesis of the neurotransmitters. Many insecticides are cholinesterase inhibitors, that is, they interfere with the normal breakdown of acetylcholine, a neurotransmitter, by blocking the enzyme cholinesterase. This results in a longer lifespan for acetylcholine and a disruption of normal neural function.

The peculiar lability of sex differentiation in fishes (Chan and Yeung 1983) suggests that endocrine-disrupting chemicals in the environment may profoundly alter the sex ratios and breeding capabilities of our native fish. This field is still young and effects on fish population are largely unknown. With the development of more sophisticated methods for detection of pseudo-estrogens and pseudo-androgens in the environment, we should have a better understanding of the effects of chemicals that find their way into our watersheds.

Pesticides May Have Other Endocrine Effects

Although disruption of sex steroid function has received the most attention, it is not the only type of endocrine disruption that may result from exposure to environmental chemicals. Brucker-Davis (1998) listed over 90 synthetic chemicals that affected function of the thyroid gland. Thyroid hormones are thought to influence seasonal adaptations, reproduction, migration, and growth of fish. Hypothyroidism resulting in the formation of goiters was shown in coho and chinook salmon rearing in the Great Lakes and was related to environmental contaminants in the lakes (Moccia et al. 1977; Moccia et al. 1981). Thyroid impairment has been shown in the catfish *Clarias batrachus* after exposure to carbaryl (Sinha et al. 1991) and malathion (Sinha et al. 1992). Both malathion and carbaryl suppressed levels of thyroxine in the kidney, but increased thyroxine in the pharyngeal thyroid. Peripheral conversion of thyroxine to the active hormone, triiodothyronine, was suppressed with both compounds.

Skeletal deformities have long been recognized as an effect of environmental pollution and have been suggested as an excellent indicator of the extent of pollution (Bengtsson 1979). A comparison of reports of skeletal damage from old and new sources has given rise to the suspicion that water pollution may be having a larger impact on fish populations that we expect.



OREGON DEPT. OF ENVIRONMENTAL QUALITY

Squawfish from the Newberg pool in the Willamette River have a high incidence of skeletal deformities.

Bone formation and structure relies upon the mobilization and handling of calcium ions, which are controlled by a variety of hormones, including growth hormone, calcitonin, parathyroid hormone, and stanniocalcin (Davis 1997). These are in turn controlled by the brain-pituitary axis described earlier and subject to similar feedback mechanisms to regulate their synthesis and concentration. Unfortunately, calcium regulation mechanisms have not received the study that reproductive hormones have and consequently we know little about specific mechanisms underlying skeletal defects. Evidence that links environmental pesticides to skeletal abnormalities, however, is widespread.

In the early 1970s, fish biologists studying members of the drum family in James River, Virginia, found that a large number of specimens had shortened vertebral columns compared to similar species in the nearby York River. With the help of toxicologists, they found that a manufacturing facility for chlordecone, a widely-used insecticide, had been releasing large amounts of the compound into the James River. The stunted fish contained high chlordecone levels (Davis 1997). Exposure to chlordecone interfered with calcium deposition into bone and produced scoliosis and compression of the vertebrae (Couch et al. 1977). After 1976, when manufacture of chlordecone had stopped, the frequency of vertebral abnormalities decreased.

Wells and Cowan (1982) reported dysplasia in Atlantic salmon parr after a simulated spill of trifluralin, a pre-emergent herbicide, in the Eden River of Scotland. The fish were exposed to the herbicide for less than 1 day. Parr developed scoliotic vertebral abnormalities that caused them to appear stumpy. Laboratory tests showed that exposure to trifluralin resulted in hyperosteosis, a proliferation of the bone tissue (Couch et al. 1979). This was accompanied by fusion of vertebrae, persistence of osteoblasts, abnormal vertebral processes, and, strangely, the presence of notochord tissue in the vertebral canal. Clearly, the mechanisms of bone formation had been severely disrupted.

Trifluralin, a pesticide found in the Willamette River, has been shown to cause bone abnormalities in Atlantic salmon.

A number of other environmental pollutants, including chlorinated hydrocarbons (toxaphene, DDT), organophosphate pesticides (malathion, parathion, demeton), crude oil, and heavy metals (Bengtsson 1979), have been implicated in bone abnormalities in fishes. In most of these studies, researchers feel that the damage is done in the egg or early fry stages, possibly by effects on neuromuscular interactions during early development.

Recent studies on squawfish in the Willamette River have shown elevated numbers of abnormalities in bone formation, especially in the Newberg Pool (RM 26 to RM 60) (Altman et al. 1997). In the Newberg Pool, skeletal deformities ranged from 22.6 to 52.0 percent. No specific cause for the deformities has been identified, but a variety of factors including high temperatures, low dissolved oxygen, pesticides, PCBs, and effluents from pulp mills, paper mills, and ore smelters have been suggested. Analyses by the U. S. Geological Survey showed the presence of heavy metals (Altman et al. 1997), organochlorine pesticides, organophosphate pesticides, and even trifluralin in the Willamette River and its tributaries (Anderson et al. 1997). Although no skeletal abnormalities were found in salmonids in the watershed, the widespread occurrence of pesticides in Willamette River and the presence of skeletal abnormalities in squawfish should stimulate research on the potential endocrine-disrupting effects of the most common contaminants, such as atrazine, simazine, metolachor, and diuron. At the least, such abnormalities in the fish populations should provide a warning for the presence of endocrine disruptors in our watersheds.

Indirect Effects of Pesticides on Salmon

LYNN KETCHUM, OREGON STATE UNIVERSITY

Pesticides can indirectly affect fish by interfering with their food supply or altering the aquatic habitat, even when the concentrations are too low to affect the fish directly. Such indirect effects may greatly reduce the abundance of food organisms which in turn reduces the growth and probable survival of the fish.

Excellent examples of indirect effects of pesticides on salmonids were documented in the attempts to control spruce budworm in the forests of northeastern U. S. and Canada (Muirhead-Thomson 1987). In the mid 1970s, permethrin, a synthetic pyrethroid, was introduced to kill the spruce budworm. The insecticide was lethal to insects and moderately toxic to fish. Experimental sprayings of permethrin at 13 different streams between 1976 and 1980 did not result in moralities in native fish (Kingsbury and Kreutzweiser 1987). Blocking seines set at the bottom of the sprayed areas were never found to contain dead fish. Minnows and juvenile Atlantic salmon held in live cages in the sprayed area showed no mortality attributed to the spraying regimes.

Pesticides can indirectly affect fish by interfering with their food supply or altering the aquatic habitat, even when the concentrations are too low to affect the fish directly.

The same could not be said for the aquatic larvae in the streams, however (Kreutzweiser and Kingsbury 1987). Huge numbers of poisoned nymphs were found drifting downstream after spraying. Comparisons with pre-spray drift samples showed that the numbers of dead nymphs increased as much as 6000 times after spraying. This massive mortality was followed by a feeding frenzy among the salmonids. Atlantic salmon increased the number of nymphs in their stomachs by 2-4 fold. Brook trout fed even more voraciously, increasing their stomach contents by as much as 10-fold.

The poisoned insects seem to have no apparent direct effects on the salmonids, which fed as long as the drifting larvae were available. With time, however, the diets of the salmonids, especially the brook trout, changed considerably. With few of the normal food components of stonefly and mayfly nymphs and caddis larvae left in the streams, the trout changed to a diet of primarily *Dipteran* (fly) larvae and terrestrial insects.

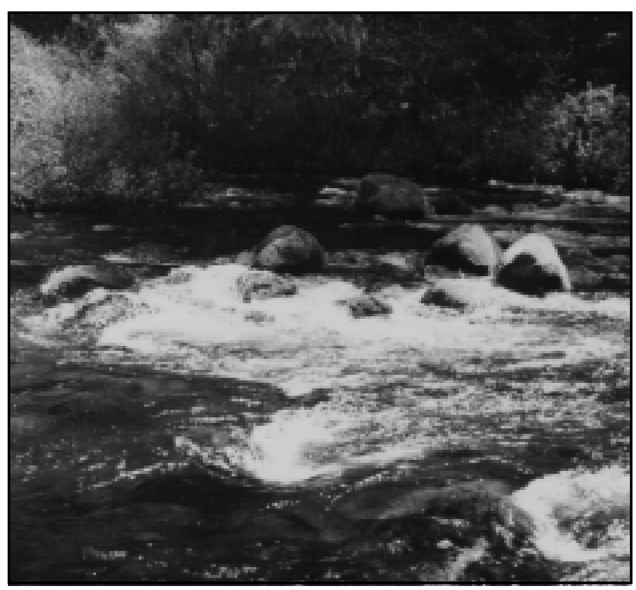
Densities of Atlantic salmon and brook trout were followed for a year after spraying. One month after spraying, the density of Atlantic salmon rearing in the area of highest application had decreased substantially. Brook trout populations did not show as great a decrease. It was felt that the reduced density of the Atlantic salmon was due to migration from the area in response to a reduced food supply.

Insect larvae and crustaceans are very sensitive to many types of pesticides, reducing the availability of these organisms to fish as prey (Macek et al. 1976; Brown 1978; Muirhead-Thomson 1987). Experimental spraying of ponds with carbaryl to simulate overspraying for spruce budworm control resulted in massive mortality of aquatic invertebrates (Muirhead-Thomson 1987). Some species of amphipods were not found for three years after the spray event (Gibbs et al. 1984). Walker (1964) found no fish mortality after application of 2-6 ppm of atrazine, a common herbicide, to ponds but insect larvae were killed immediately.

Pesticides may also indirectly affect prey species. Aquatic herbicides kill algae and diatoms on which aquatic insect larvae feed, thereby reducing the number of aquatic insects and decreasing growth and survival of fish feeding on the insects (Brown 1978).

In experimental treatments of replicate ponds with atrazine, deNoyelles et al. (1989) found that phytoplankton and aquatic plants were reduced by as much as 95% at the higher concentrations. Zooplankton did not seem to be directly affected. However, several months later, zooplankton populations also were reduced (deNoyelles et al. 1982). In the same study on atrazine, Dewey (1986) reported a decrease in the number of nonpredatory insects, while the predatory insect population was unaffected. She suggested that the decrease in nonpredatory insects was due to a reduction of algae and plants used as food and a reduction of water plants used as habitat for the insects. Further study of the fish populations indicated that the reproduction of bluegills was greatly reduced, probably from a lack of food. The stomach contents of the bluegills in the treated ponds were significantly lower than those in the control ponds (Kettle et al. 1987). Populations of young bluegills were further reduced by the lack of aquatic vegetation in which to hide. Without the aquatic vegetation, the young were preyed upon by the adult bluegills (Kettle et al. 1987).

At present, we have no information on whether deleterious effects of pesticides to algae (Stadnyk et al. 1971; Brown 1978; Bester 1995) or diatoms (Menzel et al. 1970; Brown 1978) may have trickle-down effects on salmonid populations, although it seems likely that they would be subject to population dynamic forces similar to those demon-



NEVA HASSANEIN

The integrity of aquatic ecosystems is essential for healthy salmon populations.

strated with bluegills. Interference with growth of salmonids, however, has deleterious effects on smolting (Ewing et al. 1980), immune function (Tatner 1996), predation (Parker 1971), seaward migration (Wedemeyer et al. 1980), and seawater adaptation (Folmar and Dickhoff 1980; Mahnken et al. 1982).

Increased predation on small fish may occur when water plants are killed by herbicides in the aquatic environment.

In addition to interference with their food supply, pesticides may also affect the habitat of the fish. Increased predation on small fish may occur when water plants are killed by herbicides in the aquatic environment (Lorz et al. 1979; Kettle et al. 1987). Adverse water temperatures may result from the removal of riparian vegetation by herbicides (Norris et al. 1991). Fish may avoid contaminated areas with a consequent reduction in availability and suitability of feeding and spawning areas (Beitinger 1990; Birge et al. 1993). Brown (1978) found that application of the herbicide paraquat caused death and decomposition of aquatic plants which in turn lowered the oxygen content of the water enough to kill trout.

These results and others of similar nature (Brown 1978; Muirhead-Thomson 1987) indicate that disruption of the food chains in aquatic ecosystems can have relatively long-lasting effects on fish populations. In complex ecosystems, indirect effects can be more important than direct effects (Lampert et al. 1989) and non-target organisms may be better indicators of ecosystem health than the organisms of interest. Management of pesticide applications for protection of aquatic habitats is obviously much more complex than promoting the use of LC_{50}s for setting application limits.

Conclusions

Salmonids require coordinated responses to environmental stimuli and internal physiological reactions for healthy development, territorial defense, smolting, seaward migration, and adaptation to seawater. Interruptions of this interplay may cause inappropriate physiological responses that have deleterious effects on survival of the salmon. As we have shown here, sublethal concentrations of pesticides can cause such interruptions and therefore result in harmful changes in physiology and behavior. The wide array of effects that have been demonstrated is cause for concern (Table 3).

The magnitude of the problem from pesticides is difficult to assess because of the lack of monitoring programs for the use and distribution of pesticides. Yet, where water quality monitoring has been done, pesticides are usually detected. A recent report commissioned by the Oregon Legislature (Botkin et al. 1995:104) concluded that fertilizers and pesticides were detri-

Table 3. Sublethal effects of selected pesticides found in the Willamette River Basin Study by USGS.[1]

	Herbicides		Insecticides		
	Atrazine	Simazine	Malathion	Chlorpyrifos	Carbaryl
Negative Effects on:					
Food Supply	+[2]		+[3]	+[4]	+[5]
Growth	+[2]		+[6]	+[7]	+[8]
Reproductive success	+[2]			+[9]	+[10]
Bone Abnormality			+[11]	+[12]	+[13]
Endocrine Disruptor			+[14]		+[13,14]
Immune System			+[15]		
Behavior		+[16]	+[17,18]	+[12,17]	+[12]
Schooling					+[19]
Predator Avoidance	+[20]		+[18]		

Plus indicates a detectable response from sublethal concentrations of pesticide. Blanks indicate that studies were not found analyzing these effects. Superscripts are references.

1. Anderson et al. 1997.
2. Macek et al. 1976.
3. Naqvi and Hawkins 1989.
4. Washino et al. 1972.
5. Burdick et al. 1960.
6. Hermanutz 1978.
7. Brazner and Kline. 1990.
8. Arunachalam and Palanichamy 1982.
9. Jarvinen et al. 1983.
10. Carlson 1971.
11. Weis, P. and J. S. Weis. 1976.
12. Holcombe et al. 1982.
13. Weis, J. S. and P. Weis. 1976.
14. Bruckner-Davis 1998.
15. Plumb and Areechon. 1990.
16. Dodson and Mayfield. 1979b.
17. Hansen 1969.
18. Hansen 1972.
19. Weis, P. and J. S. Weis. 1974.
20. Lorz et al. 1979.

mental to salmon: "With such large applications of pesticides and fertilizers, some portion is sure to make its way into nearby streams, especially since most agricultural land lies in close proximity to streams and rivers. *Since little monitoring is done to detect the presence of pesticides and fertilizers in streams, the levels of exposure by fish to chemicals are largely unknown* (authors' emphasis)."

Certain facts about pesticides in the environment are known and reviewed briefly here:

• Pesticides are typically found in great variety in salmonid habitats.

• Pesticides break down into products that may be less toxic, of equal toxicity, or of greater toxicity than the original compounds.

• Fish and other aquatic organisms must continue to cope daily with a variety of pesticides or breakdown products which were used years ago but remain in aquatic environments.

• Pesticides move in streams and rivers throughout the watersheds and may pose problems far from the site of application.

• In a process known as bioaccumulation, pesticides absorbed into plant and animal tissues may become concentrated and reach levels many times higher than the concentrations in the surrounding water.

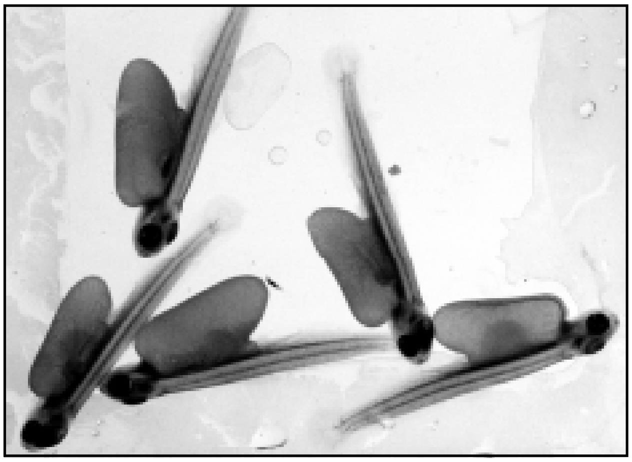
The levels of pesticides encountered in streams and rivers have occasionally reached lethal concentrations for salmonids, as evidenced by the major fish kills that have occurred in the Rogue River Basin and elsewhere. Loss of each individual in a sensitive population thwarts our efforts to recover wild salmonids.

In contrast to these dramatic fish kills, sublethal concentrations of pesticides are more subtle and their effects are largely unseen. From laboratory experimentation, researchers have found that sublethal concentrations of pesticides can affect many aspects of salmonid biology, including swimming performance, predator avoidance, temperature selection, schooling behavior, seaward migration, immunity to disease, reproduction, and food supply. Specifically, the literature reviewed here shows that:

• A variety of pesticides impair swimming performance which can reduce such critical behaviors as the ability to feed, to avoid predators, to defend territories, and to maintain position in the river system.

• Chronic exposure to certain pesticides can increase stress in juvenile salmonids and thereby render them more susceptible to predation.

OREGON DEPT. OF FISH AND WILDLIFE


OREGON DEPT. OF FISH AND WILDLIFE

Pesticides can affect salmon biology at many stages in their development.

• Several pesticides have been shown to cause fish to seek abnormal water temperatures, thus subjecting them to increased dangers of disease and predation.

• Many pesticides interrupt normal schooling behavior of salmon, a critical tactic for avoiding predation during migration.

• Several herbicides have been shown to inhibit normal seawater migration patterns; however, there is a dearth of research looking at this effect for common insecticides.

• Several studies suggest that certain pesticides can impair salmonids' ability to transition from fresh water to sea water; however there is a need for further research in this area, placing particular emphasis on the critical period of transition that takes place in the estuary.

• As with other pollutants, pesticides may interrupt the spawning migrations of adult salmon by interfering with the timing of migration and by draining energy reserves.

• Pesticides can suppress the normal functioning of the immune system, resulting in a higher incidence of disease.

• Certain pesticides can act as hormone mimics or blockers, causing abnormal sexual development, feminization of males, abnormal sex ratios, and abnormal mating behavior.

• Pesticides can interfere with other hormonal processes, such as normal thyroid functioning and bone development.

• Fishes and other organisms are especially vulnerable to endocrine-disrupting effects during the embryonic and early development stages of life.

• Pesticides can indirectly affect salmonids by altering the aquatic habitat and food supply for salmonids.

The Ecoepidemiological Approach

An extensive literature on the effects of synthetic chemicals on survival, physiology and reproduction of fishes is available (Murty 1986). The literature is disparate, however, and largely fails to attribute causality to events occurring in particular aquatic ecosystems. A new approach to appreciate the dangers pesticides pose for salmonids may be necessary. One method that has proved useful to establish the relationship between chlorinated hydrocarbon concentrations and the decrease in lake trout populations in the Great Lakes is the application of principles borrowed from epidemiology (Mac and Edsall 1991).

Epidemiology is the medical science used to determine the causes of disease in a particular population when many confounding factors are present simultaneously. The subdiscipline used for examining environmental health hazards to wildlife populations is called ecoepidemiology (Morgenstern and Thomas 1993). In epidemiology, researchers form a central hypothesis about the cause of an outbreak of a disease. The validity of the central hypothesis is then tested by five criteria, as illustrated by Mac and Edsall's (1991) study of the Great Lakes' trout.

The hypothesis in Mac and Edsall's study was that large concentrations of chlorinated hydrocarbons were responsible for the reduced reproductive success in populations of lake trout. The first criterion is that the problem must be temporally related to the appearance of the causative agent. In the Great Lakes, during the period 1975 to 1988, survival of trout eggs improved significantly as concentrations of PCBs, DDT, and oxychlordane declined. The second criterion is that there must be a strong correlation between exposure to the hypothetical agent and the disease. In a study of egg and fry survival at different locations, the extent of egg and fry loss was directly proportional to the concentration of PCBs in the water and in female fish at each location. Thirdly, the causative agent must be shown to induce the disease. Tests done in various laboratories established that PCBs could decrease egg and fry survival. Fourth, the observed cause-effect relationship must be shown in repeated studies. Comparison of different species at the same time, the same species at different times, and the same species at different locations established that reproductive impairment correlated with PCB exposure. Finally, there must be a biological plausibility of the relationship between the causative agent and the symtoms. Laboratory studies of tissue histology, exposure to other chlorinated hydrocarbons, and studies of enzyme induction were all consistent with the hypothesis that PCBs and chlorinated hydrocarbons were impairing egg and fry survival.

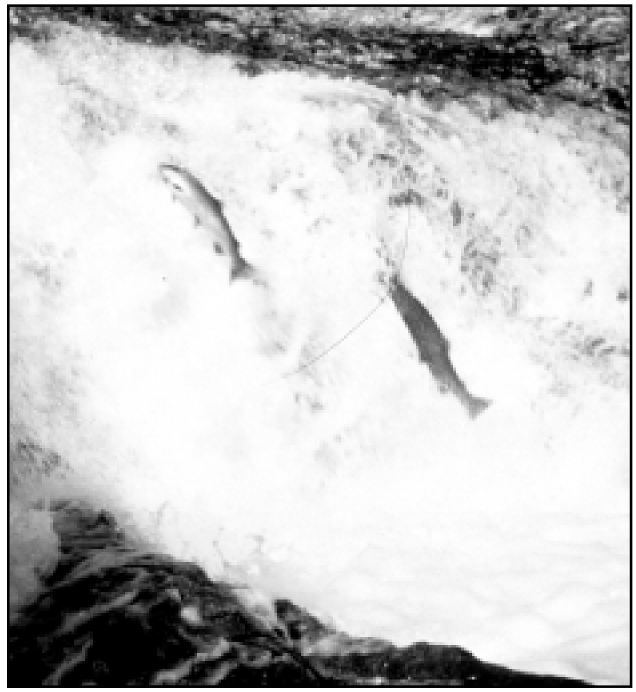
The authors concluded that their hypothesis was strongly supported, although they point out that "much of this evidence is circumstantial with little definitive proof of causality, which is why the epidemiological approach is used" (Mac and Edsall 1991). In other words, the strength of the ecoepidemiological approach is that it involves trying to understand relationships between environmental contaminants and population survival in real-world settings where a host of factors come into play. Restrictions on the use and dumping of organochlorine compounds into the Great Lakes resulted in partial restoration of the lake trout populations.

Recommendations

From the evidence available at present, it can be concluded that there is a plausible basis for considering pesticides as one causative factor in the decrease of salmon populations. Based on this review, we offer several policy recommendations and identify areas for further research:

1. Address the impacts of pesticides on salmon when developing and implementing recovery plans for threatened and endangered species. As federal, state, provinicial, and local agencies work to recover salmon in the Pacific Northwest, all factors contributing to the decline should be addressed. To date, pesticides have been overlooked as a factor deserving attention in our recovery efforts. Although there is a need for more information on the effects of pesticide use in salmon habitats, our salmon runs may be extinct by the time lengthy and laborious studies using strict scientific methods are complete. Thus, we must act now using available information to formulate management strategies that will minimize the potential danger from sublethal concentrations of pesticides.

2. Conduct ecoepidemiological studies in critical salmonid habitat. Most of the effects of pesticides referred to in this report have been determined in experimental laboratories. In the field, however, environmental conditions are not controlled, and many factors interact to confuse the determination of direct relationships. An ecoepidemiological approach (see inset opposite page) would be particularly valuable because it is designed to attribute causality to events occurring in real-world situations. This approach requires data which are not currently being collected. In particular, ecoepidemiological studies at the watershed level require specific pesticide use information in order to establish correlations between salmon decline and pesticide use.



INSTITUTE FOR FISHERIES RESOURCES

We must act now to protect salmon from pesticides.

3. Create comprehensive pesticide tracking systems in the Pacific Northwest. To better understand the relationship between pesticides and salmon decline, we must have accurate, site-specific data on the patterns of pesticide use in the watersheds of the Northwest. State and provincial governments need to collect data on which pesticides are used where, when, and in what amounts. Such data can then be combined with watershed-specific information on a host of physiological and behavioral indicators of salmon health. (Currently, California is the only state with salmon habitat where such information is collected.) Pesticide use information will also enable efficient instream monitoring for pesticide contamination.

4. Establish instream monitoring programs in critical salmon habitats. A systematic monitoring program for pesticides and their breakdown products needs to be undertaken. Clearly, not all pesticides can be tested for in all locations, but current testing is woefully inadequate for understanding the role of pesticides in salmon decline. In conjunction with pesticide use information, these analyses can be targeted to the compounds of most concern. Such targeting can greatly improve the cost-effectiveness of monitoring.

5. **Err on the side of caution when setting water quality standards for pesticides.** As discussed in this report, there are few established criteria for the protection of aquatic life from pesticides. Moreover, evidence reviewed here shows that sublethal effects on salmonids have not been fully appreciated, that juvenile salmonids succumb more easily to toxins in the water, that laboratory studies do not reflect the natural life cycle of the fish, and that little is known about how pesticides affect aquatic ecosystems. These factors must be considered when setting standards, and a precautionary approach must be adopted. Much can be gained by emphasizing how to eliminate the introduction of these toxic substances into the watersheds that comprise critical salmonid habitat.

6. Prevent pesticide contamination of salmonid habitat by reducing pesticide use. Once contaminated, water is difficult if not impossible to clean up. Therefore, pest management approaches that do not depend on pesticide use in agricultural and non-agricultural settings should be encouraged and further developed. There is ample evidence that ecologically sound and economically viable methods can be successfully implemented. The adoption of such alternatives can be encouraged through technical assistance, financial incentives and disincentives, demonstration programs, and information exchange opportunities.

7. Adopt state and provincial programs in the Pacific Northwest to phase out pesticides that persist and bioaccumulate in the environment. Numerous pesticides, including some that are no longer used and many that are currently used, are known to persist in the environment and to bioaccumulate in aquatic systems. Washington State's Department of Ecology is now considering a plan to end the release of such toxins, including certain pesticides, into the environment. To ensure salmon recovery, all state and provincial governments in the Pacific Northwest should adopt similar programs.

References

Ahei, M., J. McEvoy, and W. Giger 1993. Bioaccumulation of the lipophilic metabolites of nonionic surfactants in freshwater organisms. *Environ. Poll.* 79:243-248.

Altman, B., C. M. Henson, and I. R. Waite. 1997. Summary of information on aquatic biota and their habitats in the Willamette Basin, Oregon, through 1995. U. S. Geological Survey Water Resources Investigations Report 97-4023, Portland, OR. pp. 174.

Anderson, C. W., T. M. Wood, and J. L. Morace. 1997. Distribution of dissolved pesticides and other water quality constituents in small streams, and their relation to land use, in the Willamette River Basin, Oregon, 1996. Water Resources Investigations Report 97-4268, U. S. Geological Survey, Portland, OR.

Anderson, D. P., O. W. Dixon, and F. W. van Ginkel. 1984. Suppression of bath immunization in rainbow trout by contaminant bath pretreatments. pp. 289-293. IN: M. Kende, J. Gainer, and M. Chirigos, eds. *Chemical Regulation of Immunity of Veterinary Medicine.* Alan R. Liss, New York.

Anderson, D. P. 1996. Environmental factors in fish health: Immunological aspects. pp. 289-310. IN: Iwama, G. and T. Nakanishi, eds. *The Fish Immune System. Organism, Pathogen, and Environment.* Academic Press, New York.

Arunachalam, S. and S. Palanichamy. 1982. Sublethal effects of carbaryl on surfacing behavior and food utilization in the air-breathing fish, *Macropodus cupanus*. *Physiol. Behavior* 29:23-27.

Barbash, J. E. and E. A. Resek. 1996. *Pesticides in Ground Water. Distribution, Trends, and Governing Factors.* Ann Arbor Press, Inc., Chelsea, MI.

Beitenger, T. L. 1990. Behavioral reactions for the assessment of stress in fishes. *J. Great Lakes Res.* 16:495-528.

Bengtsson, B. E. 1979. Biological variables, especially skeletal deformities in fish, for monitoring marine pollution. *Phil. Trans. Roy. Soc. London* B 286:457-464.

Bennett, R. O. and R. E. Wolke. 1987. The effect of sublethal endrin exposrue on rainbow trout (*Salmo gairdneri* Richardson). I. Evaluation of serum cortisol concentrations and immune responsiveness. *J. Fish Biol.* 31:375-385.

Bern, H. A. 1990. The new endocrinology: its scope and impact. *Amer. Zool.* 30:877-885.

Bester, K., H. Huhnerfuss, U. Brockmann, and H. J. Rick. 1995. Biological effects of triazine herbicide contamination on marine phytoplankton. *Arch. Environ. Contam. Toxicol.* 29:277-283.

Birge, W. J., R. D. Hoyt, J. A. Black, M. D. Kercher, and W. A. Robison. 1993. Effects of chemical stresses on behavior of larval and juvenile fishes and amphibians. pp. 55-65. IN: Fuiman, L. A., ed. *Water Quality and the Early Life Stages of Fishes.* American Fisheries Society Symposium 14. American Fisheries Society, Bethesda, MD.

Birtwell, I. K. and G. M. Kruzinski. 1989. In situ and laboratory studies on the behavior and survival of Pacific salmon (genus *Oncorhynchus*). *Hydrobiologia* 188/189:543-560.

Bolander, F. 1994. *Molecular Endocrinology*, 2nd Edition. Academic Press, New York.

Bortleson, G. C. and D. A. Davis. 1997. Pesticides in selected small streams in the Puget Sound Basin, 1987-1995. U. S. Geological Survey Fact Sheet 067-97, U. S. Geological Survey, Tacoma, WA. 4 pp.

Bortone, S. A. and W. P. Davis. 1994. Fish intersexuality as indicator of environmental stress. *Bioscience* 44:165-172.

Botkin, D., K. Cummins, T. Dunne, H. Regier, M. Sobel, L. Talbot, and L. Simpson. 1995. Status and future of salmon of western Oregon and northern California. Findings and Options. Rept. #8. The Center for the Study of the Environment, Santa Barbara, CA.

Brazner, J. C. and E. R. Kline. 1990. Effects of chlorpyrifos on the diet and growth of larval fathead minnows, *Pimephales promelas*, in littoral enclosures. *Can. J. Fish. Aquat. Sci.* 47:1157-1165.

Brown, A. W. A. 1978. *Ecology of Pesticides.* John Wiley and Sons, New York

Brucker-Davis, F. 1998. Effects of environmental synthetic chemicals on thyroid function. *Thyroid* 8:827-856.

Burdick, G. E., H. J. Dean, and E. J. Harris. 1960. Effect of Sevin upon the aquatic environment. *N. Y. Fish and Game J.* 7:14-25.

Callard, I. P. and G. V. Callard. 1987. Sex steroid receptors and non-receptor binding proteins. pp. 355-384. IN: D. O. Norris and R. E. Jones, eds. *Hormones and reproduction in fishes, amphibians, and reptiles.* Plenum Press, New York.

Carlson, A. R. 1971. Effects of long-term exposure to carbaryl (Sevin) on the minnow (*Pimephales promelas*). *J. Fish. Res. Bd. Can.* 29:583-587.

Chan, S. T. H. and W. S. B. Yeung. 1983. Sex control and sex reversal in fish under natural conditions. pp. 171-222. IN: W. S. Hoar, D. J. Randall, and E. M. Donaldson, eds. *Fish Physiology.* Volume IX. Reproduction. Part B. Behavior and Fertility Control. Academic Press, New York.

Chapman, R. A., C. M. Tu, C. R. Harris, and C. Cole. 1981. Persistence of five pyrethroid insecticides in sterile and natural, mineral, and organic soils. *Bull. Environ. Contam. Toxicol.* 26:513-516.

Chliamovich, Y. P. and C. Kuhn. 1977. Behavioral, haematological and histological studies on acute toxicity of bis(tri-n-butyltin)oxide on *Salmo gairdneri* Richardson and *Tilapia rendalli* Boulenger. *J. Fish Biol.* 10:575-582.

Clark, G. M., T. R. Maret, M. G. Rupert, M. A. Maupin, W. H. Low, and D. S. Ott. 1998. Water quality in the upper Snake River Basin, Idaho and Wyoming, 1992-95. U. S. Geological Survey Circular 1160, U. S. Geological Survey, Boise, ID.

Cleland, G. B., P. J. McElroy, and R. A. Sonstegard. 1989. Immunomodulation in C56B1/6 mice following consumption of halogenated aromatic hydrocarbon-containing coho salmon (*Oncorhynchus kisutch*) from Lake Ontario, Canada. *J. Toxicol. Environ. Health* 27:477-486.

Colborn, T. and C. Clement, eds. 1992. *Chemically-Induced Alterations in Sexual and Functional Development: The Human/Wildlife Connection.* Princeton Scientific Publishing Co., Princeton, NJ.

Colquhoun, J. R., H. A. Simonin, H. J. Dean, and J. Symula. 1984. Swimming performance of juvenile brown trout exposed to acutely sublethal concentrations of naled. *New York Fish and Game J.* 31:133-137.

Cook, P. M., E. W. Zabel, and R. E. Peterson. (1997) The TCDD toxicity equivalence approach for characterizing risks for early life stage mortality in trout. pp. 9-27. IN: Rolland, R.M., Gilbertson, M., and R.E. Peterson, eds. *Chemically induced alterations in functional development and reproduction of fishes.* Setac Press, Pensacola, FL

Couch, J., J. Winstead, and L. Goodman. 1977. Kepone-induced scoliosis and its histological consequences in fish. *Science* 197:585-586.

Couch, J. A., J. T. Winstead, D. J. Hansen, and L. R. Goodman. 1979. Vertebral dysplasia in young fish exposed to the herbicide trifluralin. *J. Fish Dis.* 2:35-42.

Cushing, D. H. and F. R. Harden-Jones. 1968. Why do fish school? *Nature* (London) 218:918-920.

Davis, W. P. 1997. Evidence for developmental and skeletal responses as potential signals of endocrine-disrupting compounds in fishes. pp. 61-72. IN: Rolland, R.M., Gilbertson, M., and R.E. Peterson, eds. *Chemically induced alterations in functional development and reproduction of fishes.* SETAC Press, Pensacola, FL

Day, K. E. 1991. Pesticide transformation products in surface waters. Effects on aquatic biota. pp. 217-241. IN: Somasundarum, L. and J. R. Coats, eds. *Pesticide Transformation Products. Fate and Significance in the Environment.* American Chemical Society, Washington, D. C.

deNoyelles, F., W. D. Kettle, C. H. Fromm, M. F. Moffett, and S. L. Dewey. 1989. Use of experimental ponds to assess the effects of a pesticide on the aquatic environment. pp. 41-56. IN: J.R. Voshell, Jr., ed. Using Mesocosms to Assess the Aquatic Ecological Risk of Pesticides: Theory and Practice. Entomological Society of America, Misc. Public. 75.

deNoyelles, F., W. D. Kettle, and D. E. Sinn. 1982. The responses of plankton communities in experimental ponds to atrazine, the most heavily used pesticide in the United States. *Ecology* 63:1285-1293.

Dewey, S. L. 1986. Effects of the herbicide atrazine on aquatic insect community structure and emergence. *Ecology* 67:148-162.

Dodson, J. J. and C. I. Mayfield. 1979a. The dynamics and behavioral toxicology of Aqua-Kleen (2,4-D butoxyethanol ester) as revealed by the modification of rheotropism in rainbow trout. *Trans. Am. Fish. Soc.* 108:632-640.

Dodson, J. J. and C. I. Mayfield. 1979b. Modification of the rheotropic response of rainbow trout (Salmo gairdneri) by sublethal doses of the aquatic herbicides diquat and simazine. *Environ. Pollut.* 18:147-161.

Donohoe, R. M. and L. R. Curtis. 1996. Estrogenic activity of chlordecone, o,p'-DDT, and o,p'-DDE in juvenile rainbow trout: induction of vitellogenesis and interaction with hepatic estrogen binding sites. *Aquat. Toxicol.* 36:31-52.

Drummond, R. A., C. L. Russum, D. L. Geiger, and D. L. De Foe. 1986. Behavioral and morphological changes in fathead minnow (*Pimephales promelas*) as diagnostic endpoints for screening chemicals according to mode of action. pp. 415-435. IN: Poston, T. M., and R. Purdy, eds. *Aquatic Toxicology and Environmental Fate*. Vol. 9. ASTM STP 921. American Society for Testing and Materials, Philadelphia, PA.

Dubrovsky, N. M., C. R. Kratzer, L. R. Brown, J. M. Gronberg and K. R. Burow. 1998. Water quality in the San Joaquin-Tulare Basins, California, 1992-95. U. S. Geological Survey Circular 1159, U. S. Geological Survey, Sacramento, CA.

Elson, P. F., L. N. Lauzier, and V. Zitko. 1972. A preliminary study of salmon movements in a polluted estuary. pp. 325-330. IN: M. Ruivo, ed. *Marine Pollution and Sea Life, Fishing News* (Books) Ltd, London.

Evenson, M. 1998. Personal communication, District biologist, Central Point, Oregon Department of Fish and Wildlife. October 22.

Ewing, R. D., H. J. Pribble, S. L. Johnson, C. A. Fustish, J. Diamond, and J. A. Lichatowich. 1980. Influence of size, growth rate, and photoperiod on cyclic changes in gill (Na+K)-ATPase activity in chinook salmon (*Oncorhynchus tshawytscha*). *Can. J. Fish. Aquat. Sci.* 37:600-605.

Fairchild, W. 1999. Personal Communication, Department of Fisheries and Oceans, Moncton, New Brunswick. January 26.

Fitzpatrick, M. S., W. L. Gale, and C. B. Schreck. 1994. Binding characteristics of an androgen receptor in the ovaries of coho salmon, *Oncorhynchus kisutch*. *Gen. Comp. Endocrinol.* 95:399-408.

Folmar, L. C., N. D. Denslow, V. Rao, M. Chow, D. A. Crain, J. Enblom, J. Marcino, and L. J. Guillette, Jr. 1996. Vitellogenin induction and reduced serum testosterone concentrations in feral male carp (*Cyprinus carpio*) captured near a major metropolitan sewage treatment plant. *Environ. Health Perspect.* 104:1096-1101.

Folmar, L. C. and W. W. Dickhoff. 1980. The parr-smolt transformation (smoltification) and seawater adaptation in salmonids. A review of selected literature. *Aquaculture* 21:1-37.

Fox, G. A. 1992. Epidemiological and pathobiological evidence of contaminant-induced alterations in sexual development in free-living wildlife. pp. 147-158. IN: T. Colborn and C. Clement, eds. *Chemically-Induced Alterations in Sexual and Functional Development: The Wildlife/Human Connection. Princeton Scientific Publishing Co., Inc., Princeton, New Jersey.*

Freitag, D. et al. 1985. Environmental hazard profile of organic chemicals. *Chemosphere* 14:1589-1616.

Fry, D. M. and C. K. Toone. 1981. DDT-induced feminization of gull embryos. *Science* 213:922-924.

Gardner, D. R. 1973. The effect of some DDT and methoxychlor analogs on temperature selection and lethality in brook trout fingerlings. *Pesticide Biochem. Physiol.* 2:437-446.

Gibbs, K. E., T. M. Mingo, and D. L. Courtemanch. 1984. Persistence of carbaryl (Sevin-4-Oil) in woodland ponds and its effects on pond macroinvertebrates following forest spraying. *Can. Entomol.* 116:203-213.

Guillette, L. J., Jr., D. A. Crain, A. A. Rooney, and D. B. Pickford. 1995. Organization versus activation: the role of endocrine-disrupting contaminants (EDCs) during embryonic development in wildlife. *Environ. Health Perspect*. Supp. 7 103:157-164.

Hansen, D. J. 1969. Avoidance of pesticides by untrained sheepshead minnow. *Trans. Am. Fish. Soc*. 98:426-429.

Hansen, D. J. 1972. DDT and malathion: effect on salinity selection by mosquitofish. *Trans. Am. Fish. Soc.* 82:346-354.

Hatfield, C. T. and P. H. Anderson. 1972. Effects of two insecticides on the vulnerability of Atlantic salmon (*Salmo salar*) parr to brook trout (*Salvelinus fontinalis*) predation. *J. Fish. Res. Bd. Can.* 29:27-35.

Hattula, M. L., V. M. Wasenius, H. Reunanen, and A. V. Arstila. 1981. Acute toxicity of some chlorinated phenols, catechols, and cresols to trout. *Bull. Environ. Contam. Toxicol.* 26:295-301.

Hermanutz, R. O. 1978. Endrin and malathion toxicity to flagfish (*Jordanella floridae*). *Arch. Environ. Contam. Toxicol*. 7:159-165.

Hill, I. R. 1985. Effects on non-target organisms in terrestrial and aquatic environments. IN: J. P. Leahey, ed. *The Pyrethroid Insecticides.* Taylor and Francis, London.

Holcombe, G. W., G. L. Phipps and D. K. Tanner. 1982. The acute toxicity of kelthane, Dursban, disulfoton, pydrin, and permethrin to fathead minnows *Pimephales promelas* and rainbow trout *Salmo gairdneri*. *Environ. Poll.* (Ser. A) 29:167-178.

Howard, P. H. 1991. *Handbook of environmental fate and exposure data for organic chemicals.* III. Pesticides. Lewis Publishers, Inc., Chelsea, MI

Howell, W. M., D. A. Black, and S. A. Bortone. 1980. Abnormal expression of secondary sex characters in a population of mosquitofish, *Gambusia affinis holbrooki*: evidence for environmentally induced masculinization. *Copeia* 1980:676-681.

Hunter, G. A. and E. M. Donaldson. 1983. Hormonal sex control and its application to fish culture. pp. 223-303. IN: W. S. Hoar, D. J. Randall, and E. M. Donaldson, eds. *Fish Physiology.* Volume IX. Reproduction. Part B. Behavior and Fertility Control. Academic Press, New York.

Iwama, G. and T. Nakanishi, eds. 1996. *The Fish Immune System. Organism, Pathogen, and Environment.* Academic Press, New York.

Iwata, M., T. Hirano, and S. Hasegawa. 1982. Behavior and plasma sodium regulation of chum salmon fry during transition into seawater. *Aquaculture* 28:133-142.

Janz, D. M., A. P. Farrell, J. D. Morgan, and G. A. Vigers. 1991. Acute physiological stress responses of juvenile coho salmon (*Oncorhynchus kisutch*) to sublethal concentrations of Garlon 4, Garlon 3A, and Vision herbicides. *Environ. Toxicol. Chem.* 10:81-90.

Jarvinen, A. W., B. R. Nordling, and M. E. Henry. 1983. Chronic toxicity of Dursban (chlorpyrifos) to the fathead minnow (*Pimephales promelas*) and the resultant acetylcholinesterase inhibition. *Ecotoxicol. Environ. Safety* 7:423-434.

Jobling, S., T. Reynolds, R. White, M. G. Porter, and J. P. Sumpter. 1995. A variety of environmentally persistent chemicals, including some phthalate plasticizers are weakly estrogenic. *Environ. Health Perspect.* 103:582-588.

Jobling, S. and J. P. Sumpter. 1993. Detergent components in sewage effluent are weakly estrogenic to fish: an *in vitro* study using rainbow trout (*Oncorhynchus mykiss*) hepatocytes. *Aquat. Toxicol.* 27:361-372.

Kelce, W. R., E. Monosson, M. P. Gamsik, S. C. Laws, and L. E. Gray. 1994. Environmental hormone disruptors: evidence that vinclozolin developmental toxicity is mediated by antiandrogenic metabolites. *Toxicol. Appl. Pharmacol.* 126:276-285.

Kelce, W. R., C. R. Stone, S. C. Laws, L. E. Gray, J. A. Kemppainen, and E. M. Wilson. 1995. Persistent DDT metabolite p,p'-DDE is a potent androgen receptor antagonist. *Nature* 375:581-585.

Kettle, W. D., F. deNoyelles, Jr., B. D. Heacock, and A. M. Kadoum. 1987. Diet and reproductive success of bluegill recovered from experimental ponds treated with atrazine. *Bull. Environ. Contam. Toxicol.* 38:47-52.

Khan, I. A. and P. Thomas. 1992. Stimulatory effects of serotonin on maturational gonadotropin release in the Atlantic croaker, *Micropogonias undulatus*. *Gen. Comp. Endocrinol.* 88:388-396.

Khan, I. A. and P. Thomas. 1996. Disruption of neuroendocrine function in Atlantic croaker exposed to Arochlor 1254. *Mar. Environ. Res.* 42:145-149.

Kingsbury, P. D. and D. P. Kreutzweiser. 1987. Permethrin treatments in Canadian forests. Part 1: Impact on stream fish. *Pesticide Sci.* 19:35-48.

Koenig, C. C. 1977. The effects of DDT and Mirex alone and in combination on the reproduction of a salt marsh Cyprinodont fish, *Adinia xenica.* pp. 357-376. IN: F. J. Vernberg, A. Calabrese, F. P. Thurberg, and W. B. Vernberg, eds. *Physiological Responses of Marine Biota to Pollutants*. Academic Press, New York.

Kreutzweiser, D. P. and P. D. Kingsbury. 1987. Permethrin treatments in Canadian forests. Part 2: Impact on stream invertebrates. *Pesticide Sci.* 19:49-60.

Kruzynski, G. M. 1972. Some effects of dietary methoxychlor on brook trout (*Salvelinus fontinalis*). M. Sc. thesis, Sir George Williams (Concordia) University, Montreal, Quebec. Reported in Kruzinski and Birtwell (1994) *Can. J. Fish. Aquat. Sci.* 51:1780-1790

Kruzynski, G. M. and I. K. Birtwell. 1994. A predation bioassay to quantify the ecological significance of sublethal responses of juvenile chinook salmon (*Oncorhynchus tshawytscha*) to the antisapstain fungicide TCMTB. *Can. J. Fish. Aquat. Sci.* 51:1780-1790.

Lampert, W., W. Fleckner, E. Pott, U. Schober, and K. U. Storkel. 1989. Herbicide effects on planktonic systems of different complexity. *Hydrobiol.* 188/189:415-424.

Larson, S. J., P. D. Capel, and M. S. Majewski. 1997. *Pesticides in Surface Waters. Distribution, Trends, and Governing Factors.* Ann Arbor Press, Chelsea, MI.

Little, E. E., R. D. Archeski, B. A. Flerov, and V. I. Kozlovskaya. 1990. Behavioral indicators of sublethal toxicity in rainbow trout. *Arch. Environ. Contam. Toxicol.* 19:380-385.

Lorz, H. W., S. W. Glenn, R. H. Williams, C. M. Kunkel, L. A. Norris, and B. R. Loper. 1979. Effects of selected herbicides on smolting of coho salmon. U. S. Environmental Protection Agency, Environmental Research Laboratory, EPA 600/3-79-071. Corvallis, OR.

Mac, M. J. and C. C. Edsall. 1991. Environmental contaminants and the reproductive success of lake trout in the Great Lakes: An epidemiological approach. *J. Toxicol. Environ. Health* 33:375-394.

Macek, K. J., K. S. Buxton, S. Sauter, S. Gnilka, and J. W. Dean. 1976. Chronic toxicity of atrazine to selected aquatic invertebrates and fishes. USEPA Ecological Res. Serv. EPA-600/3-76-047, Duluth, MN. 50 pp.

Macek, K. J. and S. Korn. 1970. Significance of the food chain in DDT accumulation by fish. *J. Fish. Res. Bd. Can*. 27:1496-1498.

Mahnken, C., E. Prentice, W. Waknitz, G. Monan, C. Sims, and J. Williams. 1982. The application of recent smoltification research to public hatchery releases: An assessment of size/time requirements for Columbia River hatchery coho salmon (*Oncorhynchus kisutch*). *Aquaculture* 28:251-268.

Maule, A. G. and C. B. Schreck. 1990. Changes in numbers of leukocytes in immune organs of juvenile coho salmon (*Oncorhynchus kisutch*) after acute stress or cortisol treatment. *J. Aquat. Anim. Health* 2:298-

304.

Maule, A. G., C. B. Schreck, and S. L. Kaattari. 1987. Changes in the immune system of coho salmon (*Oncorhychus kisutch*) during the parr-smolt transformation and after implantation of cortisol. *Can. J. Fish. Aquat. Sci.* 44:161-166.

McBride, J. R., H. M. Dye, and E. M. Donaldson. 1981. Stress response of juvenile sockeye salmon (*Oncorhynchus nerka*) to the butoxyethanol ester of 2,4-dichlorophenoxyacetic acid. *Bull. Environ. Contam.* 27:877-884.

McLachlan, J. A. 1993. Functional toxicology: a new approach to detect biologically active xenobiotics. *Environ. Health Perspect.* 101:386-387.

McLeese, D. W., C. D. Metcalf, and V. Zitko. 1980. Lethality of permethrin, cypermethrin and fenvalerate to salmon, lobster and shrimp. *Bull. Environ. Contam. Toxicol.* 25:950-955.

Mehrle, P. M. et al. 1987. Toxicity and bioconcentration of 2,3,7,8-tetrachloro-dibenzo-p-dioxin and 2,3,7,8-tetrachloro-dibenzofuran in rainbow trout. *Environ. Toxicol. Chem.* 27:47-62.

Menzel, D. W., J. Anderson, and A. Randtke. 1970. Marine phytoplankton vary in their response to chlorinated hydrocarbons. *Science* 167:1724-1726.

Metcalf, R. L., G. K. Sangha, and I. P. Kapoor. 1971. Model ecosystem for the evaluation of pesticide biodegradability and ecological magnification. *Environ. Sci. Tech.* 5:709-716.

Miller, D. L. and D. M. Ogilvie. 1975. Temperature selection in brook trout (*Salvelinus fontinalis)* following exposure to DDT, PCB, or phenol. *Bull. Environ. Contam. Toxicol.* 14:545-551.

Mitchell, D. G., P. M. Chapman, and T. J. Long. 1987. Seawater challenge testing of coho salmon smolts following exposure to Roundup herbicide. *Environ. Toxicol. Chem.* 6:875-878.

Moccia, R. D., J. F. Leatherland, and R. A. Sonstegard. 1977. Increasing frequency of thyroid goiters in coho salmon (*Oncorhynchus kisutch*) in the Great Lakes. *Science* 198:425-426.

Moccia, R. D., J. F. Leatherland, and R. A. Sonstegard. 1981. Quantitative interlake comparison of thyroid pathology in Great Lakes coho (*Oncorhynchus kisutch*) and chinook (*Oncorhynchus tshawytscha*) salmon. *Cancer Res.* 41:2200-2210.

Moller, H. 1985. A critical review on the role of pollution as a cause of fish diseases. pp. 169-182. IN: A. E. Ellis, ed. *Fish and Shellfish Pathology.* Academic Press, New York.

Monosson, E., W. R. Kelce, M. Mac, and L. E. Gray. 1997. Environmental antiandrogens: potential effects on fish reproduction and development. pp. 53-60. IN: Rolland, R. M., M. Gilbertson, and R. E. Peterson, eds. *Chemically Induced Alterations in Functional Development and Reproduction of Fishes.* SETAC Press, Pensacola, FL

Morgan, M. J., and J. W. Kiceniuk. 1992. Response of rainbow trout to a two month exposure to Vision, a glyphosate herbicide. *Bull. Environ. Contam. Toxicol.* 48:722-780.

Morgenstern, H. and D. Thomas. 1993. Principles of study design in environmental epidemiology. *Environ. Health Perspect.* 101 (Suppl. 4):23-38.

Muirhead-Thomson, R. C. 1987. *Pesticide Impact on Stream Fauna with Special Reference to Macroinvertebrates.* Cambridge University Press, Cambridge, UK.

Murty, A. S. 1986. *Toxicity of Pesticides to Fish.* Volumes 1 and 2. CRC Press, Inc., Boca Raton, FL.

Naqvi, S. M. and R. H. Hawkins. 1989. Responses and LC50 values for selected microcrustaceans exposed to Spartan, malathion, Sonar, Weedtrine-D, and Oust pesticides. *Bull. Environ. Contam. Toxicol.* 43:386-393.

National Research Council. 1993. Soil and Water Quality: An Agenda for Agriculture. Washington, DC: National Academy Press.

Nikl, D. L. and A. P. Farrell. 1993. Reduced swimming performance and gill structural changes in juvenile salmonids exposed to 2-(thiocyanomethylthio) benzothiazole. *Aquat. Toxicol.* 27:245-264.

Norris, L. A., H. W. Lorz, and S. V. Gregory. 1991. Forest Chemicals. pp. 207-296. IN: W. R. Meehan, ed. Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. *Amer. Fish. Soc. Sp. Publ.* 19, Bethesda, MD.

Ogilvie, D. M. and J. M. Anderson. 1965. Effect of DDT on temperature selection by young Atlantic salmon, *Salmo salar*. *J. Fish. Res. Bd. Can.* 22:503-512.

Ogilvie, D. M. and D. L. Miller. 1976. Duration of a DDT-induced shift in the selected temperature of Atlantic salmon (*Salmo salar*). *Bull. Environ. Contam. Toxicol.* 16:96-99.

O'Halloran, K., J. T. Ahokas, and P. F. A. Wright. 1998. Response of fish immune cells to *in vitro* organotin exposures. *Aquat. Toxicol.* 40:141-156.

Olla, B. L., M. W. Davis, and C. B. Schreck. 1992. Comparison of predator avoidance capabilities with corticosteroid levels induced by stress in juvenile coho salmon. *Trans. Am. Fish. Soc.* 121:544-547.

Olla, B. L., M. W. Davis, and C. B. Schreck. 1995. Stress-induced impairment of predator evasion and non-predator mortality in Pacific salmon. *Aquacult. Res*. 26:393-398.

Parker, R. R. 1971. Size selective predation among juvenile salmonid fishes in a British Columbia inlet. *J. Fish. Res. Bd. Can.* 28:1503-1510.

Pasmanik, M and G. Callard. 1985. Identification and characterization of an androgen receptor in the brain of goldfish (*Carassius auratus*). Am. Zool. 25:115A.

Paul, E. A. and H. A. Simonin. 1996. Effects of naled, synergised and non-synergised resmethrin on the swimming performance of young trout. *Bull. Environ. Contam. Toxicol.* 57:495-502.

Pearcy, W. G. 1992. *Ocean Ecology of North Pacific Salmonids*. Washington Sea Grant Program. Seattle: University of Washington Press.

Peterson, R. H. 1973. Temperature selection of Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) as influenced by various chlorinated hydrocarbons. *J. Fish. Res. Bd. Can.* 30:1091-1096.

Pickering, A. D. 1993. Endocrine-induced pathology in stressed salmonid fish. *Fish. Res*. 17:35-50.

Pickering, A. D. and T. G. Pottinger. 1989. Stress responses and disease resistance in salmonid fish: Effects of chronic elevation of plasma cortisol. *Fish Physiol. Biochem.* 7:253-258.

Piferrer, F., S. Zanuy, M. Carrillo, I. I. Solar, R. H. Devlin, and E. M. Donaldson. 1994. Brief treatment with an aromatase inhibitor during sex differentiation causes chromosomally female salmon to develop as normal, functional males. *J. Exp. Zool.* 270:255-262.

Plumb, J. A. and N. Areechon. 1990. Effect of malathion on humoral immune response of channel catfish. *Develop. Comp. Immunol*. 14:355-358.

Racka, K. D. 1993. Environmental fate of chlorpyrifos. *Rev. Environ. Contam. Toxicol.* 13:1-151.

Repetto, R. and S. S. Baliga. 1996. *Pesticides and the immune system: The public health risks.* World Resources Institute, Washington, D. C.

Risebrough, R. W., D. B. Menzel, D. J. Marsten, Jr., and H. S. Olcott. 1967. DDT residues in Pacific sea birds: A persistent insecticide in marine food chains. *Nature* 216:589-591.

Rolland, R. M., M. Gilbertson, and R. E. Peterson, eds. 1997. *Chemically Induced Alterations in Functional Development and Reproduction of Fishes.* SETAC Press, Pensacola, FL.

Sinha, N., B. Lal, and T. P. Singh. 1991. Carbaryl-induced thyroid disfunction in the freshwater catfish, *Clarias batrachus*. *Ecotoxicol. Environ. Safety* 21:240-247.

Sinha, N., B. Lal, and T. P. Singh. 1992. Thyroid physiology impairment by malathion in the freshwater catfish, *Clarias batrachus*. *Ecotoxicol. Environ. Safety* 24:17-25.

Smith, L. S., R. D. Cardwell, A. J. Mearns, T. W. Newcomb, and K. W. Watters. 1972. Physiological changes experienced by Pacific salmon migrating through a polluted urban estuary. pp. 322-325. IN: M. Ruivo, ed. *Marine Pollution and Sea Life,* Fishing News (Books) Ltd, London.

Smith, T. M. and G. W. Stratton. 1986. Effects of synthetic pyrethroid insecticides on nontarget organisms. *Residue Reviews* 97:93-120.

Somoza, G. M., K. L. Yu, and R. E. Peter. 1988. Serotonin stimulates gonadotropin release in female and male goldfish, *Carassius auratus* L. *Gen. Comp. Endocrinol.* 72:374-382.

Soto, A. M., K. L. Chung, and C. Sonnenschein. 1994. The pesticides endosulfan, toxaphene, and dieldrin have estrogenic effects on human estrogen-sensitive cells. *Environ. Health Perspect.* 102:380-383.

Specker, J. L. and C. B. Schreck. 1982. Changes in plasma corticosteroids during smoltification of coho salmon, *Oncorhynchus kisutch*. *Gen. Comp. Endocrinol.* 46:53-59.

Spitzbergen, J. M., K. A. Schat, J. Kleeman, and R. E. Peterson. 1986. Interaction of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) with immune responses of rainbow trout. *Vet. Immunol. Immunopath.* 12:263-280.

Spitzbergen, J. M., K. A. Schat, J. Kleeman, and R. E. Peterson. 1988. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) or Arochlor 1254 on the resistance of rainbow trout, *Salmo gairdneri* Richardson, to infectious hematopoetic necrosis virus. *J. Fish Dis.* 11:73-83.

Stadnyk, L., R. S. Campbell, and B. T. Johnson. 1971. Pesticide effect on growth and ^{14}C assimilation in a freshwater algae. *Bull. Environ. Contam. Toxicol.* 6:1-8.

Symons, P. E. K. 1973. Behaviour of young Atlantic salmon (*Salmo salar*) exposed to or force-fed fenitrothion, an organophosphate insecticide. *J. Fish. Res. Bd. Can.* 30:651-655.

Tatner, M. F. 1996. Natural changes in the immune system of fish. IN: Iwama, G. and T. Nakanishi, eds. *The Fish Immune System. Organism, Pathogen, and Environment.* Academic Press, New York.

Thomas, P. and I. A. Khan. 1997. Mechanisms of chemical interference with reproductive endocrine function in sciaenid fishes. pp 29-51. IN: Rolland, R. M., M. Gilbertson, and R. E. Peterson, eds. *Chemically Induced Alterations in Functional Development and Reproduction of Fishes.* SETAC Press, Pensacola, FL.

Thorpe, J. E., M. G. McConway, M. S. Miles, and J. S. Muir. 1987. Diel and seasonal changes in resting plasma cortisol levels in juvenile Atlantic salmon, *Salmo salar* L. *Gen. Comp. Endocrinol*. 65:19-22.

Torstensson, N. T. L., L. N. Lundgren, and J. Stenstrom. 1989. Influence of climate and edaphic factors on persistence of glyphosate and 2,4-D in forest soils. *Ecotoxicol. Environ. Safety* 18:230-239.

Veith, G. D., D. L. Defoe, and B. V. Bergstedt. 1979. Measuring and estimating the bioconcentration factor of chemicals in fish. *J. Fish. Res. Bd. Can.* 36:1040-1049.

Wagner, R. J., J. C. Ebbert, and L. M. Roberts. 1996. Are agricultural pesticides in surface waters of the central Columbia plateau? U. S. Geological Survey Fact Sheet 241-95, U. S. Geological Survey, Tacoma, WA.

Walker, C. R. 1964. Simazine and other s-triazine compounds as aquatic herbicides in fish habitats. *Weeds* 12:134-139.

Washino, R. K., W. Ahmed, J. D. Linn, and K. G. Whitesell. 1972. Effects of low volume Dursban sprays upon aquatic non-target organisms. *Mosq. News* 32:531-537.

Wedemeyer, G. A., R. L. Saunders, and W. C. Clarke. 1980. Environmental factors affecting smoltification and early marine survival of anadromous salmonids. *Mar. Fish. Rev.* 42:1-15.

Weis, J. S. and P. Weis. 1976. Optical malformation induced by insecticides in embryos of the Atlantic silverside *Menidia menidia*. *Fish. Bull.* 74:208-214.

Weis, P. and J. S. Weis. 1974. Sevin and schooling behavior of *Menidia menidia* in the presence of the insecticide Sevin (carbaryl). *Mar. Biol.* 28:261.

Weis, P. and J. S. Weis. 1976. Abnormal locomotion associated with skeletal malformations in the sheepshead minnow, *Cyprinodon variegatus*, exposed to malathion. Environ. Res. 12:196-200.

Wells, D. E. and A. A. Cowan. 1982. Vertebral dysplasia in salmonids caused by the herbicide trifluralin. *Environ. Pollut.* 29:249-260.

Wentz, D. A., B. A. Bonn, K. D. Carpenter, S. R. Hinkle, M. L. Janet, F. A. Rinella, M. A. Ulrich, I. R. Waite, A. Laenen, and K. E. Bencala. 1998. Water quality in the Willamette Basin, Oregon, 1991-1995. U. S. Geological Survey Circular 1161, Denver, CO.

White, R., S. Jobling, S. A. Hoare, J. P. Sumpter, and M. G. Parker. 1994. Environmentally persistent alkylphenolic compounds are estrogenic. *Endocrinol.* 135:171-181.

Williamson, A. K., M. D. Munn, S. J. Ryker, R. J. Wagner, J. C. Ebbert, and A. M. Vanderpool. 1998. Water quality in the central Columbia plateau, Washington and Idaho, 1992-1995. U. S. Geological Survey Circular 1144, U. S. Geological Survey, Tacoma, WA. 35 pp.

Wong, C. I., W. R. Kelce, and E. M. Wilson. 1995. Androgen receptor antagonist verses agonist activities of the fungicide vinclozolin relative to hydroxyflutamide. *J. Biol. Chem.* 270:19998-20003.

Woodwell, G. M., C. F. Wurster, Jr., and P. A. Isaacson. 1967. DDT residues in an East Coast estuary: A case of biological concentration of a persistent insecticide. *Science* 156:821-824.

World Health Organization. 1994. *Glyphosate*. Environmental Health Criteria #159. United Nations Environment Programme, Geneva, Switzerland.

World Health Organization. International Programme on Chemical Safety 1996. *Chlorothanlonil.* Environmental Health Criteria 183. Geneva, Switzerland. p. 53.

Glossary, Abbreviations and Scientific Names

2,4-D — 2,4-dichlorophenoxyacetic acid. A herbicide that mimics natural plant hormones.

DDT — dichlorodiphenyltrichloroethane. An insecticide widely used in the past but now illegal in the United States.

DDE — dichlorodiphenylethane. A long-lived breakdown product of DDT.

Eulachon — *Thaleichthys pacificus*

Euryhaline — the ability to tolerate wide ranges of salinity. Euryhaline fish are often found in estuaries where the salinities can range from fresh water to seawater.

Gonopodium — front rays of the anal fin of livebearing fish (Family Poeciliidae) are elongated to form an organ that assists in internal fertilization.

IHN virus — infectious hematopoetic necrosis virus, a major cause of mortalities in Columbia basin salmonids

Lamprey, Pacific — *Lampetra tridentata*

Minnow, fathead — *Pimophales promelas*

Mosquitofish — *Gambusia* sp.

Parr — a juvenile salmonid characterized by vertical bars, or parr marks, along its sides. Fish at this stage are brownish in color and hold territories in the stream.

Ppb — parts per billion; a concentration of pesticide equal to one gram of the compound dissolved in a billion grams of water. Equivalent to micrograms per liter (µg/L).

Ppm — parts per million; a concentration of pesticide equal to one gram of the compound dissolved in a million grams of water. Equivalent to milligrams per liter (mg/L).

Pesticide — anthropogenic chemicals used for control of target organisms. These include insecticides, herbicides, fungicides, and other biocides.

Redd — a depression dug in the streambed gravel in which salmonids lay their eggs.

Salmon, Atlantic — *Salmo salar*

Salmon, chinook — *Oncorhynchus tshawytscha*

Salmon, coho — *Oncorhynchus kisutch*

Salmon, sockeye — *Oncorhynchus nerka*

Shad, American — *Alosa sapidissima*

Smolt — a juvenile salmonid characterized by a uniform silvery color. Fish at this stage undergo a number of physiological changes, lose their ability to hold territories, tend to school, and begin migration toward the sea.

Steelhead — *Oncorhynchus mykiss*, a migratory form of rainbow trout

Surfactant — a chemical added to formulations of pesticides to reduce surface tension and cause wetting by the pesticide solution.

TCDD — 2,3,7,8-tetrachlorodibenzo-dioxin

TCMTB — 2-(thiocyanomethythio)-benzothiazole, an antisapstain fungicide used on timber to be exported.

Trout, brook — *Salvelinus fontinalis*

Trout, brown — *Salmo trutta*

Trout, cutthroat — *Oncorhynchus clarki*

Trout, lake — *Salvelinus naymaycush*

Trout, rainbow — *Oncorhynchus mykiss*

For more information...

Northwest Coalition for Alternatives to Pesticides
P.O. Box 1393
Eugene, OR 97440-1393
(541) 344-5044; fax (541) 344-6923
www.efn.org/~ncap

Oregon Environmental Council
520 SW 6th Ave., Suite 940
Portland, OR 97204
(503) 222-1963; fax (503) 222-1405
www.orcouncil.org

OSPIRG Foundation
1536 SE 11th Ave.
Portland, OR 97214
(503) 231-4181; fax (503) 231-4007
www.pirg.org/ospirg

Institute for Fisheries Resources
PO Box 11170
Eugene, OR 97440-3370
(541) 689-2000; fax (541) 689-2500
www.pond.net/~fish1ifr

WRONCY v. BUREAU OF LAND MANAGEMENT

CIV. NO. 91-1174-FR.

777 F.Supp. 1546 (1991)

Jan WRONCY, Plaintiff,
v.
BUREAU OF LAND MANAGEMENT, Defendant.

United States District Court, D. Oregon.
November 13, 1991.

Gary K. Kahn, Reeves, Kahn & Eder, Portland, Or., for plaintiff.

Charles H. Turner, U.S. Atty., Thomas C. Lee, Asst. U.S. Atty., Portland, Or., for defendant.

OPINION

FRYE, District Judge:

The matter before the court is the motion of plaintiff, Jan Wroncy, for a temporary restraining order.

INTRODUCTION

Wroncy brings this action against the defendant, the Bureau of Land Management (BLM), alleging that it violated the National Environmental Policy Act (NEPA), 42 U.S.C. § 4321 *et seq.*, and the Administrative Procedures Act, 5 U.S.C. § 551 *et seq.*, when it failed to provide for public participation in its decision to fertilize approximately 20,000 acres of forest lands in the Coast Range Mountains of the State of Oregon.

UNDISPUTED FACTS

Wroncy resides in the Coast Range Mountains of the State of Oregon near the target lands included in the fertilization project of the BLM. Wroncy has been active in monitoring the decisions of the BLM in the Coast Range Mountains for a number of years. Wroncy is sensitive to many chemicals, such as formaldehyde and ammonia.

On July 22, 1991, the BLM decided to fertilize approximately 20,000 acres of intensively managed forest lands in the Coast Range Resource Area of the Eugene District of the Coast Range Mountains with a urea fertilizer to be applied aerially in the form of pellets. In the environmental assessment prepared for this project, the BLM concluded "that completion of the proposed action does not constitute a major Federal action having a significant effect on the human environment. Therefore, an environmental impact statement or a supplement to the existing environmental impact statement is not necessary and will not be prepared." Exhibit B to Memorandum in Support of Plaintiff's Motion for Temporary Restraining Order (Decision Record, Environmental Assessment No. OR090-91-40), p. 1. The BLM had provided no public notification of the proposed fertilization project during the decision making stages and had provided no public notification of the Decision Record of July 22, 1991 within the thirty-day period following the decision.

Formaldehyde-based additives are used in the production of the fertilizer.

On September 16, 1991, Wroncy became aware of the fertilization project through a conversation with an official from the BLM. On September 19, 1991 and September 24, 1991, public affairs posters, which described the project in detail, were posted at six locations in the general area in an effort to inform the public about the fertilization project.

As a result of a telephone call from Wroncy to Daniel Bowman of the BLM, the

[777 F.Supp. 1548]

BLM sent Wroncy a copy of the environmental assessment relating to the fertilization project issued on July 22, 1991 and other documents upon which the BLM relied in making its environmental assessment.

The contract to apply the fertilizer was awarded by the BLM on October 3, 1991. The project is in progress.

On October 25, 1991, Wroncy filed a notice of appeal with the BLM requesting administrative review of the decision of the BLM to forego a formal period of time for public comment and the finding of the BLM in the environmental assessment of July 22, 1991 that the fertilization project would have no significant environmental impact.

On October 29, 1991, the Coast Range Area Manager of the BLM informed Wroncy in writing that the BLM was unable to consider her notice of appeal under the regulations on the grounds that the action of the BLM was final on July 22, 1991 and the regulations of the BLM did not provide for an appeal at the time she filed her appeal. The Coast Range Area Manager further stated that:

> The Interdisciplinary Team completed the EA, ROD, and FONSI and I signed them on July 22, 1991. Based on the fact that an almost identical project was reviewed by the public and state and local clearinghouse without comment and completed in two other Resource Areas on the Eugene District during January, 1991, that no public comments were received on the EA for this project, and the fact that no significant impacts were identified by the team of specialists who completed the EA; it was my decision to not make this EA available for public review prior to making a final decision.

Exhibit D to Memorandum in Support of Plaintiff's Motion for Temporary Restraining Order, p. 1.

In a letter dated November 4, 1991, Wroncy's physician states that Wroncy has reported to him that "she has been acutely ill for the past few days in association with application of methylene diurea containing formaldehyde urea binder to forest lands in her vicinity." Exhibit A to Memorandum in Support of Plaintiff's Motion for Temporary Restraining Order, p. 1. Her physician reports that Wroncy's symptoms are consistent with her past medical history of chemical sensitivity. Her physician states that further exposure to formaldehyde and ammonia-containing urea can be expected to have an additional detrimental impact on her health and could potentially become very serious with the risk of long-term permanent effects. *Id.*

Wroncy would have submitted information to the BLM about the impacts on her health and the environment of formaldehyde and ammonia had she been informed of the project in a timely manner. Wroncy moves this court for an order temporarily restraining the BLM from implementing the project set out in Environmental Assessment No. OR090-91-40 until a hearing can be held on her motion for a preliminary injunction.

APPLICABLE STANDARD

The Ninth Circuit uses two tests for determining whether a court should grant a preliminary injunction: a traditional test and an alternative test. *Caribbean Marine Servs. Co. v. Baldrige,* 844 F.2d 668 (9th Cir.1988). Under the traditional test, the court must consider 1) the likelihood that the moving party will prevail on the merits; 2) whether the balance of irreparable harm favors the moving party; and 3) whether the public interest favors the moving party. *Northern Alaska Envtl. Ctr. v. Hodel,* 803 F.2d 466, 471 (9th Cir.1986).

Under the alternative test, the court must consider 1) whether the motion raises serious questions on the merits; and 2) whether the balance of hardships tips decidedly in favor of the moving party. *Los Angeles Memorial Coliseum Comm'n v. Nat'l Football League,* 634 F.2d 1197, 1202 (9th Cir.1980).

CONTENTIONS OF THE PARTIES

Wroncy contends that the BLM has violated the provisions of NEPA and the regulations relating to NEPA when it failed to provide public participation in its decision

[777 F.Supp. 1549]

to carry out the fertilization project. Wroncy argues that her environment and her health, as well as her statutory rights under NEPA, have been and continue to be irreparably injured by the actions of the BLM.

The BLM contends that the hardship to the BLM in stopping the fertilization project outweighs the hardship to Wroncy in continuing the fertilization project. The BLM further argues that the finding of the BLM that there was no significant impact on the environment because of this project was a correct finding and should stand.

1. *SUCCESS ON THE MERITS*

40 C.F.R. § 1501.4(e)(1) governs the actions of the BLM in this case and provides:

In determining whether to prepare an environmental impact statement the Federal agency shall:

....

(e) Prepare a finding of no significant impact (§ 1508.13), if the agency determines on the basis of the environmental assessment not to prepare a statement.

(1) The agency shall make the finding of no significant impact available to the affected public as specified in § 1506.6.

Section 1506.6(b)(3) sets out a number of ways an agency can provide notice of a finding of no significant impact in the case of an action, the effects of which are primarily of local concern, including publication in local newspapers or through other media and direct notice to concerned individuals and organizations.

On December 9, 1988, in the case of *Friends of Walker Creek Wetlands, Inc. v. BLM,* Civil No. 88-779-MA, 1988 WL 163053, the Honorable Malcolm M. Marsh, United States District Court Judge, reviewed the above-cited provisions of the law and issued an injunction enjoining a timber harvest and requiring a 45-day period of public notice and comment on the grounds that "there was no public notice of the [finding of no significant impact]." Opinion, p. 15, lns. 18-19.

The BLM has long been required to make its finding of no significant environmental impact available to the affected public. There appears to have been no effort to make public the environmental assessment of July 22, 1991 in which the BLM concluded that there would be no significant environmental impact as a result of the fertilization project. The court finds, therefore, that Wroncy has shown a high likelihood of success on the merits in this case.

2. *IRREPARABLE INJURY AND THE PUBLIC INTEREST*

If environmental harm is shown to be likely, the balance of harm will usually favor the issuance of an injunction to protect the environment. *Amoco Prod. Co. v. Village of Gambell, Alaska,* 480 U.S. 531, 545, 107 S.Ct. 1396, 1404, 94 L.Ed.2d 542 (1987). Wroncy has shown that personal injury to her as a result of the harm upon her environment is likely.

Further, the public interest in this case favors the issuing of the requested preliminary injunction. NEPA is a purely procedural statute which requires meaningful public involvement in decision making in accordance with specific regulations. A disregard for public participation is not a wrong compensable in any other manner. In this situation, the responsibility of the BLM was clearly set out in the regulations to NEPA and has been explained by prior case law. It would not be in the public interest for the court to excuse the BLM for its failure to involve the public because the fertilization project is now in progress.

CONCLUSION

The motion of Jan Wroncy for a temporary restraining order is granted. The BLM and all of its officers, agents, servants and employees are temporarily restrained from taking any action pursuant to its Decision Record accompanying Environmental Assessment No. OR090-91-40 approved by the Coast Range Area Manager on July 22, 1991.

Pursuant to Federal Rule of Civil Procedure 65, this order shall expire ten days from the date of its entry. The motion of Jan Wroncy for a preliminary injunction

[777 F.Supp. 1550]

shall be heard on November 21, 1991 at 9:00 a.m. The BLM shall file responses to the motion for a preliminary injunction by November 19, 1991. Any reply shall be filed by November 20, 1991. All briefings should be hand-delivered or faxed between the parties.

FIVE RIVERS

GYPSY MOTH SPRAY PROGRAM

Here is a link that has more information about Gypsy Moths:

Photos from: http://www.forestry.ubc.ca/fetch21/FRST308/lab5/lymantria_dispar/gypsy.html

- Website - Gypsy Moth in North America
- The Complete Tree List showing Gypsy Moth Preferences
- Gypsy Moth Life Cycle
- Natural Enemies of the Gypsy Moth
- The 20 most common Gypsy Moth host trees

AERIAL SPRAY IN FIVE RIVERS, OREGON ON APRIL 30, 2003

Small pictures of spray in Five Rivers

Big pictures of spray in Five Rivers

Small pictures with descriptions

AERIAL SPRAY #2 IN FIVE RIVERS, OREGON ON MAY 9, 2003

Small pictures with descriptions

Pictures of spray in Five Rivers

Set 1 (Digital)

Set 2

DOCUMENTS:

Oregon Department of Agriculture/USDA Forest Service/USDA APHIS:

- Final Environmental Assessment - Gypsy Moth Lincoln County - March 11, 2003
- 1995 Gypsy Moth Management in US - Final Environmental Impact Statement Summary
- Forest Service Fisheries Report and Biological Evaluation, Febuary 20, 2003, by Michael Northrop

Oregon Department of Environmental Quality (DEQ):

- NPDES permit Evaluation Report 1-21-2003
- NPDES permit 3-05-2003
- DEQ memorandum Gypsy Moth Spray Water Quality Analysis 3-05-2003
- Notice of Public Hearing - April 10, 2003 - NPDES permit issued 3-07-2003

Oregon Department of Agriculture:

- ODA News 1-23-03
- ODA News 4-16-03
- ODA News 4-21-03

INFORMATION RESOURCES:

Information about pesticides and alternatives to pesticides:

- Northwest Coalition for Alternatives to Pesticides (NCAP)

PubMed: Information from the National Library of Medicine about chemicals and toxicity:

- http://www.ncbi.nlm.nih.gov/entrez/query.fcgi?CMD=search&DB=PubMed

Search Engine:

- http://www.google.com/

Excellent Website for Information on Gypsy Moth spray program problems:

- http://www.nosprayzone.org/

FORAY 48B - BIOPESTICIDE:

FORAY 48B Biopesticide by Valent BioSciences Corporation, EPA No. 73049-46:

- FORAY 48B Long Label (agriculture)
- FORAY 48B label obtained upon request from manufacturer
- FORAY 48B obtained from ODA
- FORAY 48B Material Safety Data Sheet (MSDS)
- Manufacturer's Manual for FORAY and Dipel (see page 4 "ample surfactants")

"Inert" ingredients in FORAY 48B:

- Freedom of Information Act request for list of inerts in FORAY 48B

1, 2-Benzisothiazolin-3-one (BIT) biocide in FORAY 48B: enter the chemical name or formula name in the PubMed Search Link above for information about toxicity

Organo Silicone Surfactant:

See page 22-23 for list of "inerts" in FORAY 48B, and Table 8 page 36 for settling time for spray:

- Airborne Exposures to Bacillus thuringiensis var. kurstaki During Gypsy Moth Eradication by Teschke, et. al 2000

HEALTH AND ENVIROMENTAL EFFECTS OF FORAY 48B:

- The Case Against Overhead Pesticide Spraying in the Township of Waskesiu Lake, Prince Albert National Park of Canada by Saskatchewan Environmental Society, January 28, 2003
- Health Risk Assessment of the 2002 Aerial Spray - Auckland, March 2002
- Human Exposures to Bacillus thuringiensis after Aerial Applications by Levin 2002
- Airborne Exposures to Bacillus thuringiensis var. kurstaki During Gypsy Moth Eradication by Teschke, et. al 2000

Effects of Pesticides on Salmon - Report on NCAP's website:

- Diminishing Returns: Salmon Decline and Pesticide by Dr. Richard Ewing

Toxicity of the "inert" of organo silicone surfactant, trisiloxane by Cowles, et al, 2000:

- "Inert" Formulation Ingredients with Activity: Toxicity of Trisiloxane Surfactant Solutions to Twospotted Spider Mites (Acari: Tetranychidae) by Cowles, et al 2000
- Multistate Research Project Proposal - Bioherbicides with Organosilicone Surfactants
- Characteristics of Organosilicone Surfactants and Their Effects on Sulfonylurea Herbicide Activity by Jinxia Sun, PhD Thesis 1996

"Active" ingredient = Bacillus Thuringiensis:

- Bacillus Thuringiensis (B.T.) Factsheet from NCAP
- Human Exposures to Bacillus thuringiensis after Aerial Applications by Levin 2002

Spray Drift Articles:

- Northwest Coalition for Alternatives to Pesticides - "Indiscriminately From the Skies"
- Pesticide Action Network North America - Secondhand Pesticides: Airborne Pesticide Drift in California, May 7, 2003

COMMENTS AND RESPONSES:

Comments:

- Comments to ODA on Draft EA for Gypsy Moth Eradication

Responses to comments:

- DEQ's Response to Comments on the NPDES permit
- Forest Service Response to Comments on Final Environmental Assessment

AGENCIES INVOLVED AND CONTACT INFORMATION

FOREST SERVICE (USDA FS):

- http://www.fs.fed.us/r6/
- http://www.fs.fed.us/r6/nwfp.htm
- http://www.fs.fed.us/r6/siuslaw/
- http://www.fs.fed.us/r6/siuslaw/contact/

OREGON DEPARTMENT OF AGRICULTURE (ODA):

- http://www.oda.state.or.us/
- http://www.oda.state.or.us/Plant/index.html

ANIMAL AND PLANT HEALTH INSPECTION SERVICE (USDA APHIS):

- http://www.aphis.usda.gov/

OREGON DEPARTMENT OF ENVIRONMENTAL QUALITY (DEQ):

- http://www.deq.state.or.us/
- http://www.deq.state.or.us/wq/

ENVIRONMENTAL PROTECTION AGENCY (US EPA):

- http://www.epa.gov/region10/

NATIONAL MARINE FISHERIES SERVICE (NMFS) - NOAA FISHERIES:

- http://www.nwr.noaa.gov/

NATIONAL OCEANIC AND ATMOSPHERIC ADMINISTRATION (NOAA):

- http://www.noaa.gov/

UNITED STATES FISH AND WILDLIFE SERVICE (USFWS) - PACIFIC REGION (1):

- http://pacific.fws.gov/

OREGON GOVERNOR TED KULONGOSKI:

- http://governor.oregon.gov/

FEDERAL AVIATION ADMINISTRATION:

- http://www1.faa.gov/

ADMINISTRATIVE ACTIONS INFORMATION:

APPEAL OF FOREST SERVICE DECISION RECORD AND FINDING OF NO SIGNIFICANT IMPACT (DR AND FONSI):

- Appeal Information
- Forest Service Decision Record and Finding of No Significant Impact

LEGAL ACTION INFORMATION:

Oregon Department of Agriculture Form - Report of Alleged Loss against a Pesticide Operator

Oregon Statute regarding Notice of Tort Claim against the State of Oregon

Lane County Schools and Forestry Spray Mapping Project 1990 – 2006

Mapleton School

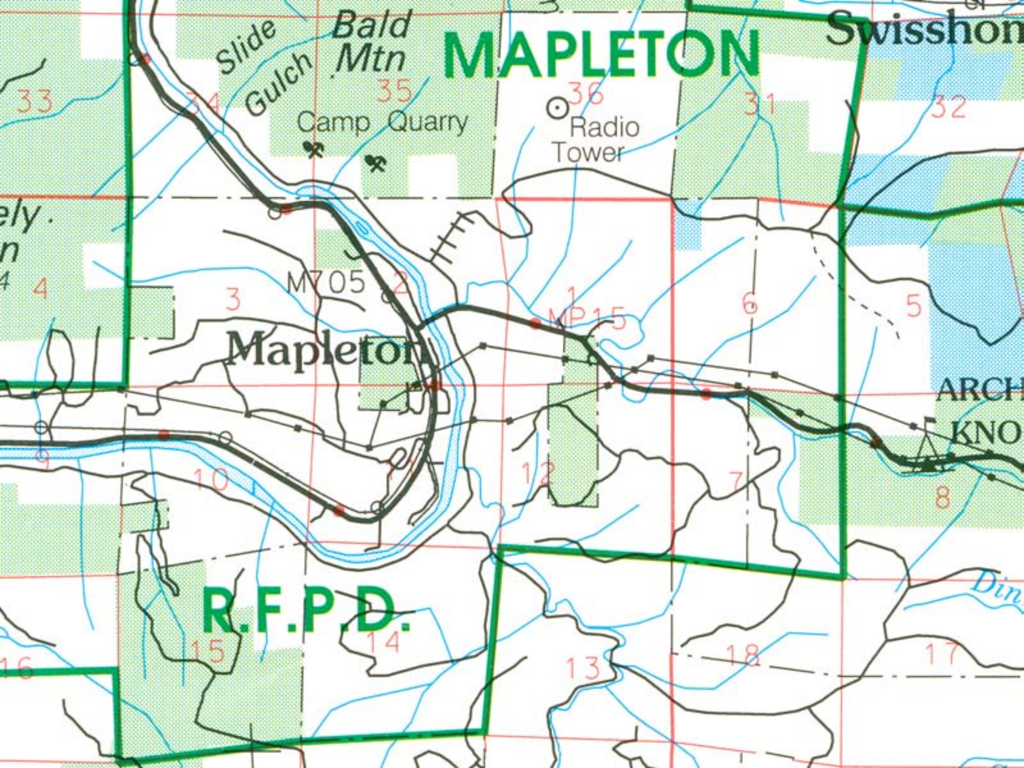
Mapleton School District ~ Mapleton, Oregon

Presented by
Oregon Toxics Alliance and Forestland Dwellers

Mapped using data from Oregon Department of Forestry Forest Activity Computerized Tracking System (FACTS)


STOP


MAPLETON SCHOOL



MAPLETON
Swisshom
Slide Gulch
Bald Mtn
Camp
Quarry
Radio Tower
33
34
35
36
31
32
4
3
2
1
6
5
M705
MP15
Mapleton
ARCH
KNO
9
10
11
12
7
8
R.F.P.D.
16
15
14
13
18
17
Din

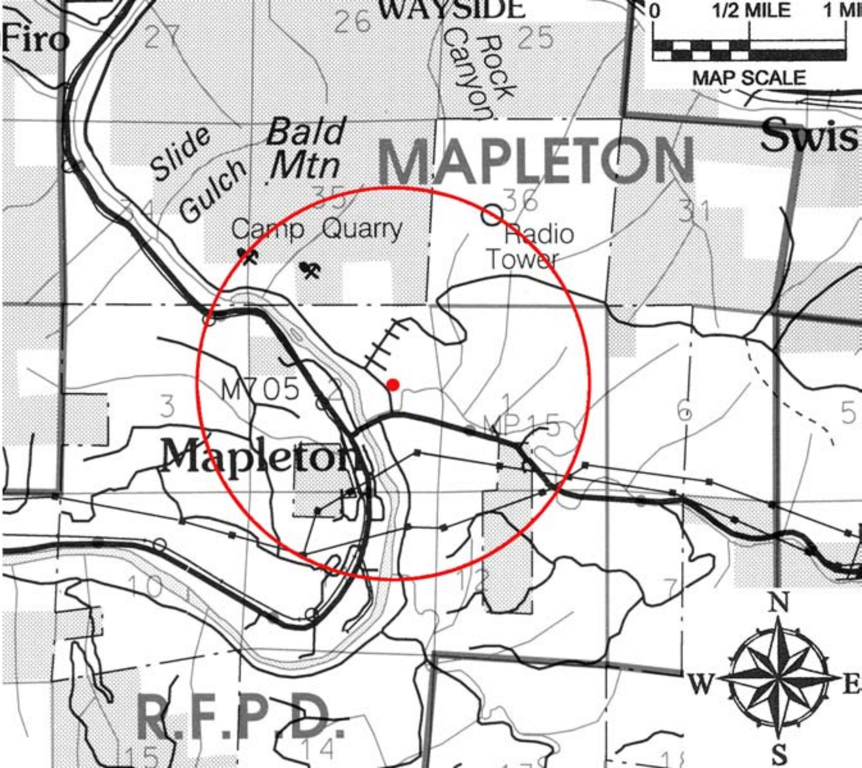
Map Notes

Red Dot is School Location

Red Circle is proposed
1 mile radius buffer

Colored Areas are Sprayed

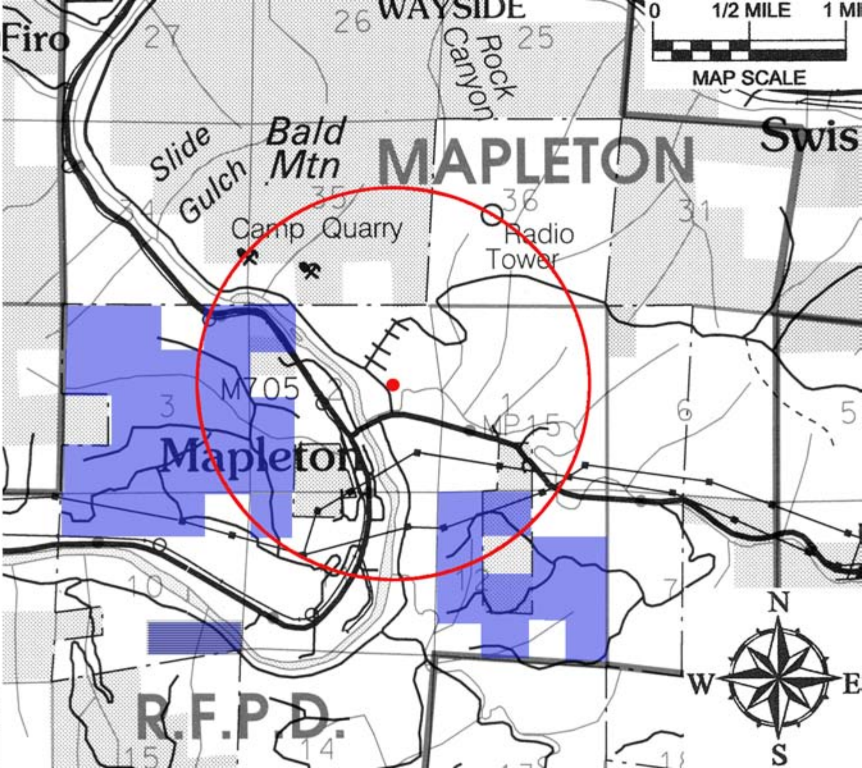
Mapleton
Schools



Mapleton Schools

Legend

1990

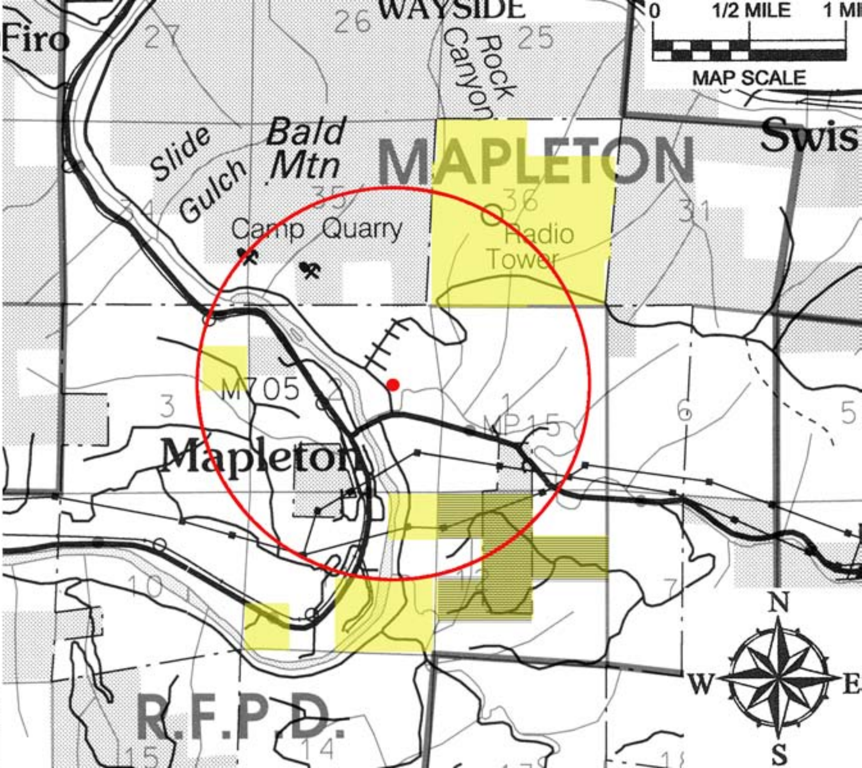


Aerial

Roadside

Mapleton Schools Legend

1991

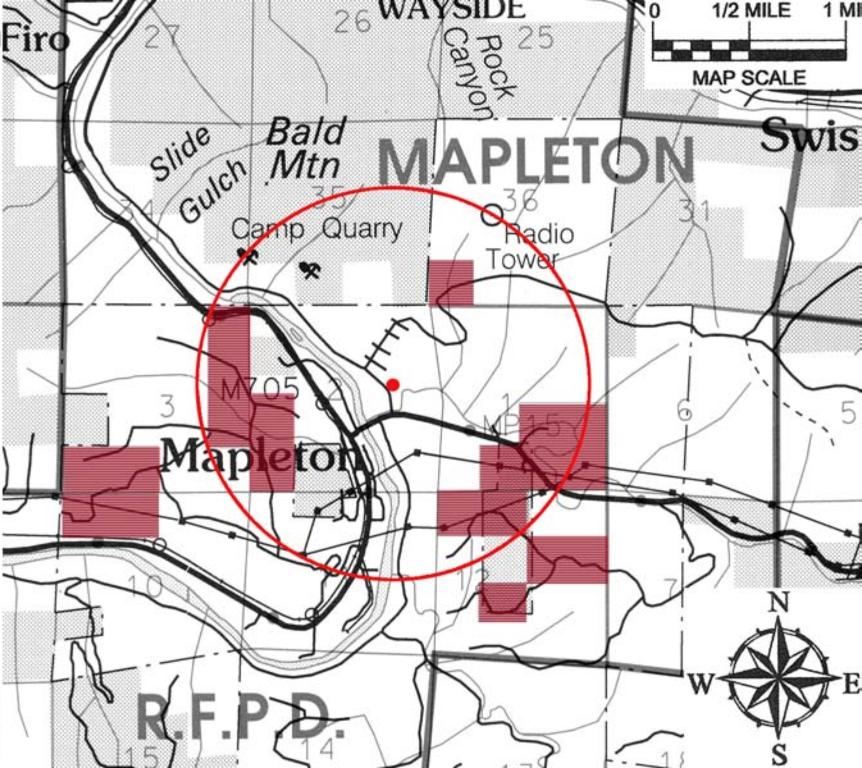


Aerial

Roadside

Mapleton Fire Schools Legend

1992

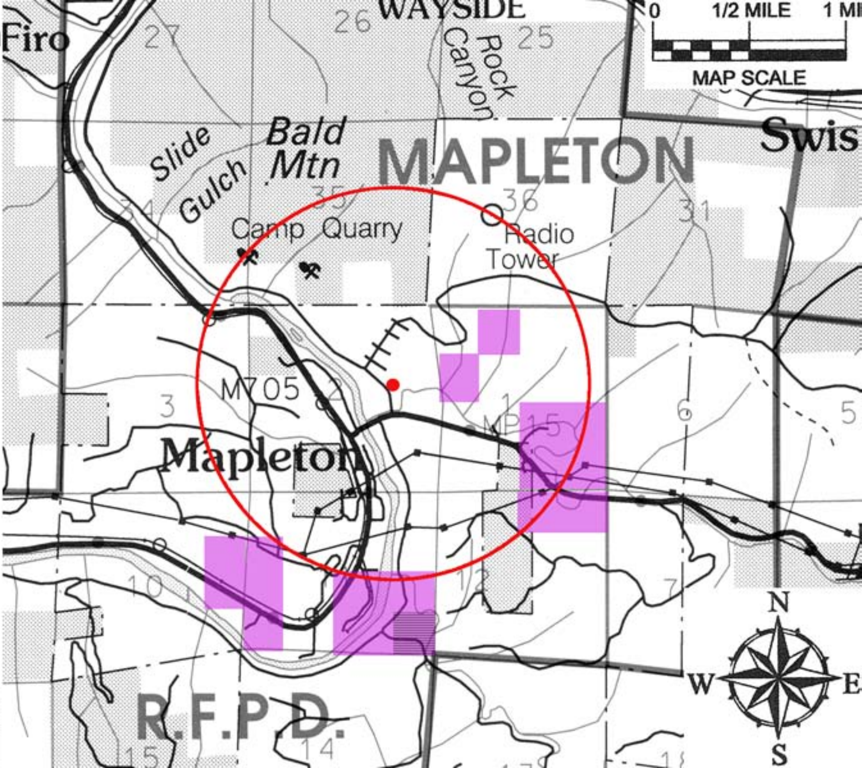


Aerial

Roadside

Mapleton Fire Schools Legend

1993

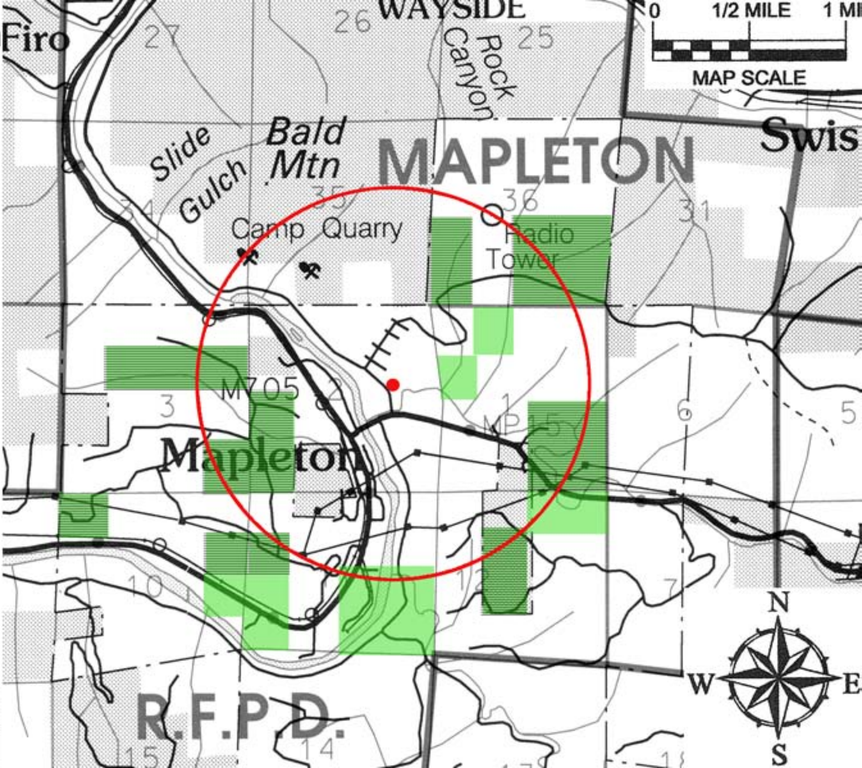


Aerial

Roadside

Mapleton Schools Legend

1994



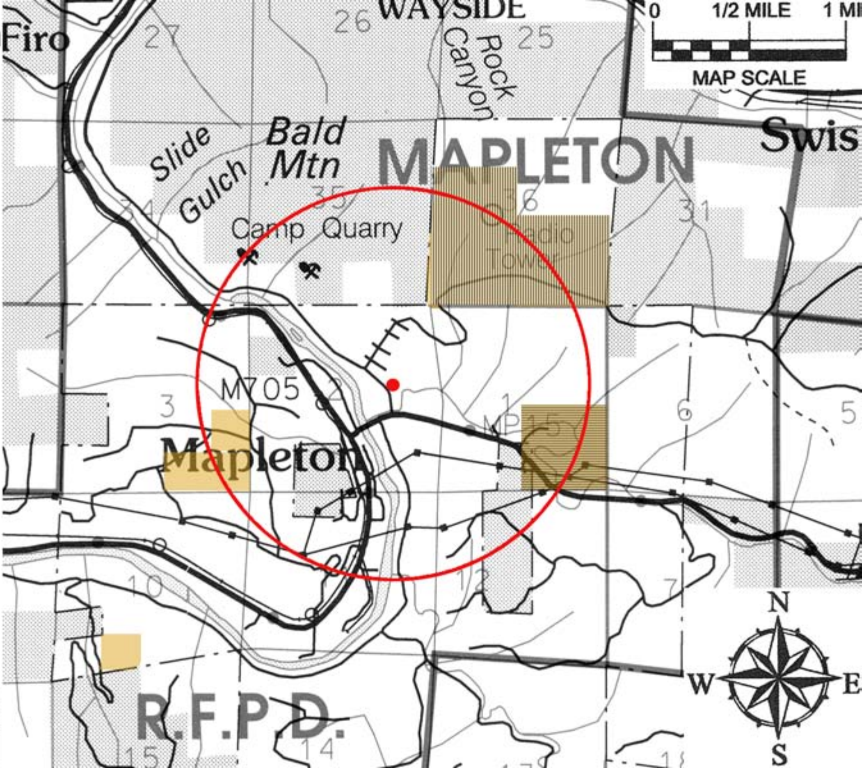
Aerial

Roadside

Mapleton Schools

Legend

1995

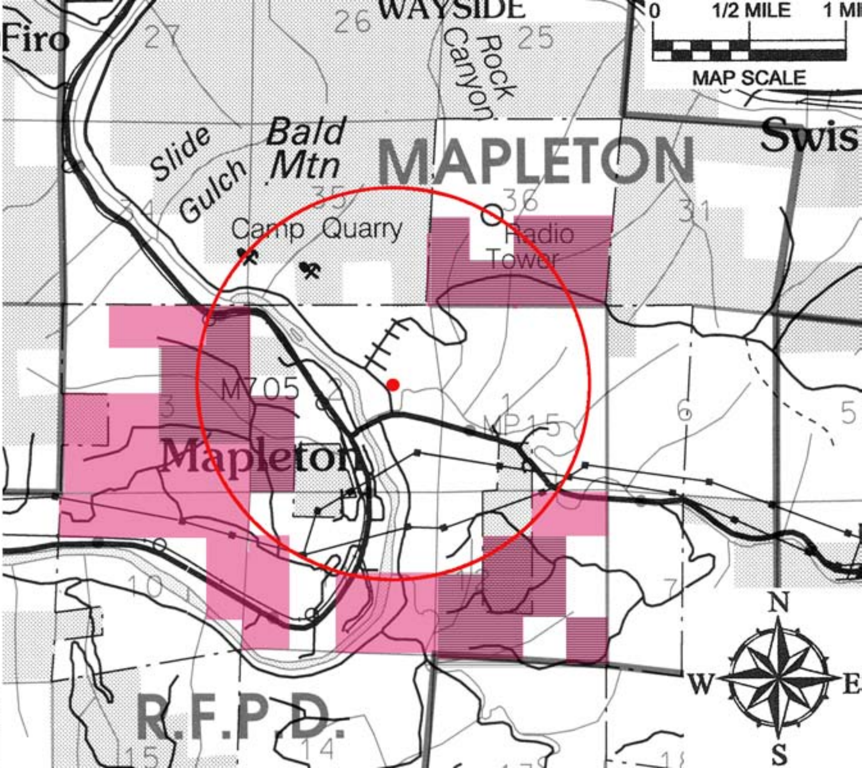


Aerial

Roadside

Mapleton Schools Legend

1996



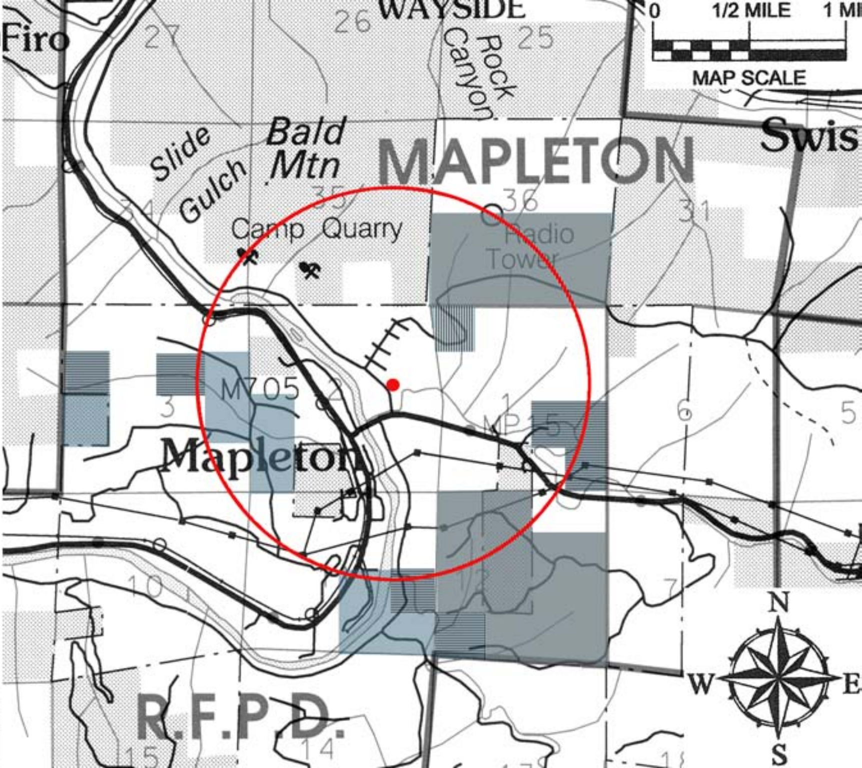
Aerial

Roadside

Mapleton Schools

Legend

1997



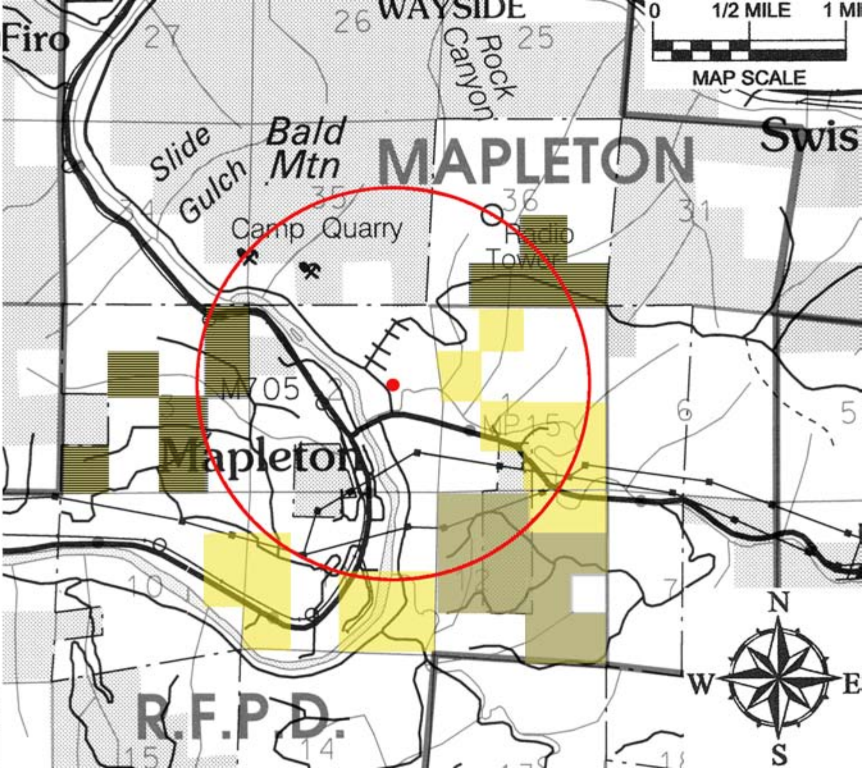
Aerial

Roadside

Mapleton Schools

Legend

1998



Aerial

Roadside

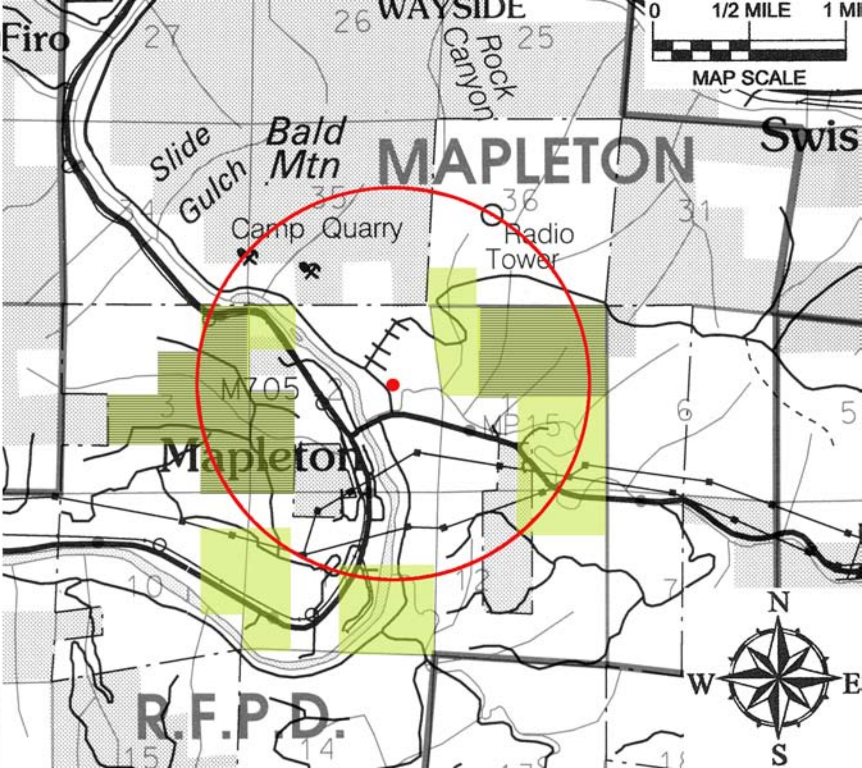
Mapleton Schools

Legend

1999

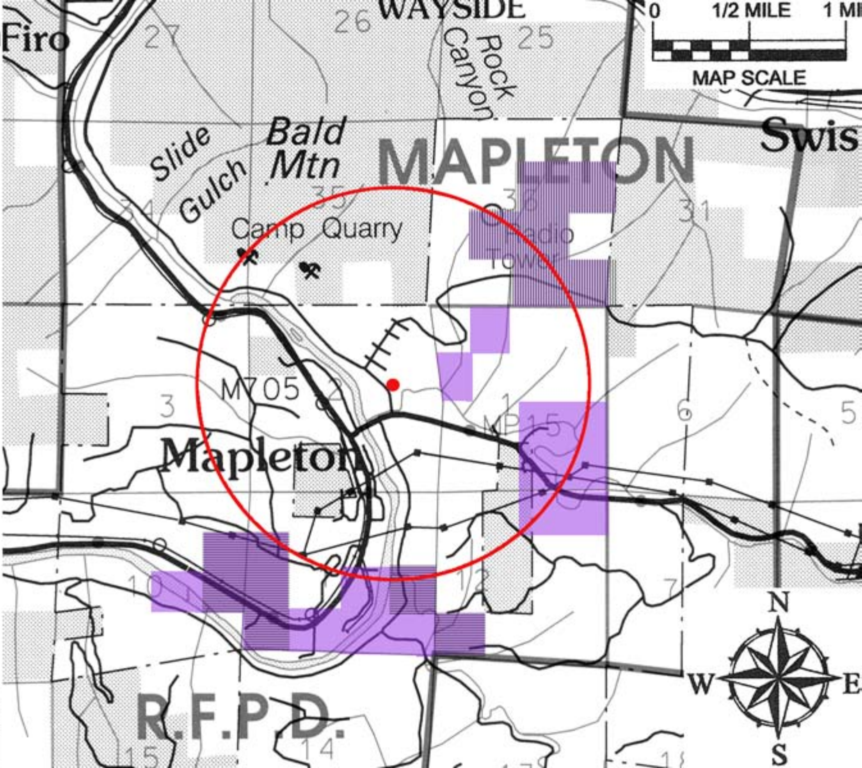
Aerial

Roadside



Mapleton Schools

Legend



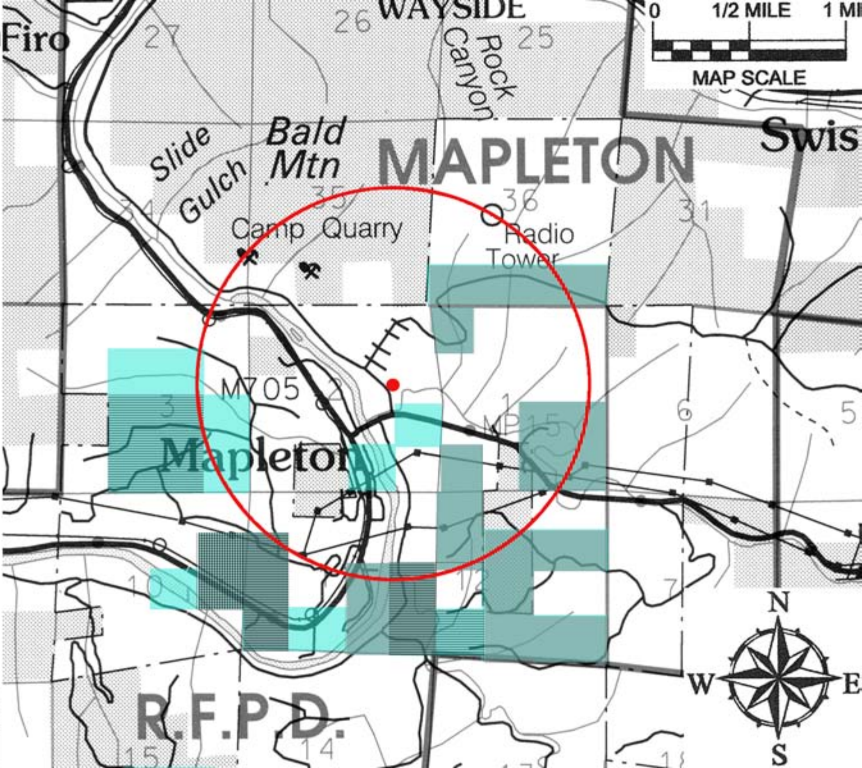
2000

Aerial

Roadside

Mapleton Schools

Legend



2001

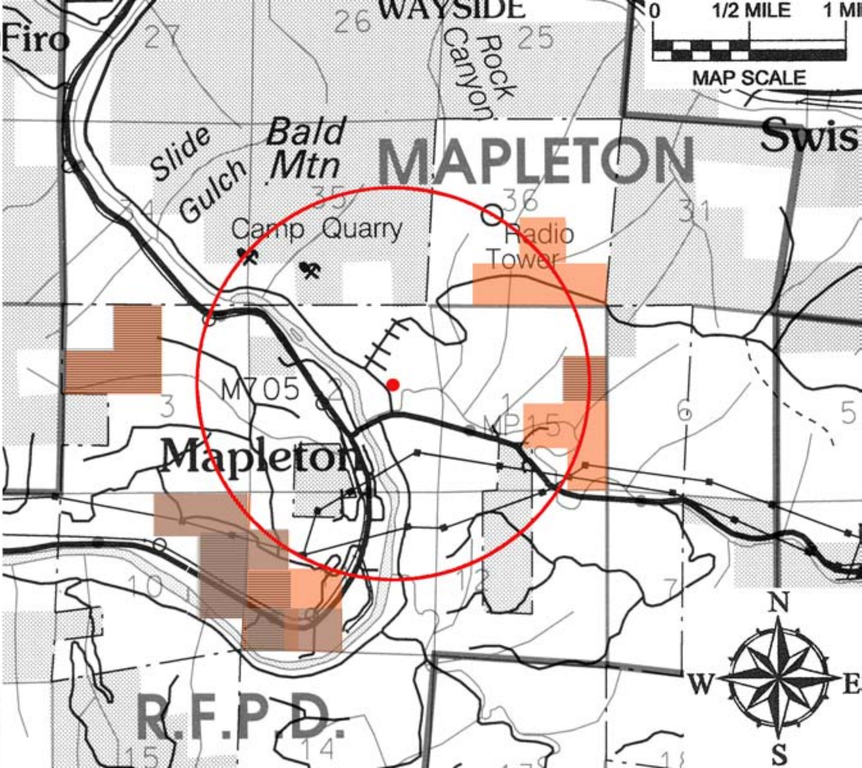
Aerial

Roadside

Mapleton Schools
Legend

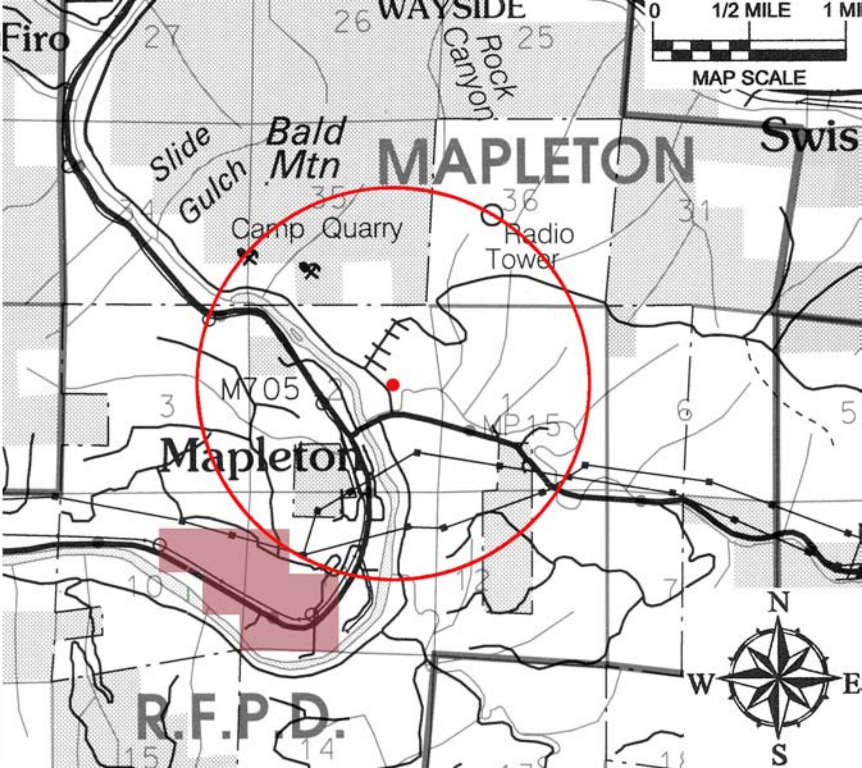
2002

Aerial
Roadside



Mapleton Schools Legend

2003



Aerial

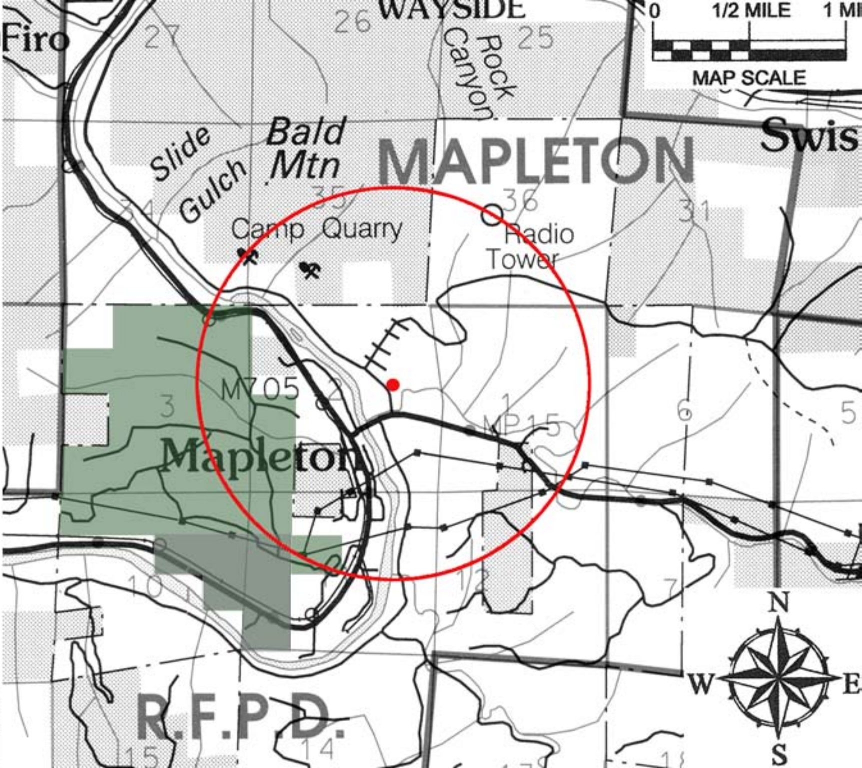
Roadside

Mapleton Schools Legend

2004

Aerial

Roadside



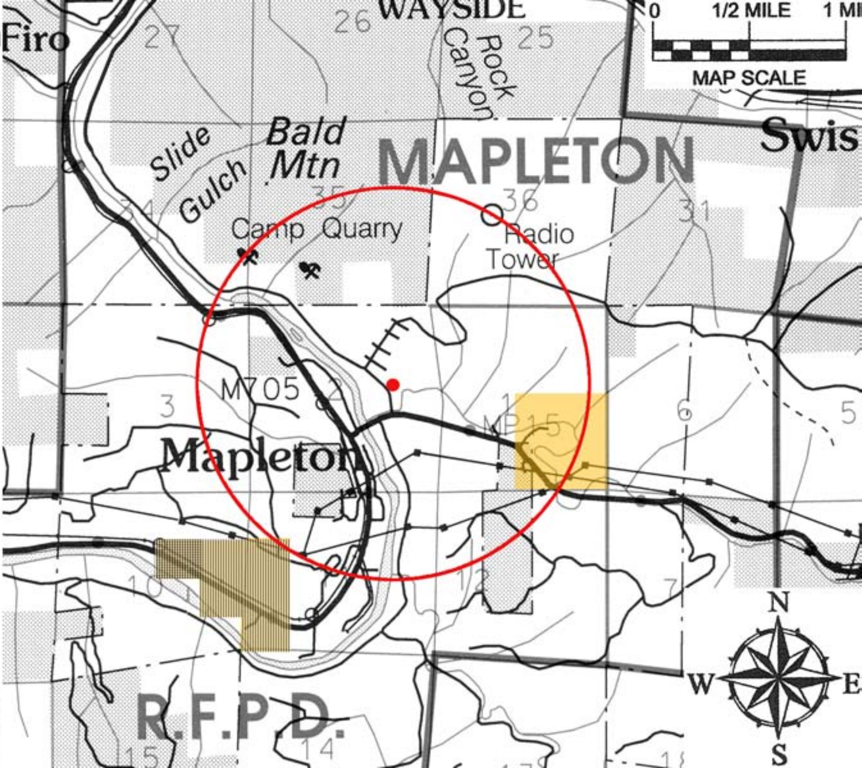
Mapleton Fire Schools

Legend

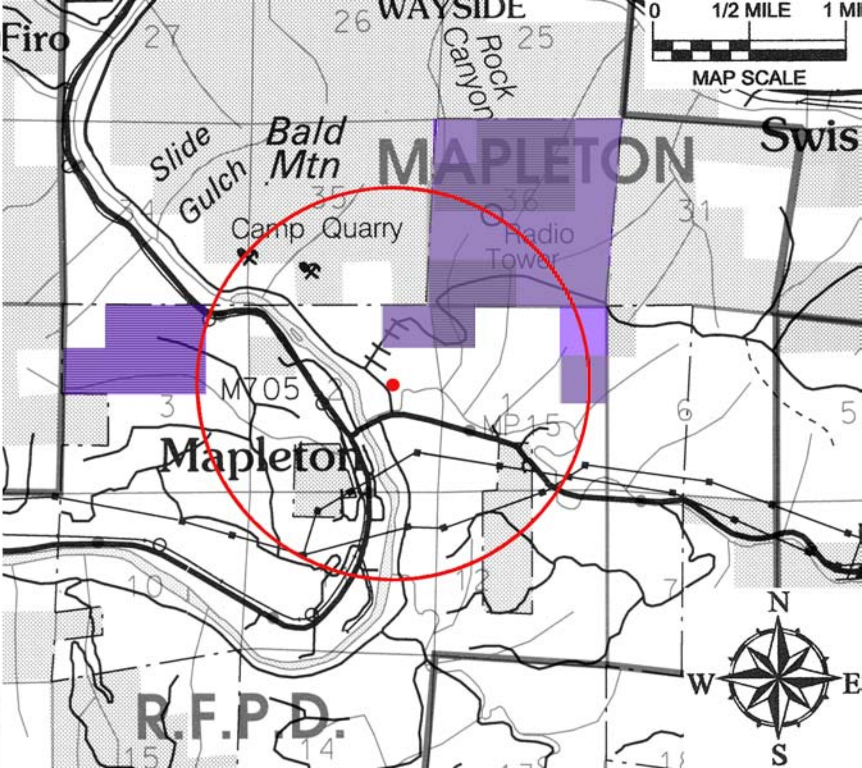
2005

Aerial

Roadside



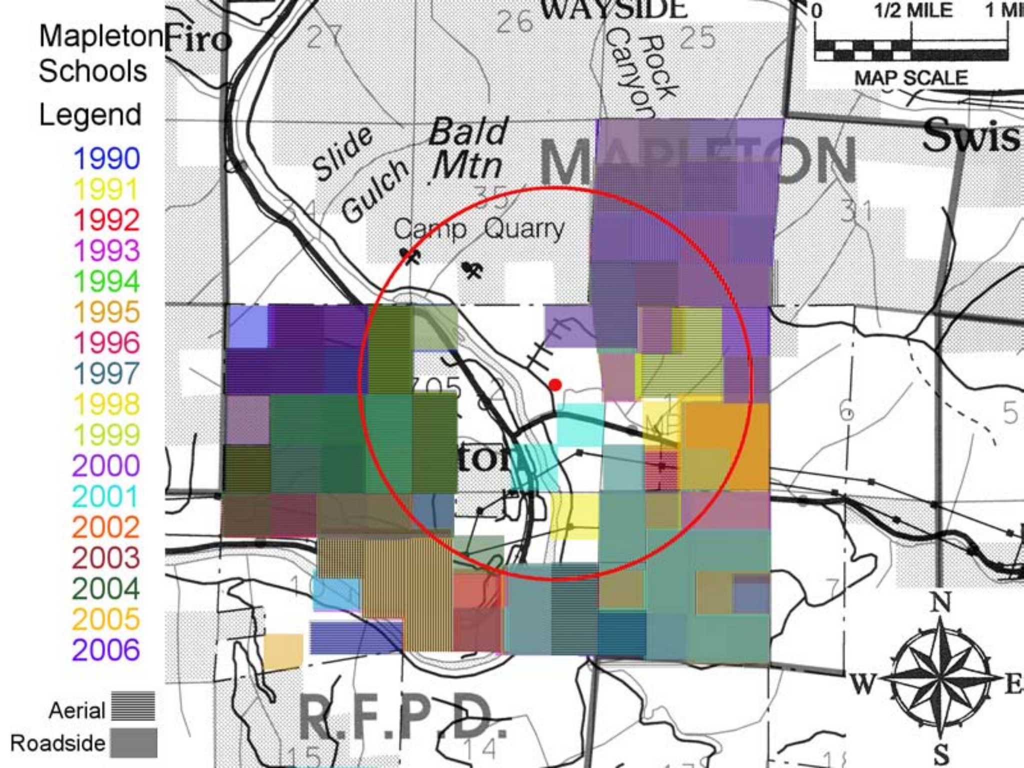
Mapleton Fire Schools Legend



2006

Aerial

Roadside


Mapleton Schools
Legend
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
Aerial
Roadside
0
1/2 MILE
1 MI
MAP SCALE
WAYSIDE
Firo
Rock Canyon
Slide Gulch
Bald Mtn
MAPLETON
Swis
Camp
Quarry
R.F.P.D.
N
S
E
W

Oregon Toxics Alliance
P.O. Box 1106
Eugene OR 97440
Phone/Fax 465-8860
http://www.oregontoxics.org/

Forestland Dwellers
P.O. Box 5954
Eugene, OR 97405
Phone 342-8332
http://www.forestlanddwellers.org/

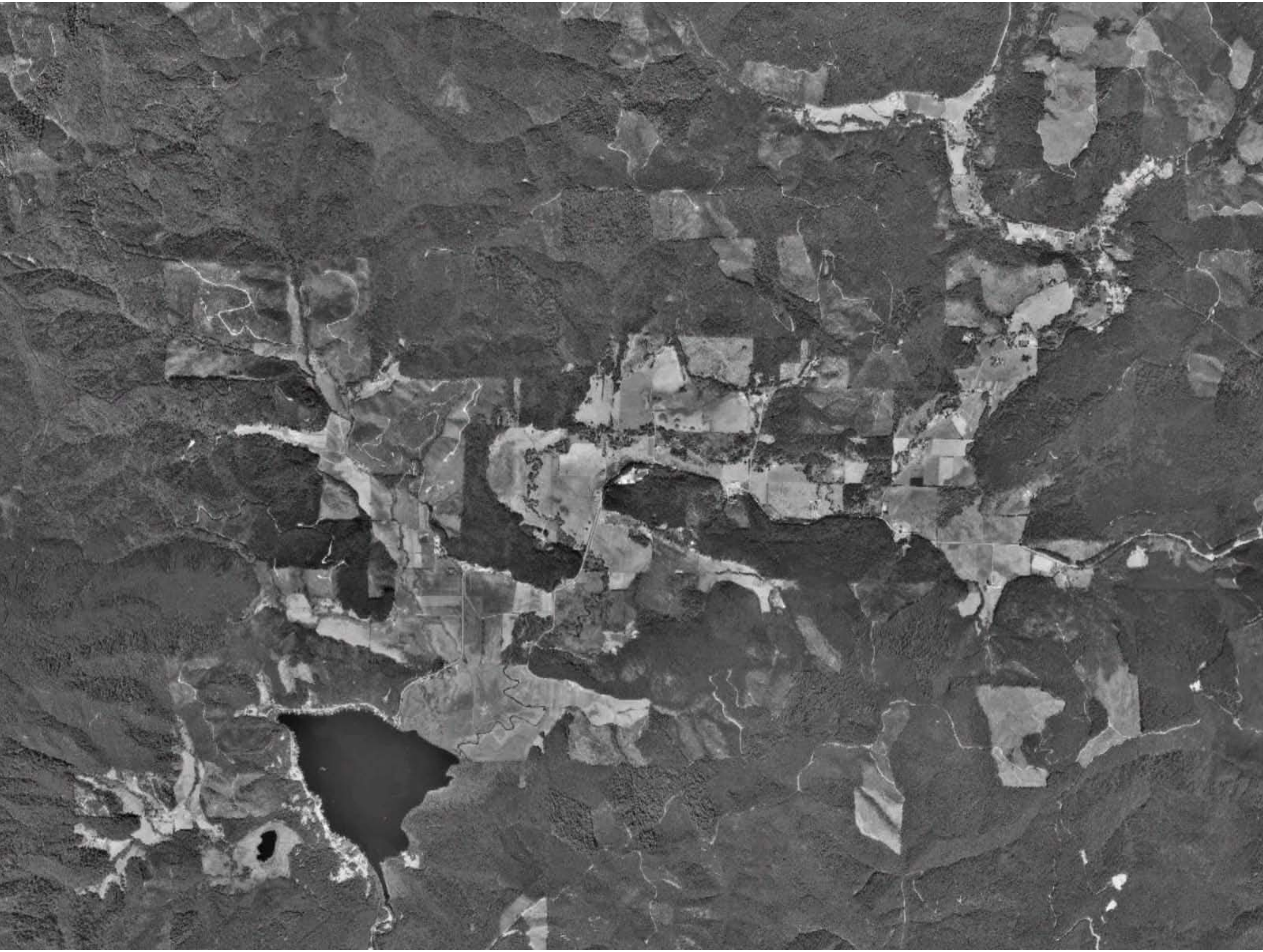
Lane County Schools and Forestry Spray Mapping Project 1990 – 2006

Triangle Lake School

Blachly School District ~ Blachly, Oregon

by Oregon Toxics Alliance and Forestland Dwellers

Mapped using data from Oregon Department of Forestry Forest Activity Computerized Tracking System (FACTS)




20264
Blachly Grange Road
TL
Welcome
to
Triangle Lake
Schools
LAKER COUNTRY
Class Motto
Over the years we've seen
losses and gains,
but through it all we've
held together.
Dedicated by
Class of 2004


POST RD
STOP

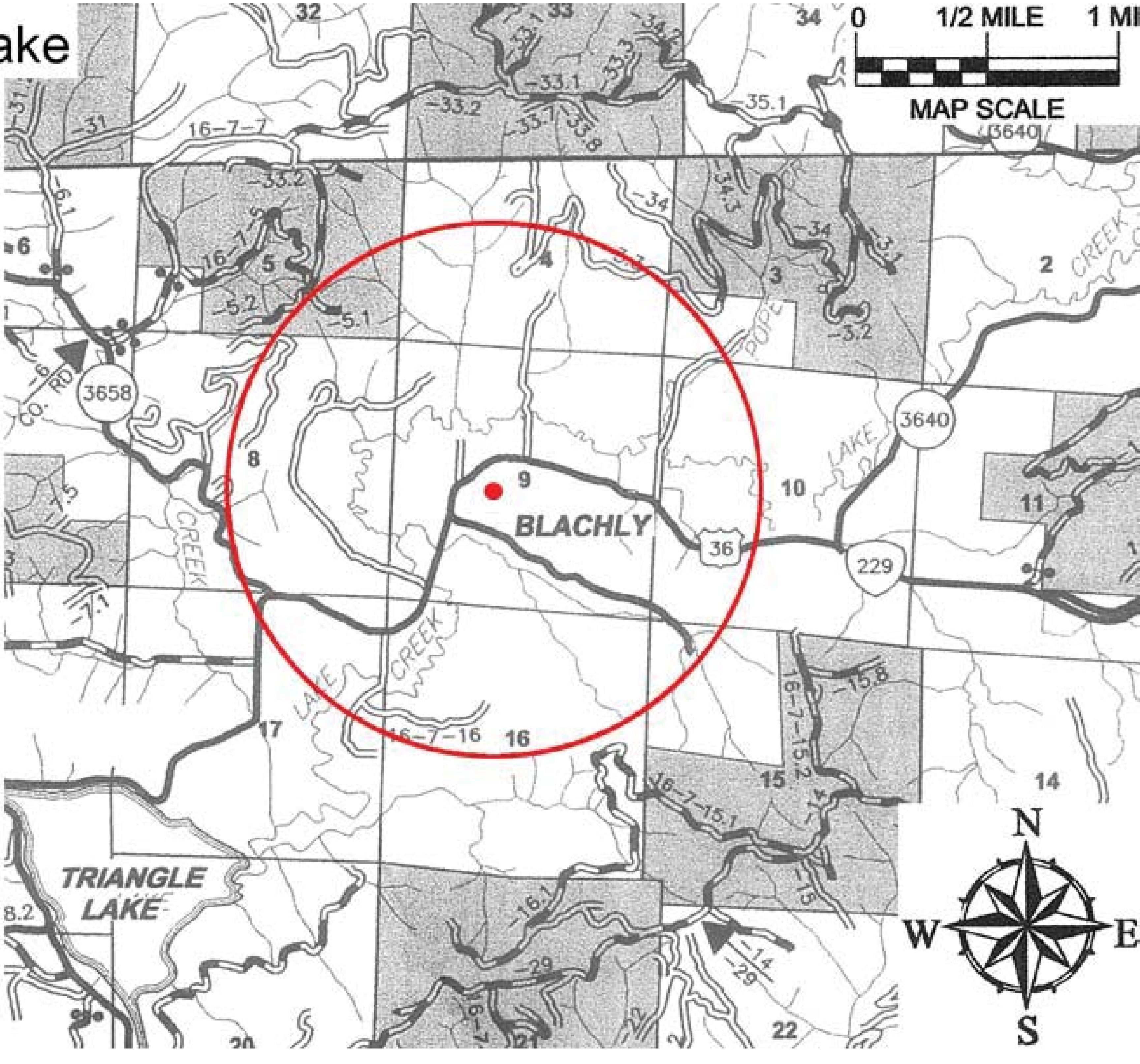
Map Notes

Red Dot is School Location

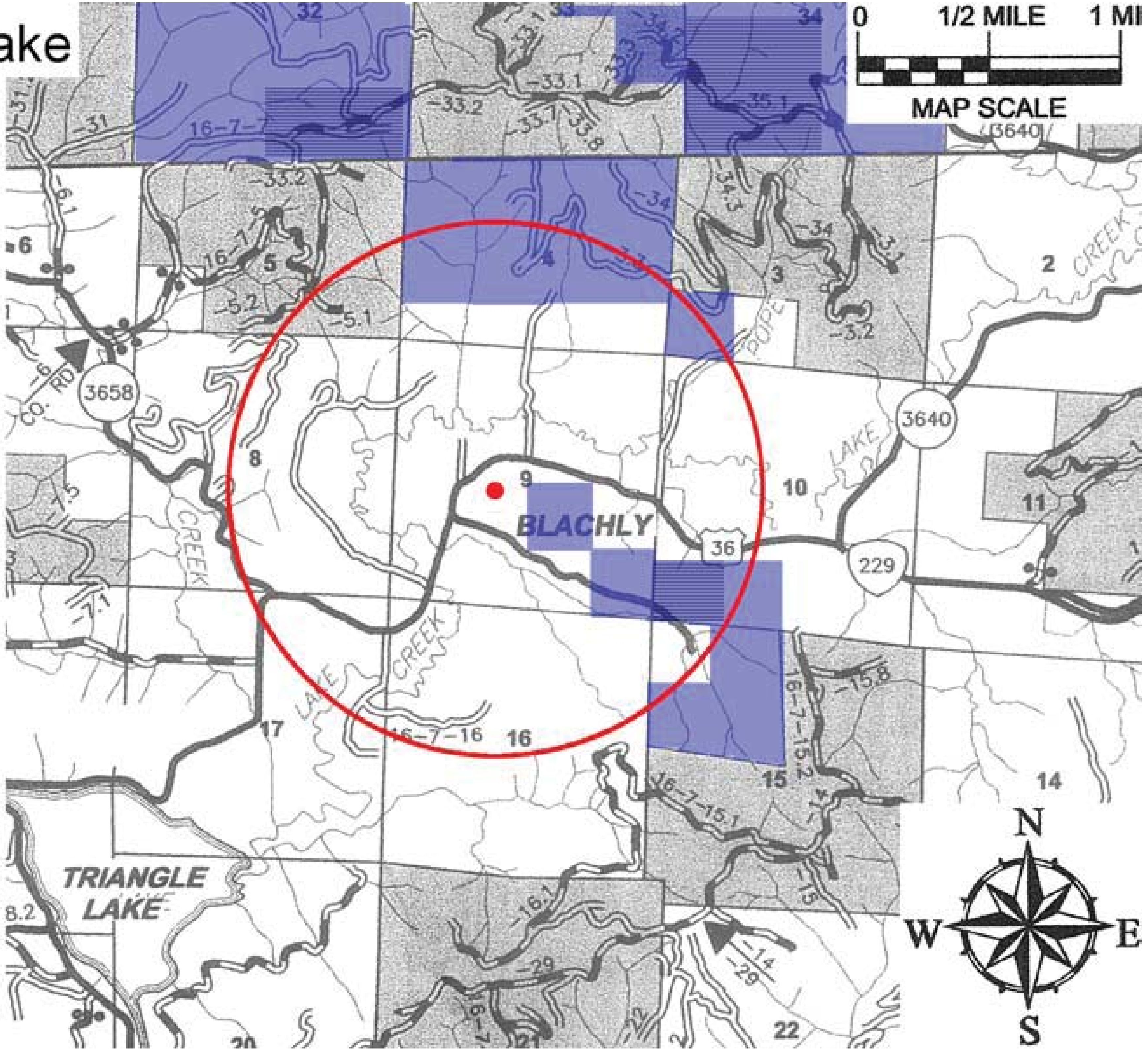
Red Circle is proposed
1 mile radius buffer

Colored Areas are Sprayed

Triangle Lake School

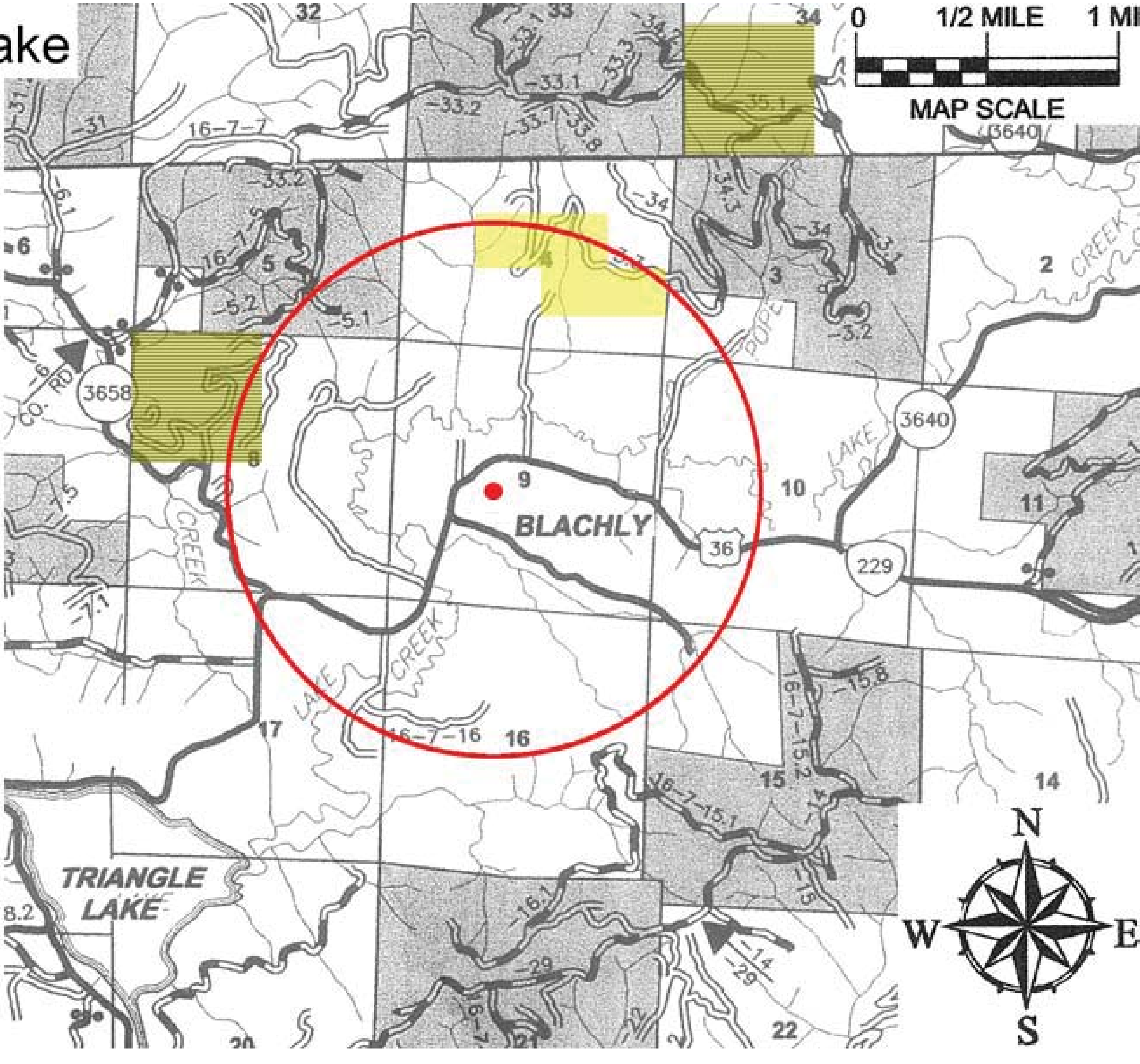


Triangle Lake School Legend 1990



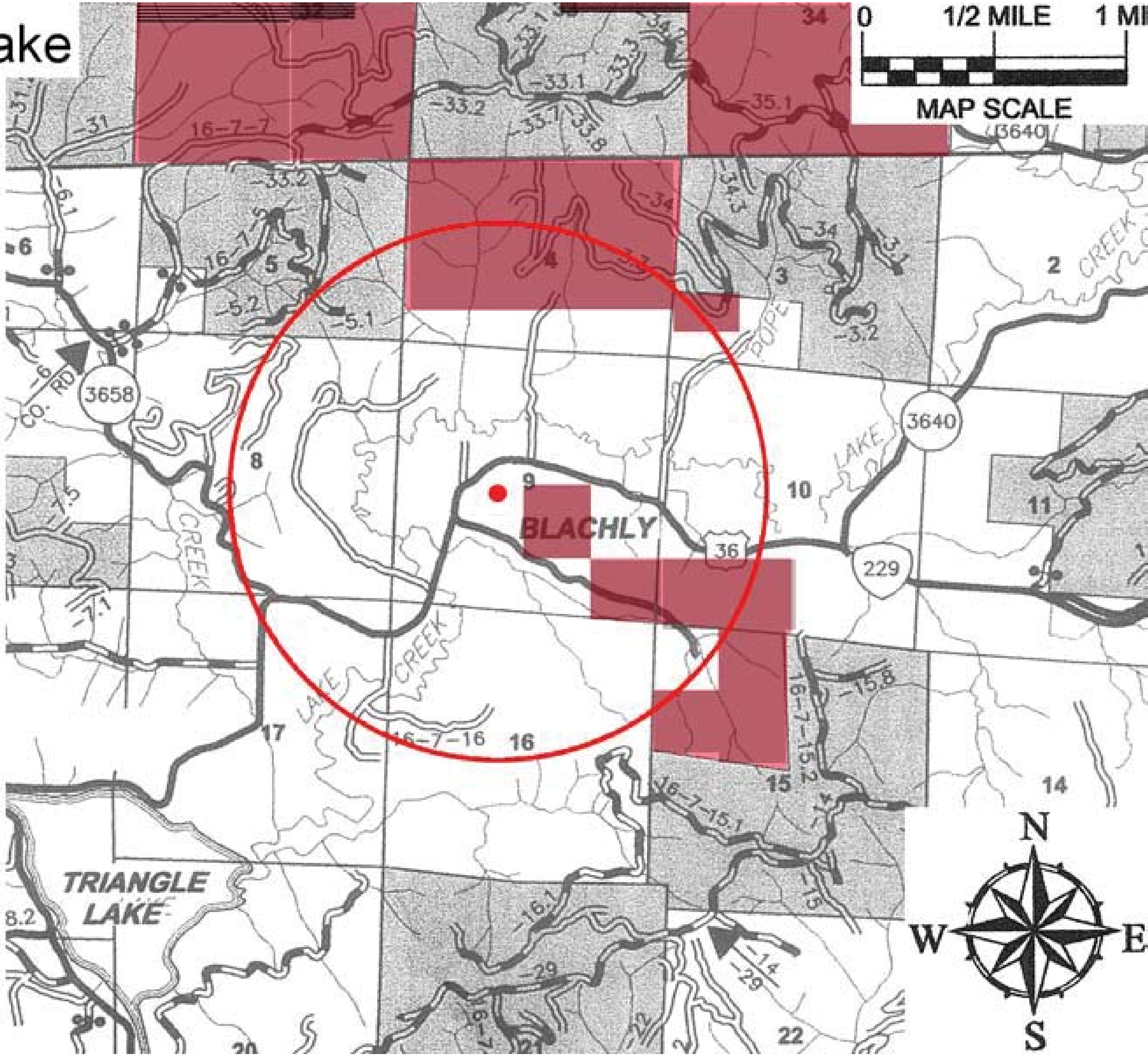
Triangle Lake School Legend

1991



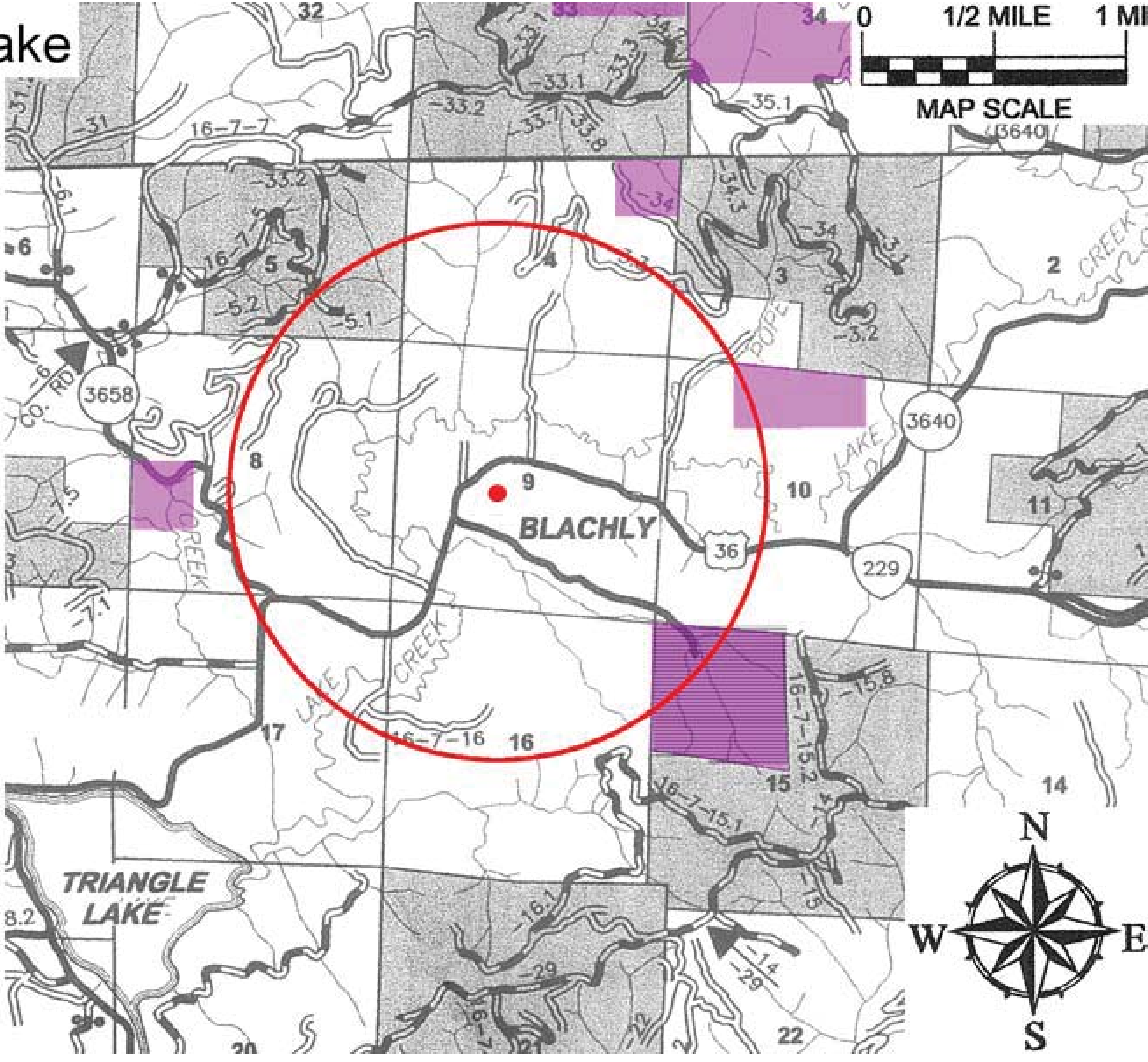
Triangle Lake
School
Legend

1992



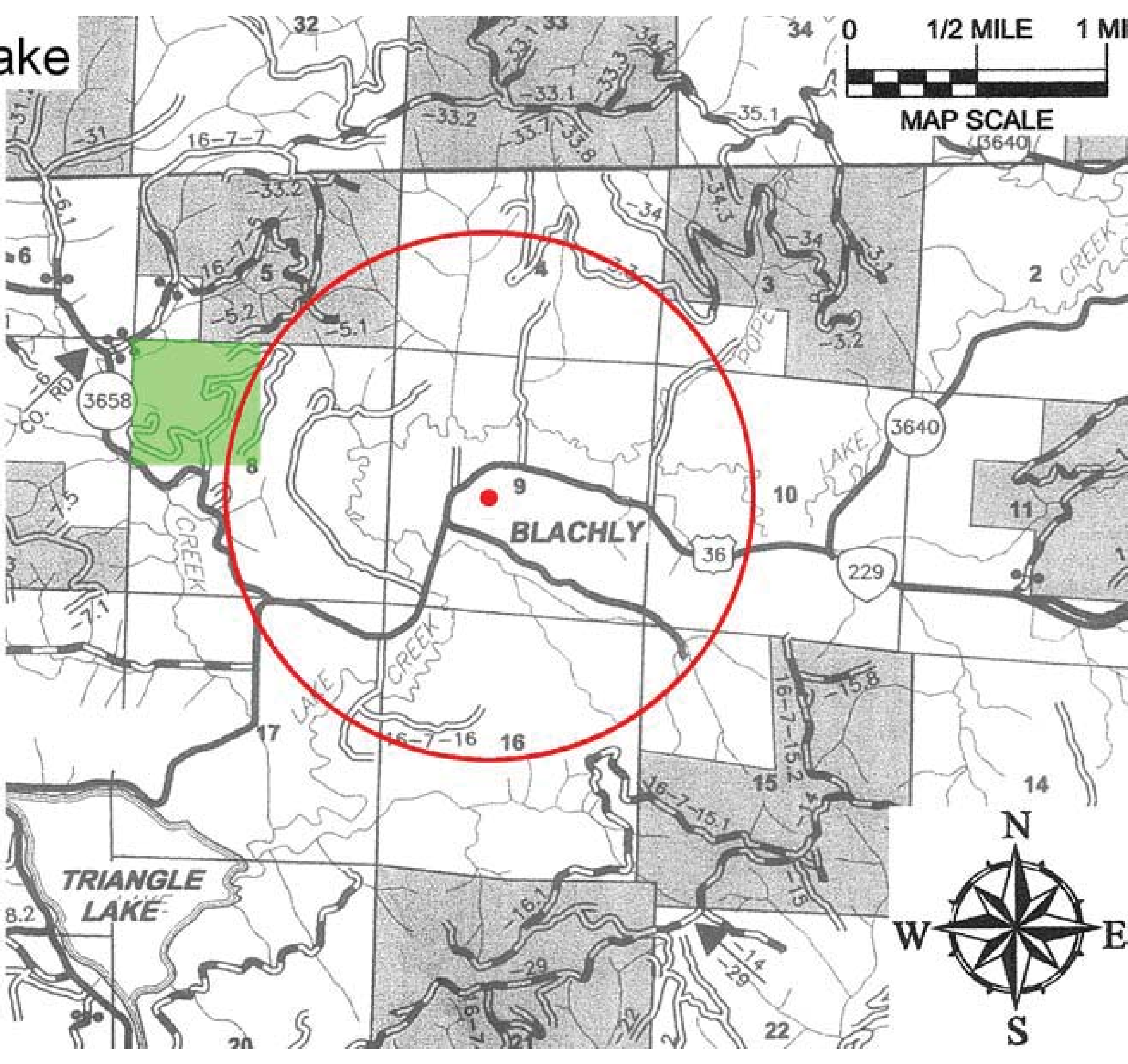
Triangle Lake School Legend

1993



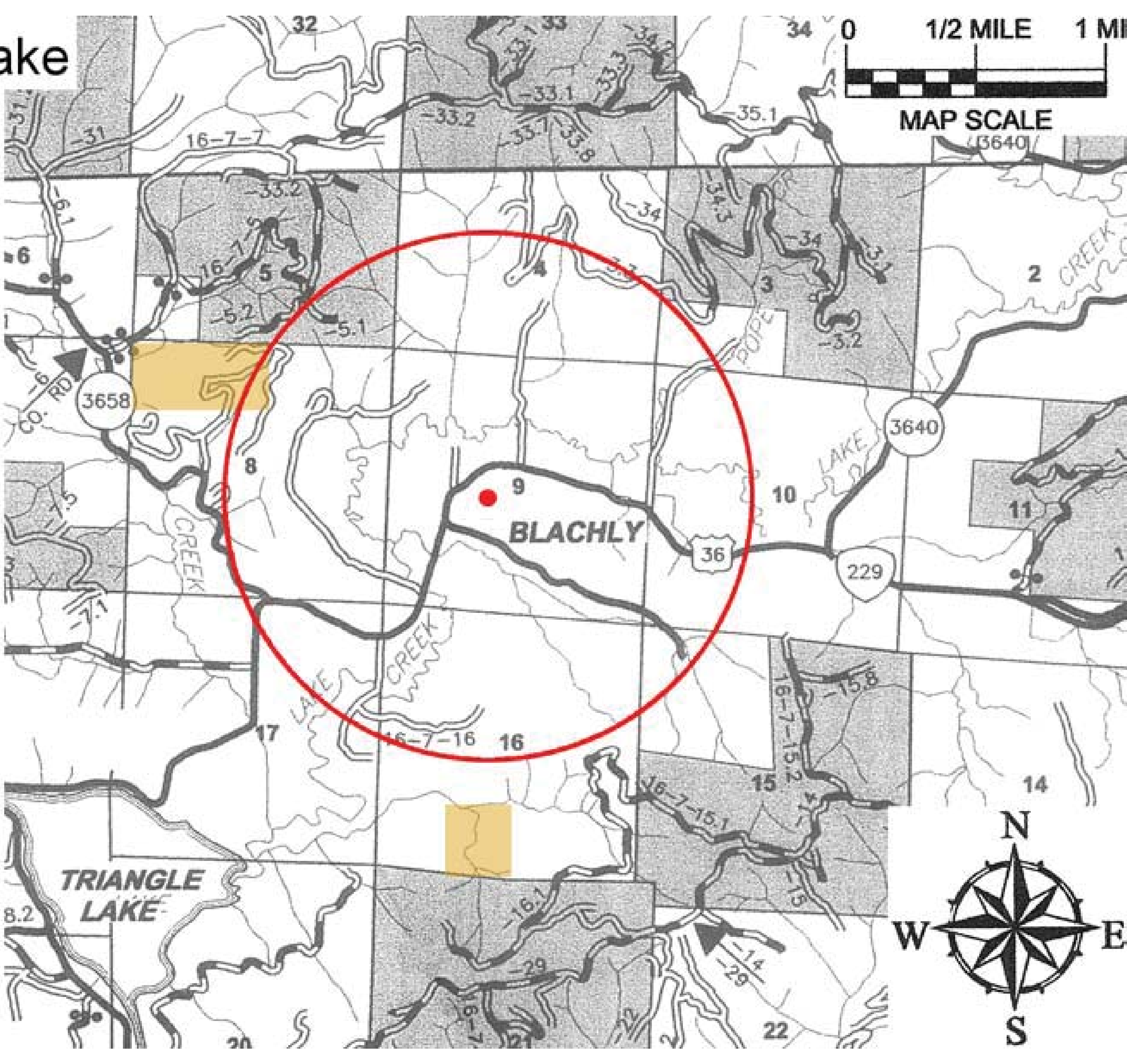
Triangle Lake School Legend

1994



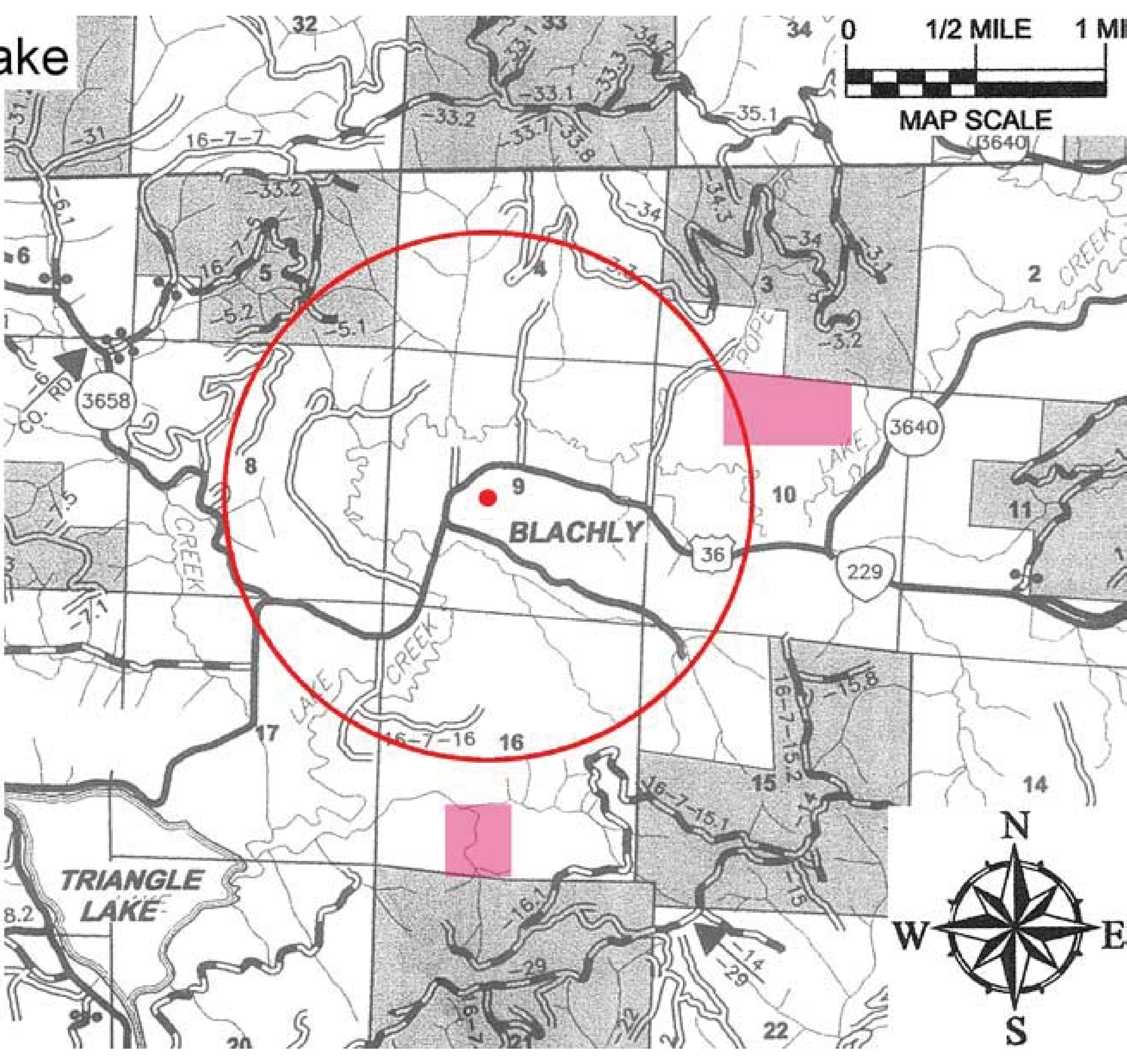
Triangle Lake School Legend

1995



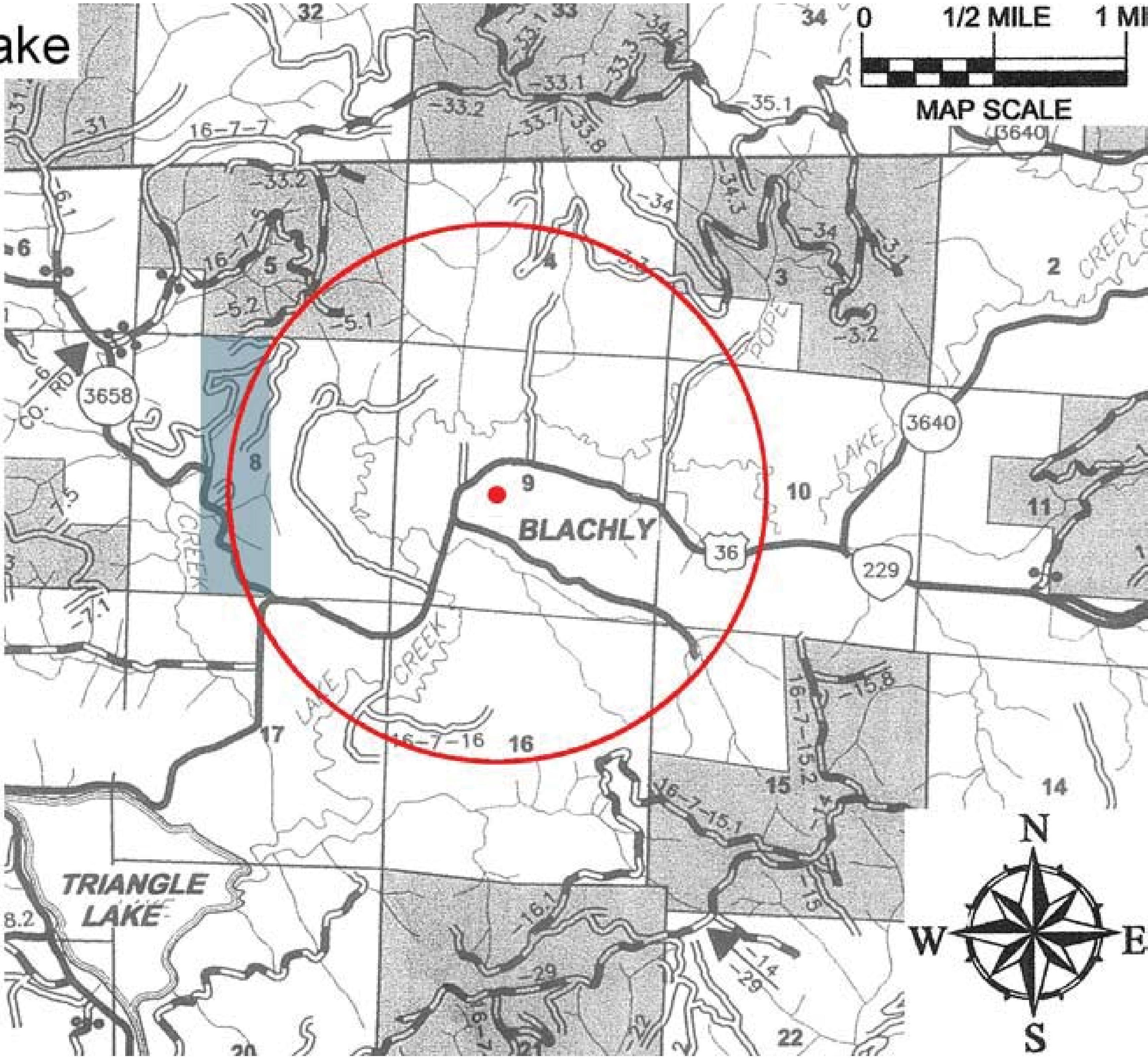
Triangle Lake School Legend

1996



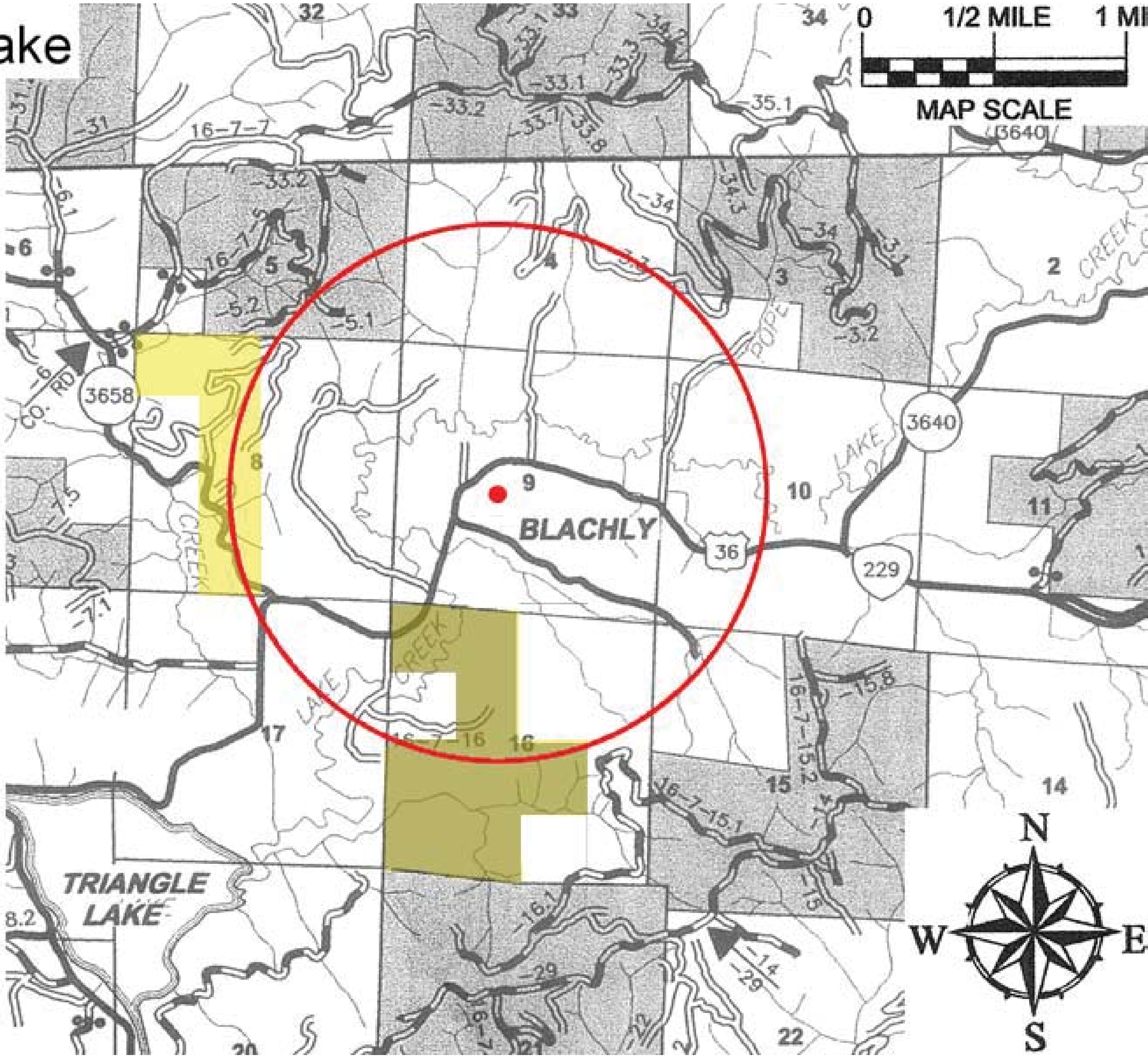
Triangle Lake School Legend

1997



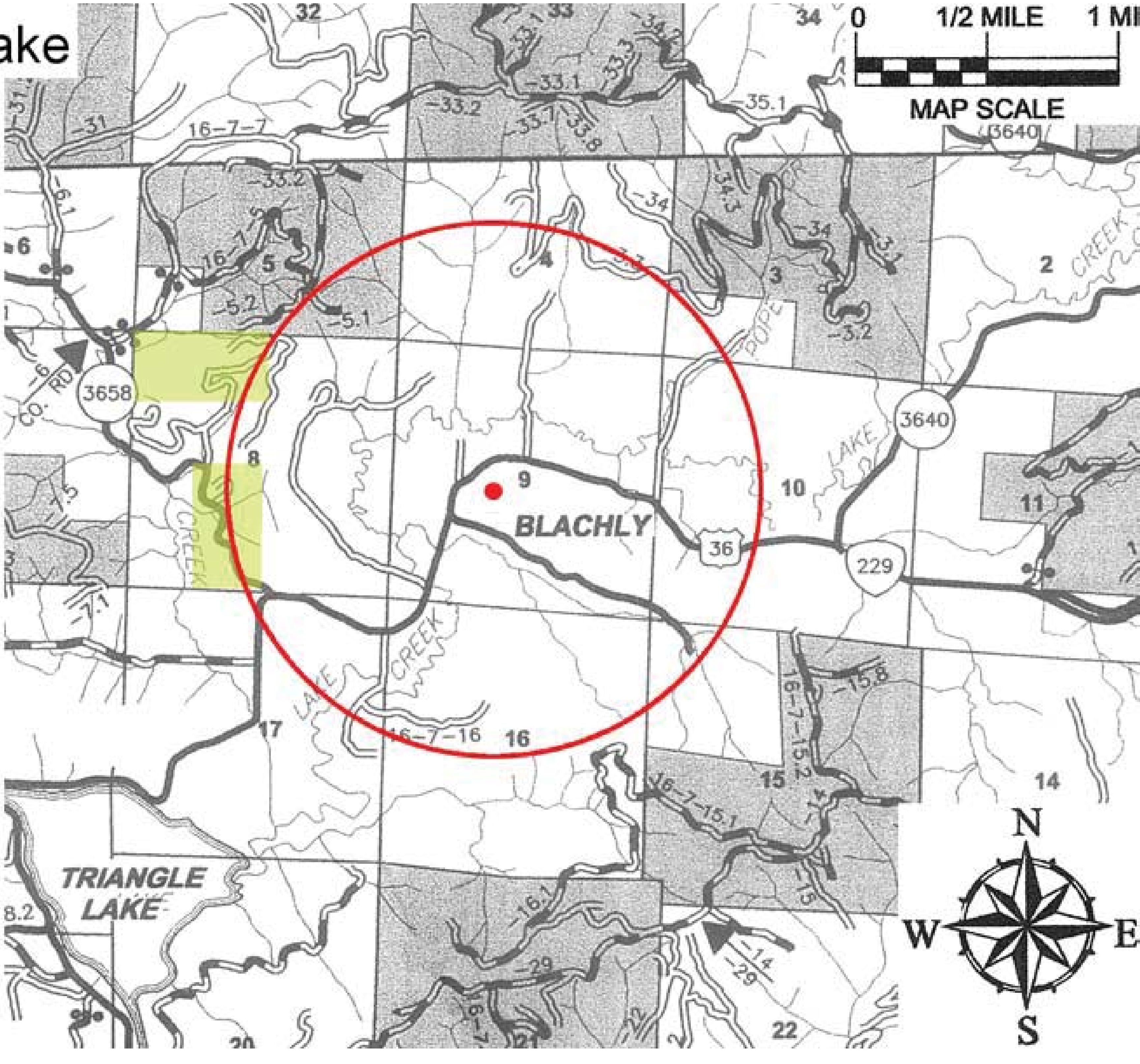
Triangle Lake
School
Legend

1998



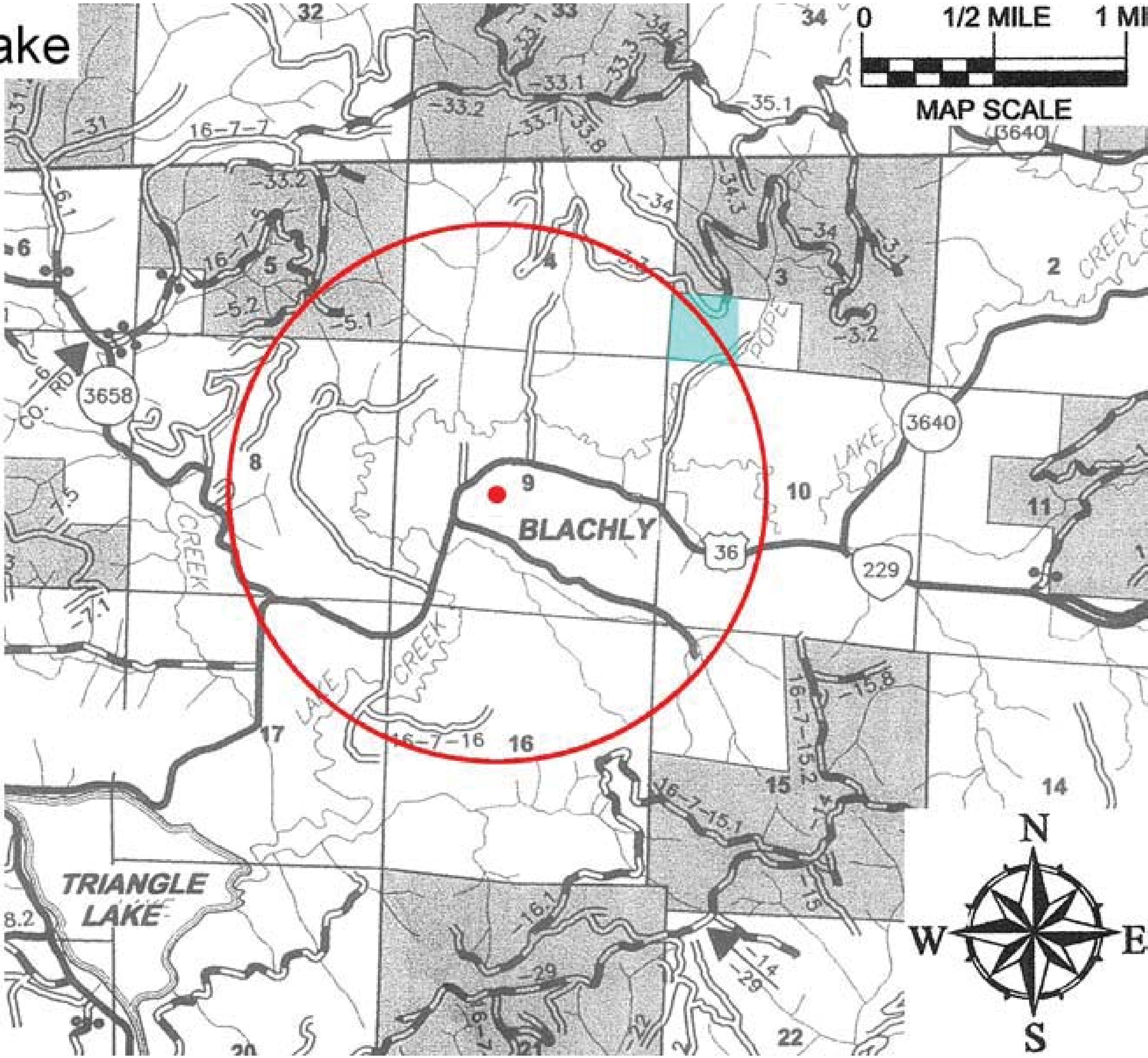
Triangle Lake School Legend

1999



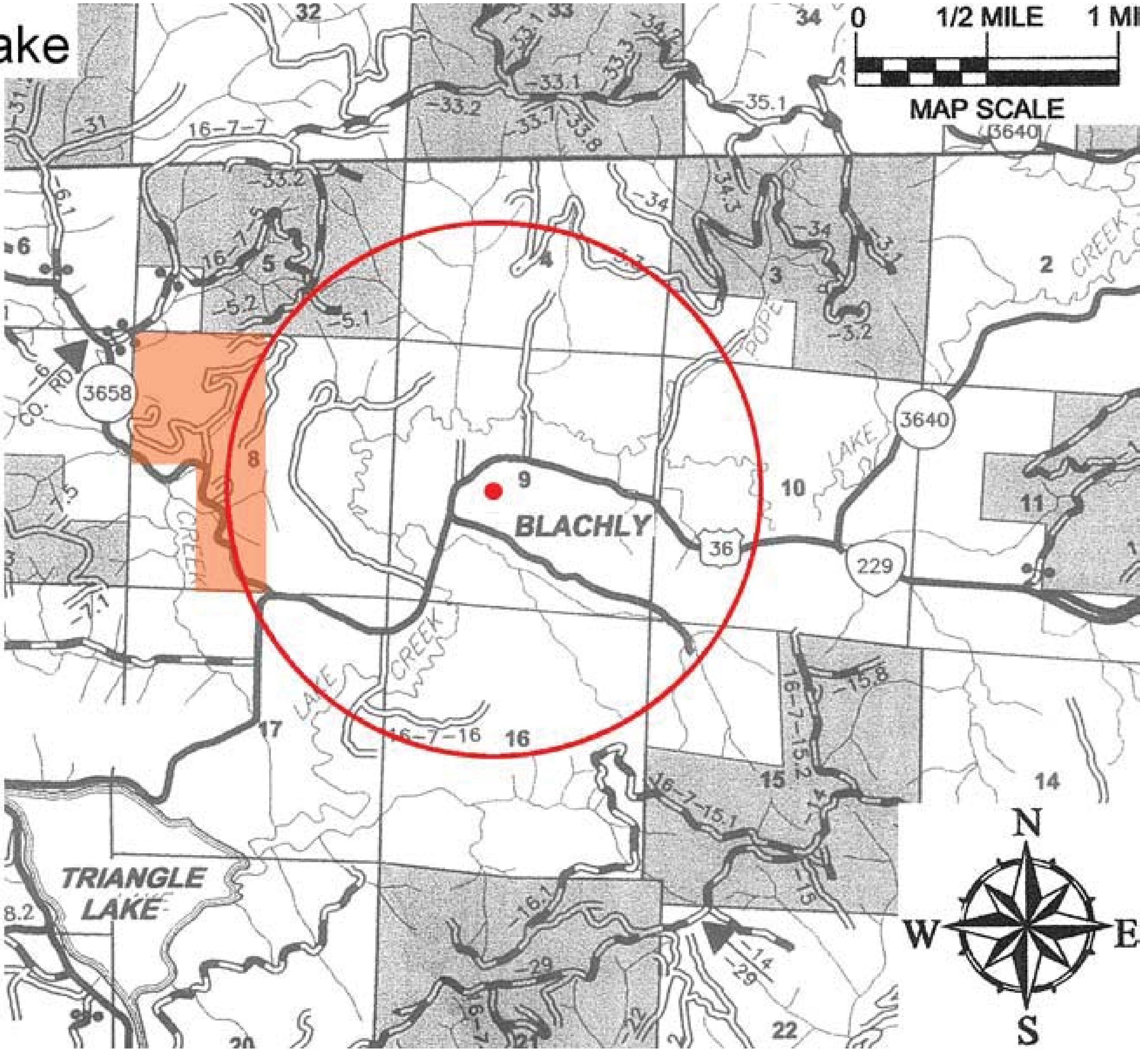
Triangle Lake
School
Legend

2001

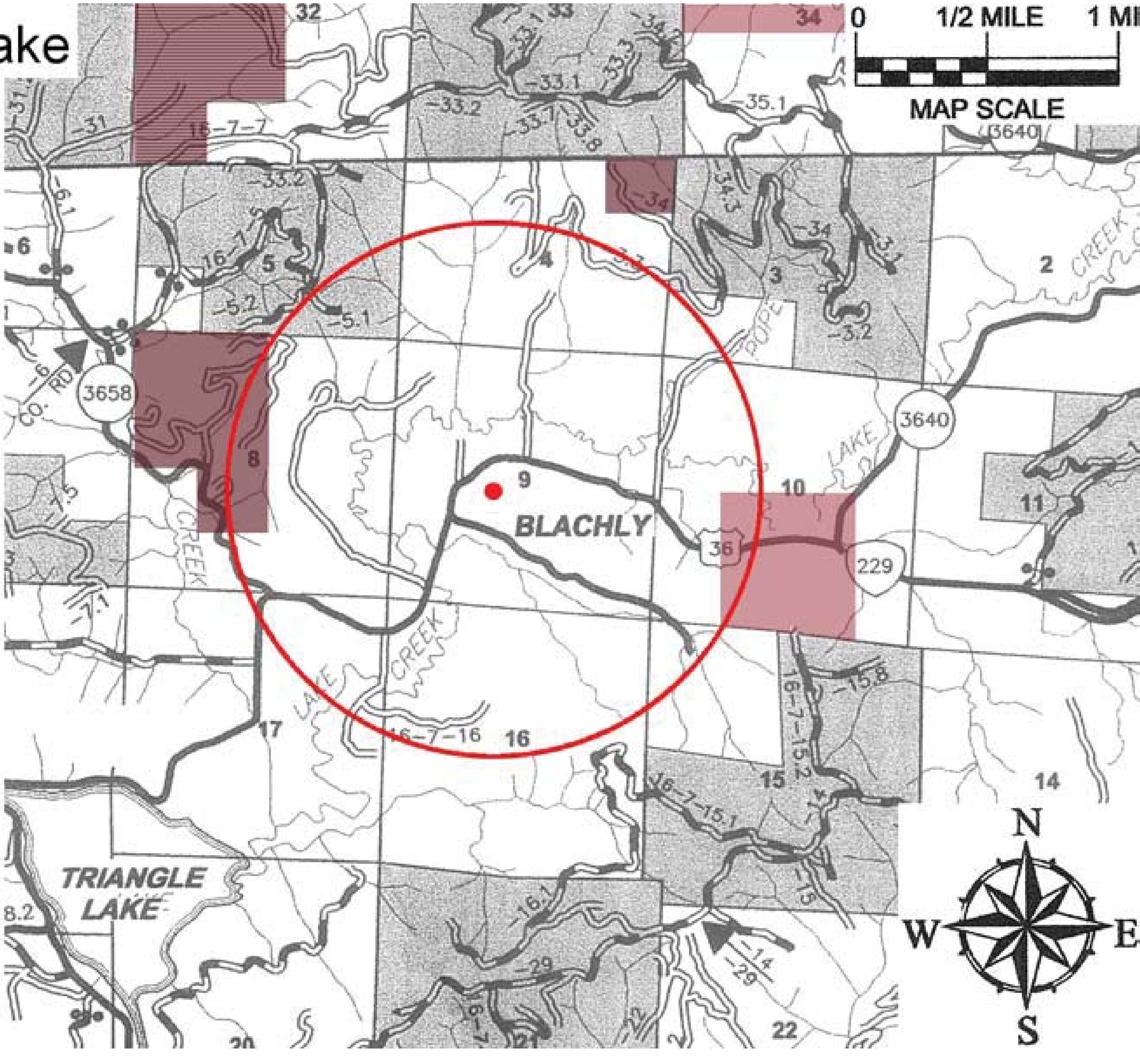


Triangle Lake School Legend

2002

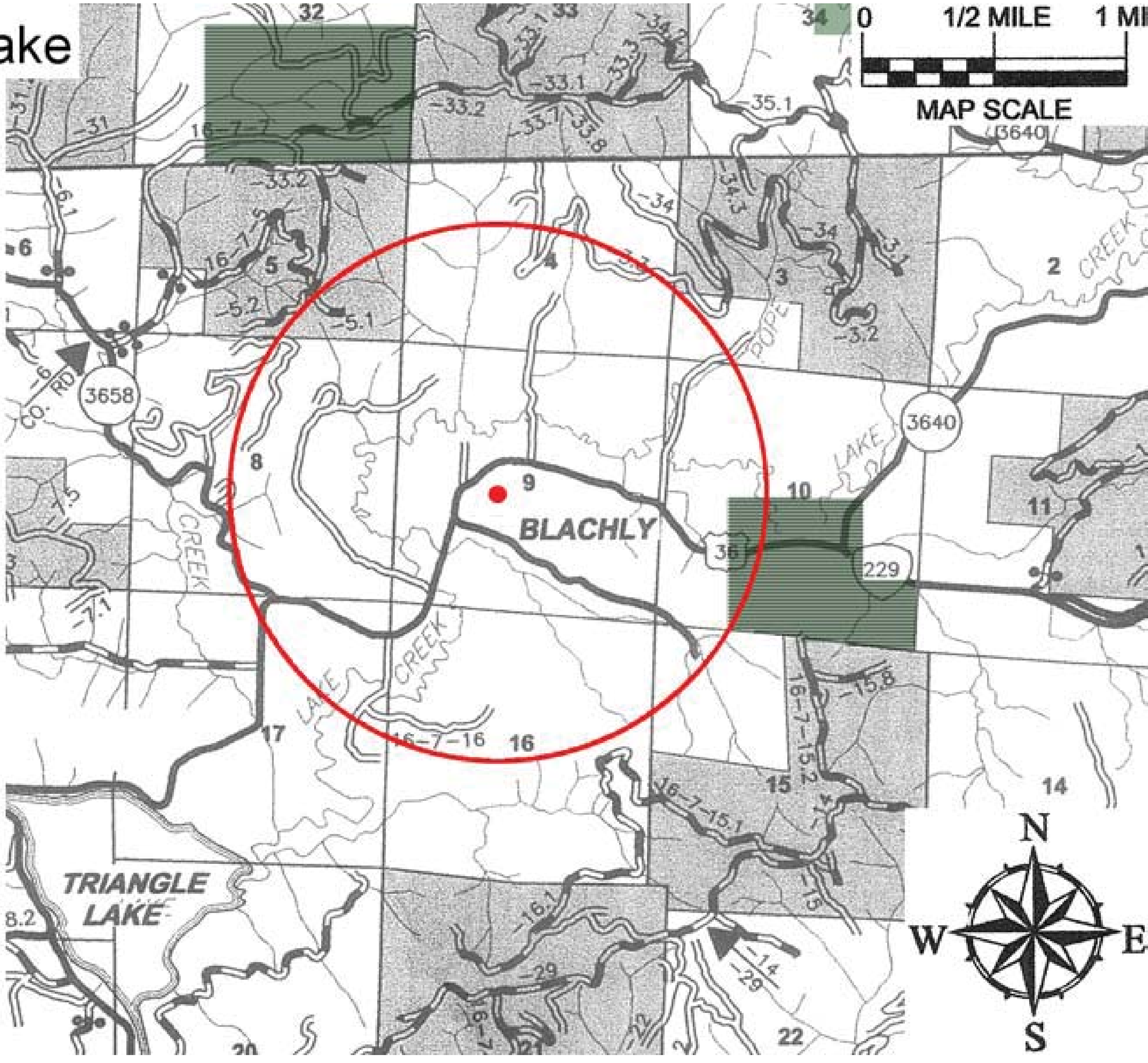


Triangle Lake School Legend



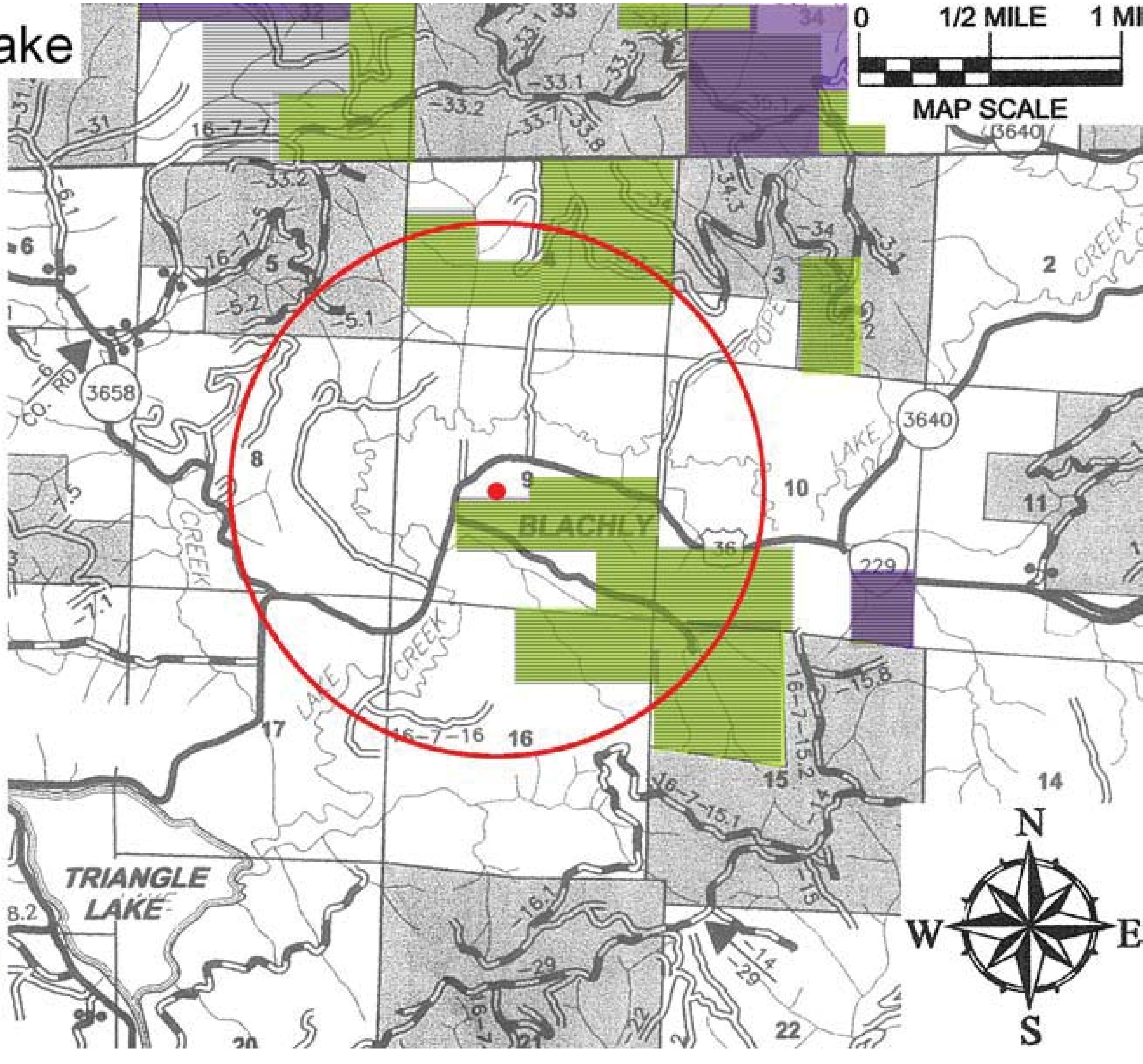
2003

Triangle Lake
School
Legend

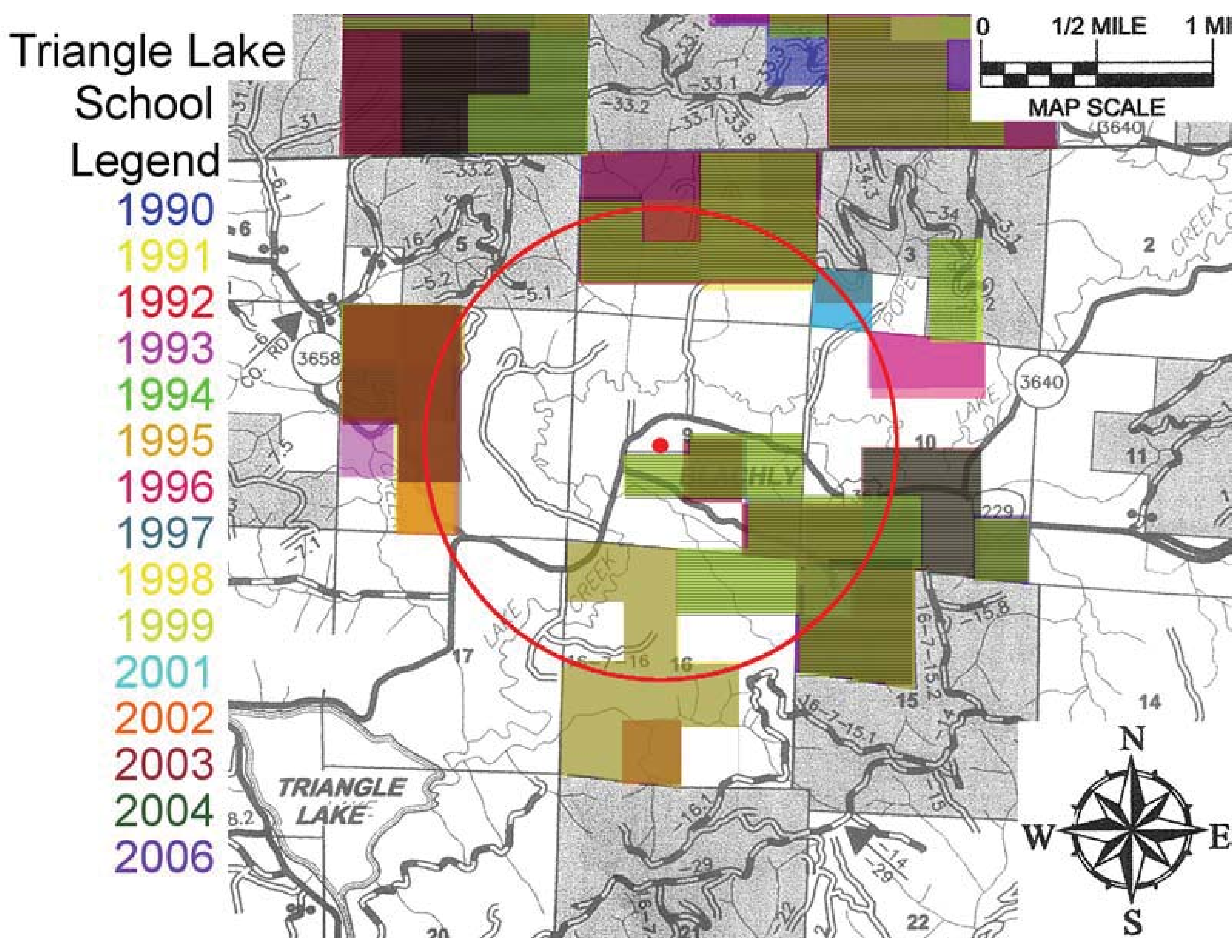


2004

Triangle Lake School Legend



2006


Triangle Lake School Legend
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2001
2002
2003
2004
2006
0
1/2 MILE
1 MI
MAP SCALE
3658
3640
CO. RD.
TRIANGLE LAKE
POPE
LAKE CREEK
CREEK
N
S
E
W

Oregon Toxics Alliance
P.O. Box 1106
Eugene OR 97440
Phone/Fax 465-8860
http://www.oregontoxics.org/

Forestland Dwellers
P.O. Box 5954
Eugene, OR 97405
Phone 342-8332
http://www.forestlanddwellers.org/

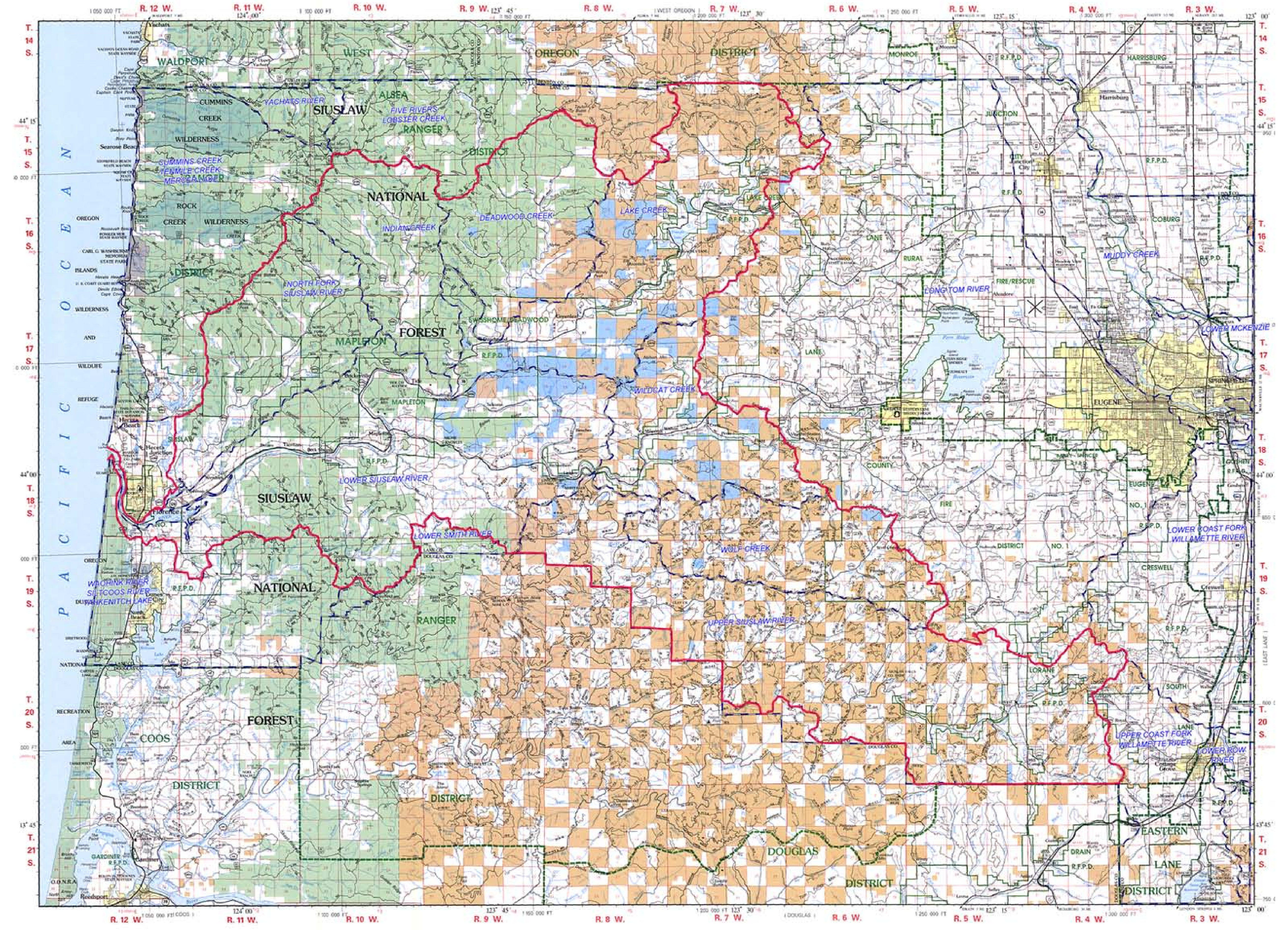
Siuslaw Watershed Forestry Spray Mapping Project 1990 – 2006

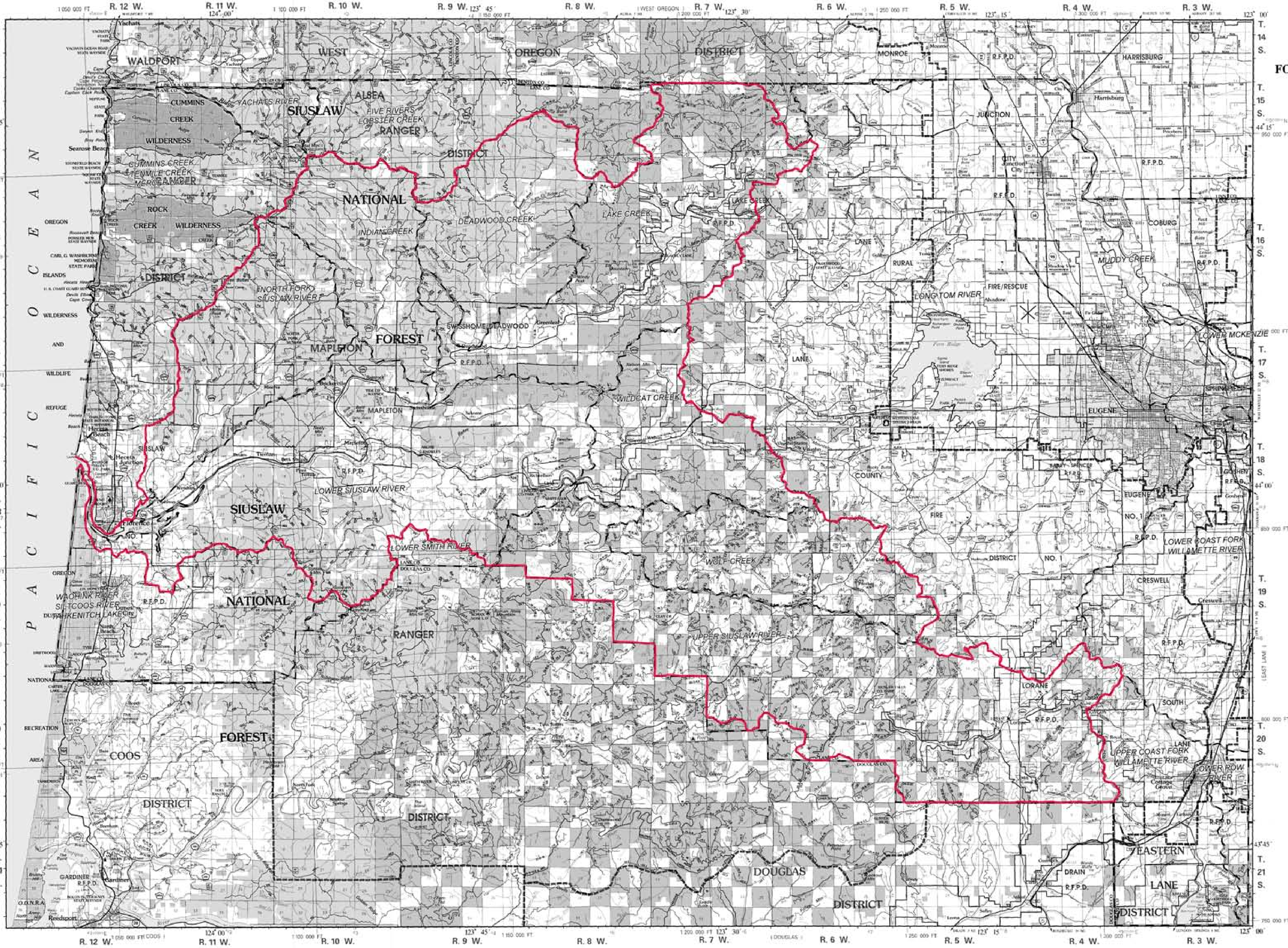
Siuslaw Watershed

Presentation ~ Mapleton, Oregon

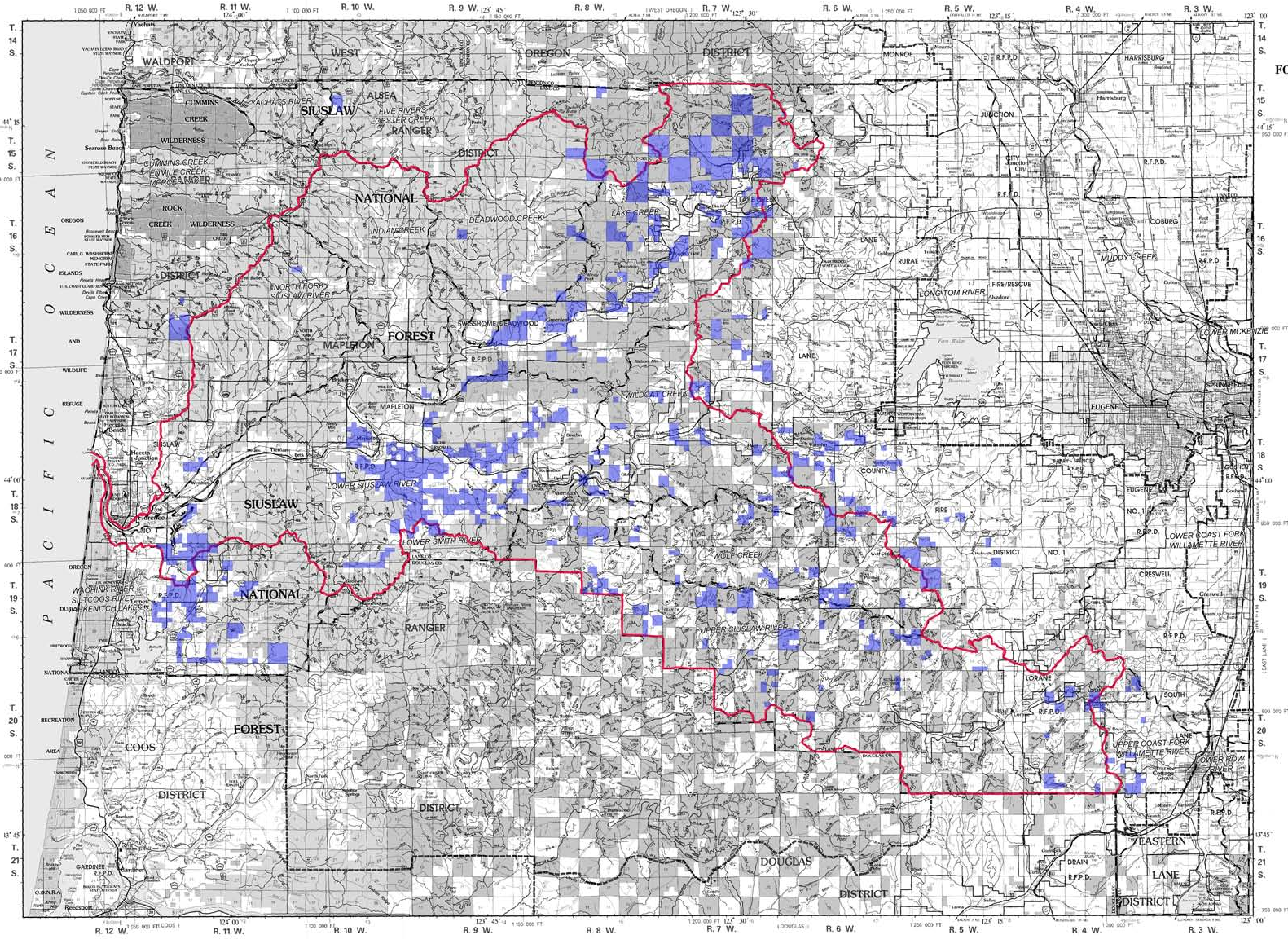
Presented by
Jan Wroncy and Gary Hale

Mapped using data from Oregon Department of Forestry Forest Activity Computerized Tracking System (FACTS)

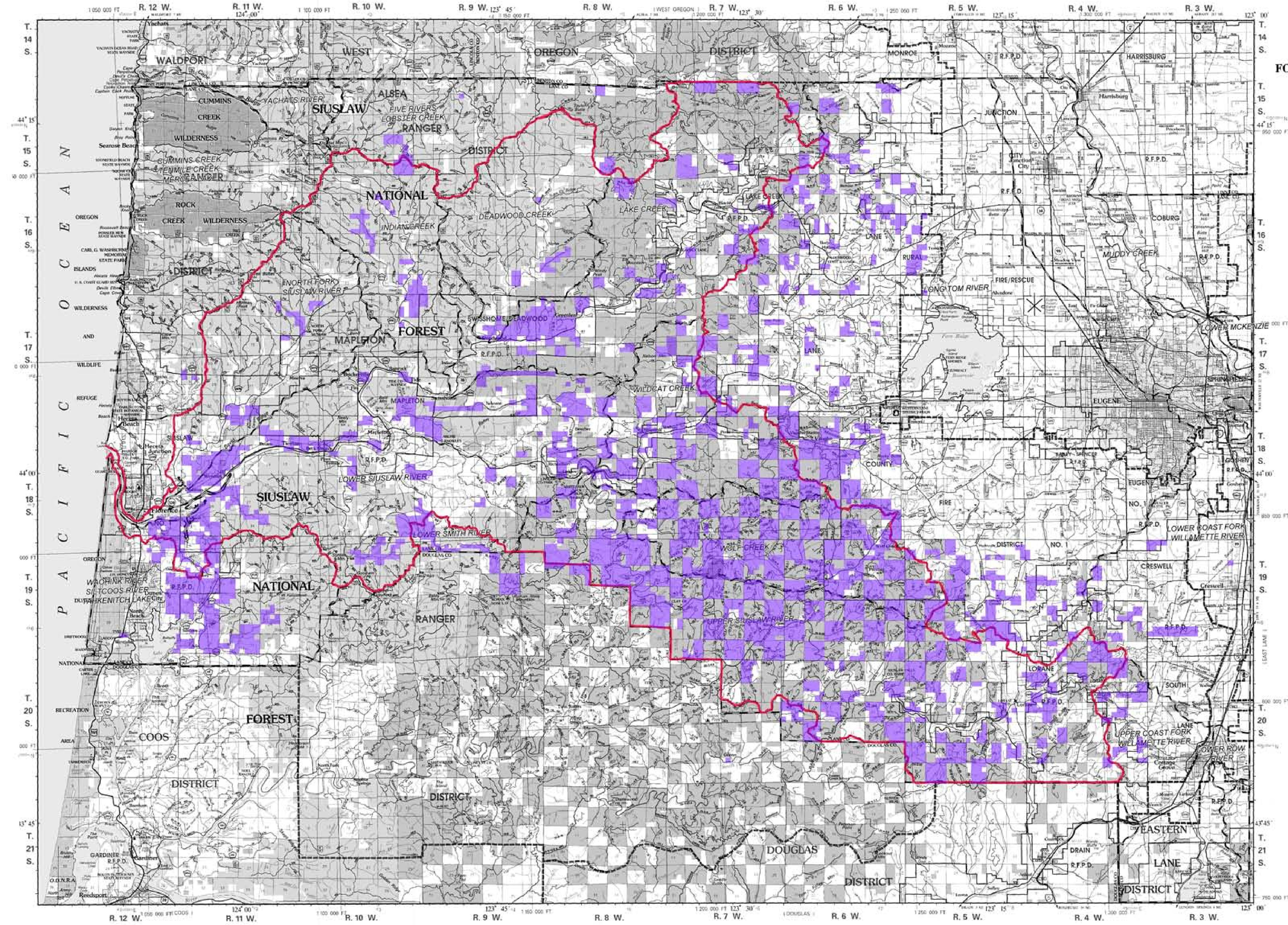

R. 12 W.
R. 11 W.
R. 10 W.
R. 9 W.
R. 8 W.
R. 7 W.
R. 6 W.
R. 5 W.
R. 4 W.
R. 3 W.
T. 14 S.
T. 15 S.
T. 16 S.
T. 17 S.
T. 18 S.
T. 19 S.
T. 20 S.
T. 21 S.
123° 45'
123° 30'
123° 15'
124° 00'
123° 00'
44° 15'
44° 00'
43° 45'
PACIFIC OCEAN
WALDPORT
CUMMINS CREEK WILDERNESS
YACHATS RIVER
SIUSLAW
ALSEA
FIVE RIVERS LOBSTER CREEK
RANGER
DISTRICT
WEST
OREGON
DISTRICT
MONROE
HARRISBURG
Harrisburg
JUNCTION
Junction City
R.F.P.D.
CUMMINS CREEK TENMILE CREEK MERCER LAKE
NATIONAL
ROCK CREEK WILDERNESS
DEADWOOD CREEK
INDIAN CREEK
LAKE CREEK
COBURG
MUDDY CREEK
NORTH FORK SIUSLAW RIVER
DISTRICT
LONG TOM RIVER
FIRE/RESCUE
RURAL
LANE
SWISSHOME-DEADWOOD
FOREST
MAPLETON
LOWER MCKENZIE
WILDCAT CREEK
SPRINGFIELD
EUGENE
Eugene
LOWER SIUSLAW RIVER
COUNTY
SIUSLAW
Florence
LOWER SMITH RIVER
FIRE
DISTRICT
NO. 1
LOWER COAST FORK WILLAMETTE RIVER
WOLF CREEK
CRESWELL
Creswell
WAHOHINK RIVER
SILTCOOS RIVER
TAHKENITCH LAKE
NATIONAL
RANGER
UPPER SIUSLAW RIVER
LORANE
SOUTH
FOREST
COOS
AREA
RECREATION
DISTRICT
UPPER COAST FORK WILLAMETTE RIVER
LOWER ROW RIVER
LANE
DISTRICT
DOUGLAS
DISTRICT
EASTERN
LANE
DISTRICT
DRAIN
GARDINER R.F.P.D.
O.N.R.A.
Reedsport
WILDERNESS
AND
WILDLIFE
REFUGE
ISLANDS
Heceta Beach
Searose Beach
Yachats



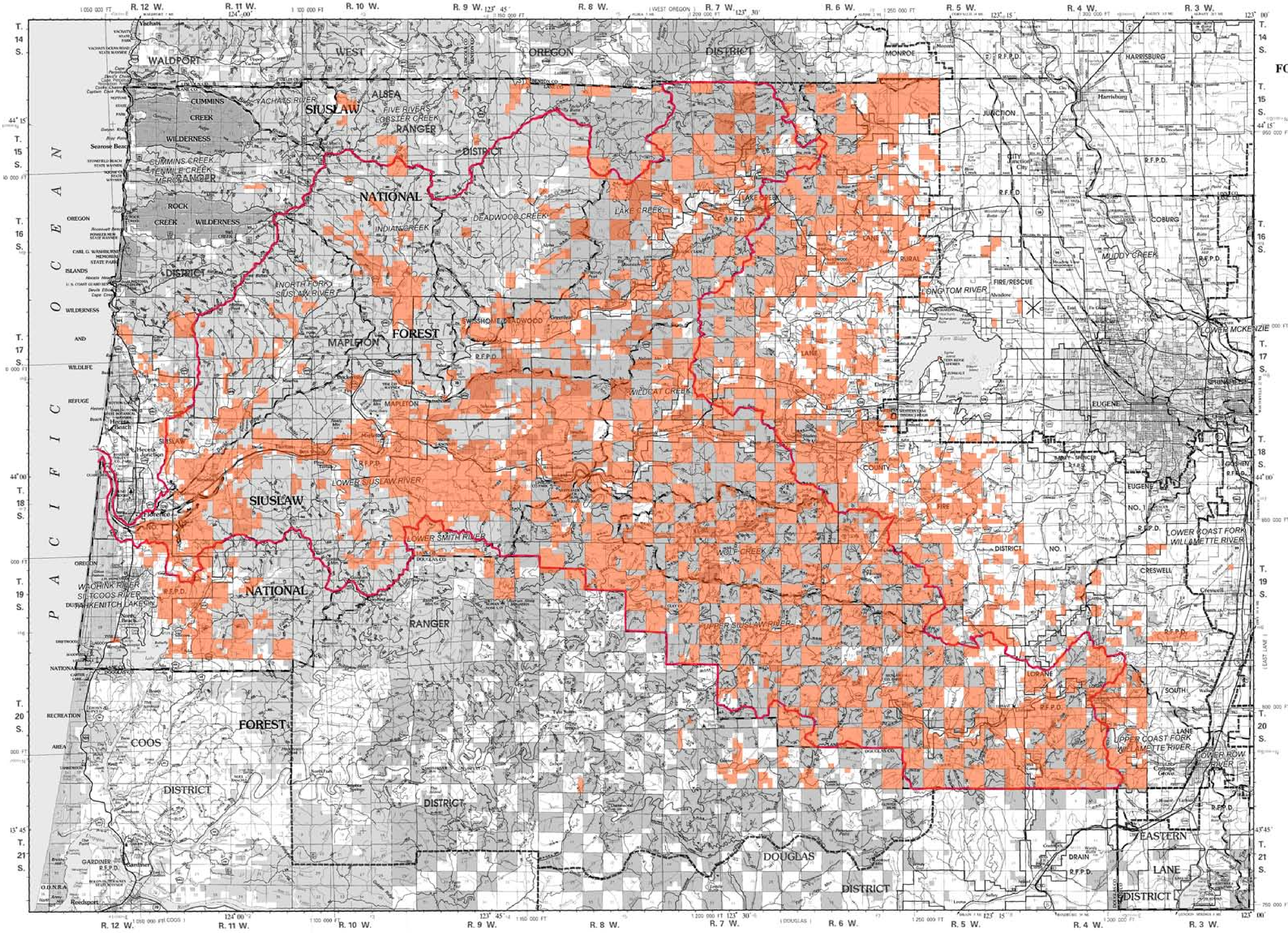
R. 12 W.
R. 11 W.
R. 10 W.
R. 9 W.
R. 8 W.
R. 7 W.
R. 6 W.
R. 5 W.
R. 4 W.
R. 3 W.
T. 14 S.
T. 15 S.
T. 16 S.
T. 17 S.
T. 18 S.
T. 19 S.
T. 20 S.
T. 21 S.
44° 15'
44° 00'
43° 45'
124° 00'
123° 45'
123° 30'
123° 15'
123° 00'
PACIFIC OCEAN
WEST OREGON DISTRICT
WALDPORT
SIUSLAW NATIONAL FOREST
ALSEA RANGER DISTRICT
MAPLETON RANGER DISTRICT
CUMMINS CREEK WILDERNESS
ROCK CREEK WILDERNESS
OREGON ISLANDS WILDERNESS AND WILDLIFE REFUGE
OREGON DUNES NATIONAL RECREATION AREA
YACHATS RIVER
FIVE RIVERS
LOBSTER CREEK
CUMMINS CREEK
TENMILE CREEK
MERCER LAKE
DEADWOOD CREEK
INDIAN CREEK
LAKE CREEK
NORTH FORK SIUSLAW RIVER
SWISSHOME DEADWOOD R.F.P.D.
WILDCAT CREEK
LOWER SIUSLAW RIVER
LOWER SMITH RIVER
WOLF CREEK
UPPER SIUSLAW RIVER
WAOHINK RIVER
SILTCOOS RIVER
TAHKENITCH LAKE
SIUSLAW R.F.P.D.
NO. 1
MAPLETON
Florence
Yachats
Searose Beach
Heceta Beach
MONROE
HARRISBURG
JUNCTION CITY R.F.P.D.
LANE RURAL FIRE/RESCUE
LONG TOM RIVER
MUDDY CREEK
COBURG R.F.P.D.
LANE COUNTY
LOWER MCKENZIE
EUGENE
SPRINGFIELD
EUGENE R.F.P.D. NO. 1
LOWER COAST FORK WILLAMETTE RIVER
CRESWELL
FIRE DISTRICT NO. 1
LORANE R.F.P.D.
SOUTH LANE
UPPER COAST FORK WILLAMETTE RIVER
LOWER ROW RIVER
COOS DISTRICT
DOUGLAS DISTRICT
DRAIN R.F.P.D.
EASTERN LANE DISTRICT
GARDINER R.F.P.D.
Reedsport
LANE CO.
DOUGLAS CO.
BENTON CO.



R. 12 W.
R. 11 W.
R. 10 W.
R. 9 W.
R. 8 W.
R. 7 W.
R. 6 W.
R. 5 W.
R. 4 W.
R. 3 W.
T. 14 S.
T. 15 S.
T. 16 S.
T. 17 S.
T. 18 S.
T. 19 S.
T. 20 S.
T. 21 S.
PACIFIC OCEAN
WALDPORT
WEST OREGON DISTRICT
SIUSLAW NATIONAL FOREST
ALSEA RANGER DISTRICT
MAPLETON RANGER DISTRICT
CUMMINS CREEK WILDERNESS
ROCK CREEK WILDERNESS
YACHATS RIVER
FIVE RIVERS
LOBSTER CREEK
CUMMINS CREEK
TENMILE CREEK
MONROE
HARRISBURG
JUNCTION CITY
COBURG
DEADWOOD CREEK
INDIAN CREEK
LAKE CREEK
NORTH FORK SIUSLAW RIVER
SWISSHOME/DEADWOOD
MAPLETON
WILDCAT CREEK
LANE COUNTY RURAL FIRE/RESCUE
LONG TOM RIVER
MUDDY CREEK
LOWER MCKENZIE
SPRINGFIELD
EUGENE
LOWER SIUSLAW RIVER
LOWER SMITH RIVER
WOLF CREEK
UPPER SIUSLAW RIVER
EUGENE R.F.P.D. NO. 1
LOWER COAST FORK WILLAMETTE RIVER
CRESWELL
LORANE
UPPER COAST FORK WILLAMETTE RIVER
COTTAGE GROVE
LOWER ROW RIVER
SOUTH LANE
Florence
WAHINK RIVER
SILTCOOS RIVER
TAHKENITCH LAKE
OREGON DUNES NATIONAL RECREATION AREA
COOS FOREST DISTRICT
DOUGLAS DISTRICT
EASTERN LANE DISTRICT
DRAIN R.F.P.D.
GARDINER R.F.P.D.
Reedsport

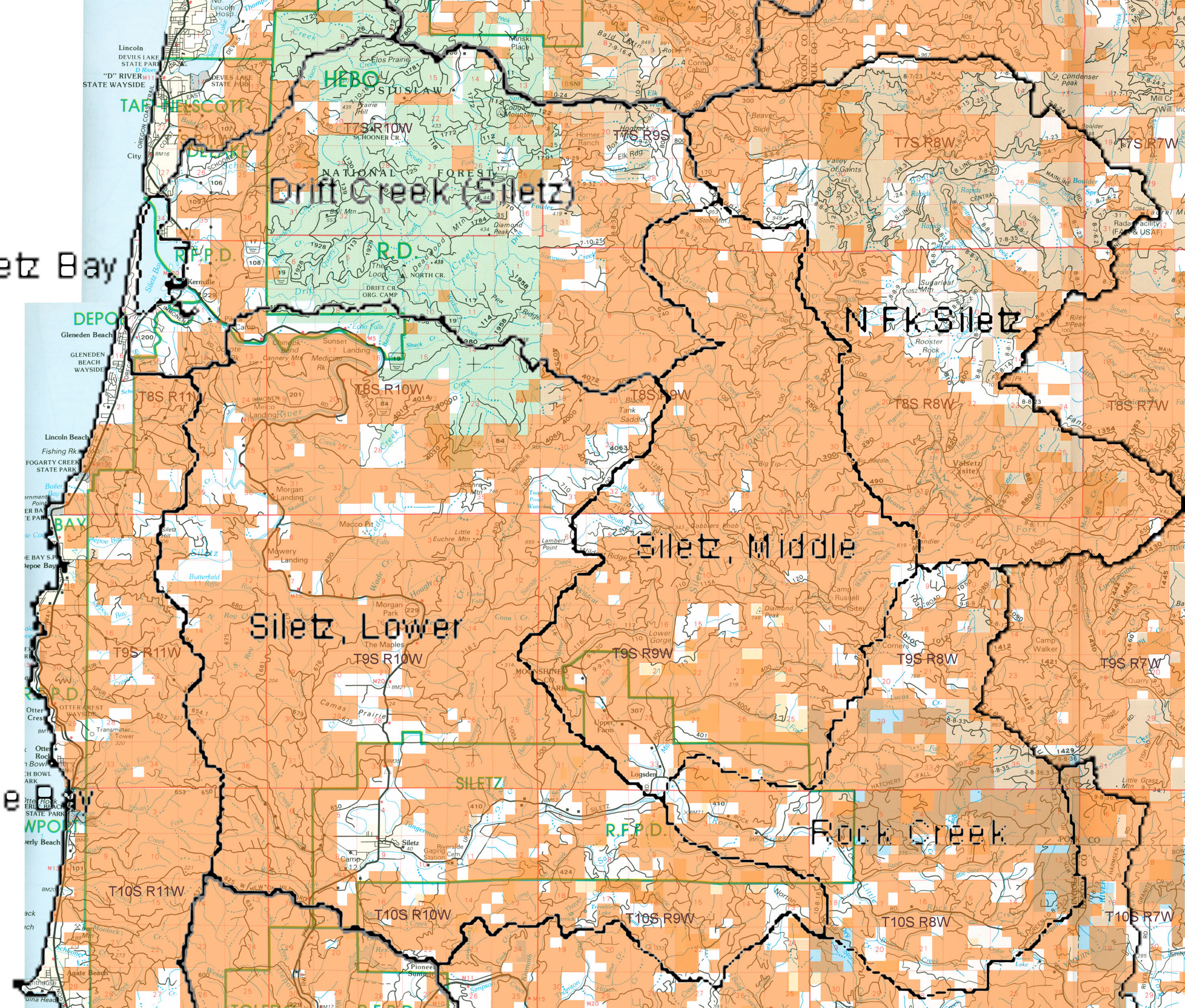


R. 12 W.
R. 11 W.
R. 10 W.
R. 9 W.
R. 8 W.
R. 7 W.
R. 6 W.
R. 5 W.
R. 4 W.
R. 3 W.
T. 14 S.
T. 15 S.
T. 16 S.
T. 17 S.
T. 18 S.
T. 19 S.
T. 20 S.
T. 21 S.
WEST
OREGON
DISTRICT
WALDPORT
MONROE
HARRISBURG
CUMMINS
CREEK
WILDERNESS
SIUSLAW
ALSEA
YACHATS RIVER
FIVE RIVERS
LOBSTER CREEK
RANGER
DISTRICT
JUNCTION
CITY
R.F.P.D.
CUMMINS CREEK
TENMILE CREEK
MERCER LAKE
ROCK
CREEK
WILDERNESS
NATIONAL
INDIAN CREEK
DEADWOOD CREEK
LAKE CREEK
LANE
RURAL
COBURG
MUDDY CREEK
OREGON
ISLANDS
WILDERNESS
AND
WILDLIFE
REFUGE
DISTRICT
NORTH FORK SIUSLAW RIVER
FIRE/RESCUE
LONG TOM RIVER
SWISSHOME-DEADWOOD
FOREST
MAPLETON
LOWER MCKENZIE
SPRINGFIELD
WILDCAT CREEK
EUGENE
SIUSLAW
LOWER SIUSLAW RIVER
COUNTY
EUGENE
FIRE
NO. 1
SIUSLAW
LOWER SMITH RIVER
WOLF CREEK
DISTRICT
NO. 1
LOWER COAST FORK WILLAMETTE RIVER
CRESWELL
OREGON
NATIONAL
WOAHINK RIVER
SILTCOOS RIVER
TAHKENITCH LAKE
RANGER
UPPER SIUSLAW RIVER
LORANE
SOUTH
LANE
NATIONAL
RECREATION
AREA
FOREST
COOS
UPPER COAST FORK WILLAMETTE RIVER
LOWER ROW RIVER
DISTRICT
DISTRICT
EASTERN
DOUGLAS
DRAIN
R.F.P.D.
LANE
DISTRICT
GARDINER
R.F.P.D.
Reedsport
PACIFIC OCEAN
44° 15'
44° 00'
43° 45'
123° 45'
123° 30'
123° 15'
123° 00'
124° 00'



R. 12 W.
R. 11 W.
R. 10 W.
R. 9 W.
R. 8 W.
R. 7 W.
R. 6 W.
R. 5 W.
R. 4 W.
R. 3 W.
T. 14 S.
T. 15 S.
T. 16 S.
T. 17 S.
T. 18 S.
T. 19 S.
T. 20 S.
T. 21 S.
PACIFIC OCEAN
WALDPORT
WEST OREGON DISTRICT
MONROE
HARRISBURG
CUMMINS CREEK WILDERNESS
ROCK CREEK WILDERNESS
SIUSLAW NATIONAL FOREST
ALSEA RANGER DISTRICT
MAPLETON RANGER DISTRICT
OREGON ISLANDS WILDERNESS AND WILDLIFE REFUGE
OREGON NATIONAL RECREATION AREA
COOS DISTRICT
DOUGLAS DISTRICT
EASTERN LANE DISTRICT
JUNCTION CITY R.F.P.D.
LAKE CREEK R.F.P.D.
LANE RURAL FIRE/RESCUE
COBURG R.F.P.D.
MUDDY CREEK
LONG TOM RIVER
LOWER MCKENZIE
EUGENE
SWISSHOME/DEADWOOD R.F.P.D.
MAPLETON R.F.P.D.
SIUSLAW R.F.P.D.
LANE COUNTY FIRE DISTRICT NO. 1
EUGENE R.F.P.D. NO. 1
LOWER COAST FORK WILLAMETTE RIVER
UPPER COAST FORK WILLAMETTE RIVER
LOWER ROW RIVER
CRESWELL
SOUTH LANE
LORANE R.F.P.D.
DRAIN R.F.P.D.
GARDINER R.F.P.D.
LOWER SIUSLAW RIVER
LOWER SMITH RIVER
UPPER SIUSLAW RIVER
WOLF CREEK
WILDCAT CREEK
DEADWOOD CREEK
INDIAN CREEK
LAKE CREEK
NORTH FORK SIUSLAW RIVER
FIVE RIVERS LOBSTER CREEK
YACHATS RIVER
CUMMINS CREEK TENMILE CREEK
WAOHINK RIVER
SILTCOOS RIVER
TAHKENITCH LAKE
Florence
Reedsport


SALMON
SPAWNING
SURVEY


Drift Creek (Siletz)
N Fk Siletz
Siletz, Middle
Siletz, Lower
Rock Creek
etz Bay
HEBO
SIUSLAW
NATIONAL FOREST
R.D.
SILETZ
R.F.P.D.
T7S R10W
T7S R9W
T7S R8W
T7S R7W
T8S R11W
T8S R10W
T8S R9W
T8S R8W
T8S R7W
T9S R11W
T9S R10W
T9S R9W
T9S R8W
T9S R7W
T10S R11W
T10S R10W
T10S R9W
T10S R8W
T10S R7W
Gleneden Beach
Lincoln Beach
Siletz
Logsden
Morgan Park
The Maples
Valley of Giants
Sugarloaf Mtn
Camp Russell
Camp Walker
Diamond Peak


WARNING

State of Oregon
Department of Forestry - Department of Revenue
Notification Number: 2014-781-00194
Timber Sale: WL Spring Aerial North

Attached is the processed information from the Notification of Operation/Application for Permit signed by Jeff Yost representing the Land Owner, and received by Department of Forestry on February 5, 2014. Please review this information and retain for future reference.

Notices and Permits

Notice is given to the State Forester that an operation will be conducted on the lands described herein.

A permit to use fire or operate power driven machinery is issued for the land described herein.

SF Comments:

Notification 15 Day Waiting Period:
This Operation is subject to the 15 day Waiting Period.

Operator:
Jeff Yost
Weyerhaeuser Co.
PO Box 1819
Eugene, OR 97440
(541) 744-4600

Fire Contact:
Jeff Yost
(541) 744-4600

Land Owner:
Jeff Yost
Weyerhaeuser Co.
PO Box 1819
Eugene, OR 97440
(541) 744-4600

Timber Owner:

District: **Western Lane**

Office: **Veneta Unit**

County: **Lane**

(Subscriber Copy)

Doug Decker, State Forester
Link Smith, **District Forester**

Unit Information - Notification: 201478100194
Unit 2 of 3 Start: 03/01/14 End: 12/30/14
Status: Open
Stewardship Forester: Robin L. Biesecker

Site Conditions Waters: Lake or stream Within 100 feet.
Soils: No mass soil movement.
Slope: 36% to 65%.
SF Phone Number: (541)935-2283

Priorities: **Fire: Low** **FPA: High**

			NE				NW				SW				SE				Government	Tax Lot	Reg
Twp	**Rge**	**Sec**	**NE**	**NW**	**SW**	**SE**	**NE**	**NW**	**SW**	**SE**	**NE**	**NW**	**SW**	**SE**	**NE**	**NW**	**SW**	**SE**	**Lot Number**	**No.**	**Use**
15S	06W	18	☐	☐	☐	☐	☐	☐	☐	☐	☐	☐	☐	☐	☑	☑	☐	☐			WT-1

Activity	Method	Acres	Feet	MBF	Comment
4a - Herbicide Application	Aerial operation / applications; Ground	54.00	0	0	Pope & Talbot; Brand: Atrazine 4L, Sulfomet, Oust, Velpar DF, 2,4-D LV6, Transline, Accord Concentrate, Rodeo; Additives: Foam Buster, Grounded

Resource Name	Resource Description
Tribs. to South Fork Alsea River	Tribs. to South Fork Alsea River: Small Type F
Tribs. to South Fork Alsea River	Small Non-Fish Stream(s), Small Unknown

Subscribers: Water Rights-Horton Road Organics, Water Rights-Jan Wroncy

Weyerhaeuser
The future is growing™

South Valley Timberlands
P.O. Box 1819
Eugene, OR 97440-1819
541-744-4600 (phone)
541-744-4688 (fax)

WRITTEN PLAN OF OPERATION FOR AERIAL/GROUND HERBICIDE APPLICATION

Weyerhaeuser Company is planning aerial and/or ground spray applications targeting broadleaf and/or herbaceous vegetation near protected resources. This letter is our submission of a written plan of operation for activities within 100 feet of protected resources. The attached plat shows the unit boundaries and resources to be protected. Chemical will not be mixed, handled or staged with 100 feet of protected resources.

No herbicide will be directly applied within 60 feet (aerial) or 10 feet (ground) of the high water mark of any protected resource defined as F or D stream, lakes, significant wetlands and other areas of standing open water greater than one-quarter acre at the time of application. No treatment of timbered buffers left during harvest within 100 feet of protected resources will be done. All application will be done in compliance with the Oregon Forest Practices Rules and label instructions. The operator will have a current map and/or aerial photo showing locations of the unit boundaries and resources requiring protection. Application of herbicide within the 100 ft adjacent to the protected resources will be flown parallel to the stream (aerial).

Unit # 1313 (ODF #2)
Legal Description: 18 - 15S - 6W
Stream & Classification SF TRIB TO ALSEA

Sincerely,

Jeff Yost
South Valley Area

2-4-14

Operator (s) ____________ Date ____________

Stewardship Forester ____________ Date ____________

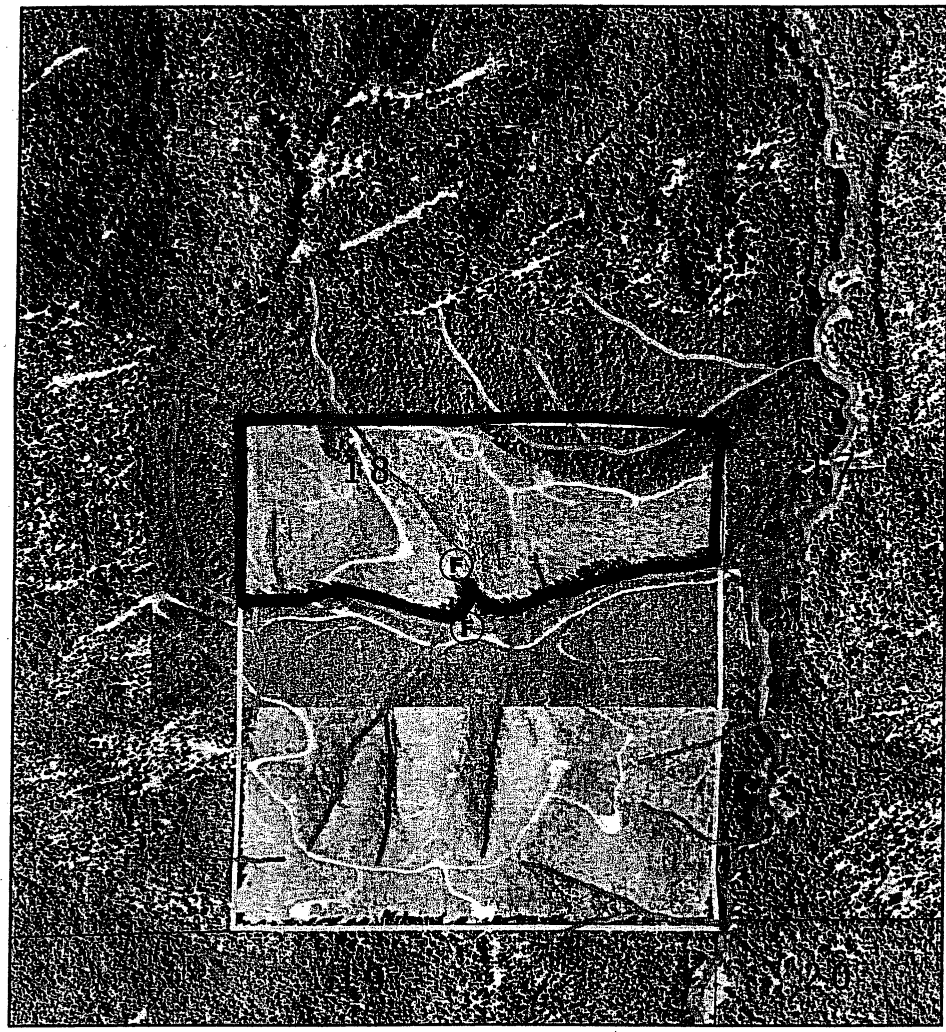
WLN-2

Weyerhaeuser Company
South Valley Western Timberlands
P.O. Box 1819, Eugene, OR 97440
(541) 744-4600 (541) 746-2511 [24 hrs.]

Area/Road No. POPE + TALBOT

Aerial Herbicide Application Report

District: South Valley **Season** SPRING 14



Sec(s)	18		
Twp(s)	15S		
Rge(s)	6W		

Elevation:________ Scale 1" = 1,000'

Unit(s)	Date(s)	Time of Day (Begin/End)
		/
		/
		/

Type of Equipment: HELICOPTER
Application Method: BROADCAST
Contractor/Co.:
Full name of applicator/certification #:
____
FAA No.:____________
Boom Type:____________
Nozzle Size:
Pressure:

Date	Time	Temp °F/°F	Wind Dir/Spd	RH%

Reforestation Unit #:____________
Chemical Supplier: __Helena __Wilbur-Ellis
___Containers rinsed at least 3x & recycled
Comments:

Date By

Note: This record shall be kept for 3 years in accordance with ORS 634.146.

Unit	Acres	Chemical/Ac (Trade Name) Chemical		Surfactant-Carrier/Ac			Water G/Acre	Total G/Acre	Site Prep	Release	Target Species
1313	54									X	G+F
Actual											

Written plan required? Yes ☒ No ☐

Resource to be protected: SF TRIBS TO ALSEA
Submitted to:____________ Date:________
Operator Signed:____________ Date________
Date State Notification Submitted:____________
Date Application can begin:

Additional Requirements
☐ Neighbor Notification:____________
☐ Wetland Protection:____________
☐ Open Water/Rainfall:____________
☐ T/E Protection:____________
☐ Other:____________

Notes: 15-day waiting period cannot be waived for aerial spray. Full name of all applicators and trainees applying pesticide must be recorded. All information must be recorded within 30 days following the pesticide application.

Senator Brad Avakian, Chair
Senate Committee on Environment and Natural Resources
Oregon State Capitol
900 Court Street NE, Room 333
Salem, Oregon 97310

April 10, 2007

RE: Senate Bill 20 (School Buffer Bill), restricts spraying of pesticides near school property and roads servicing school property.

Dear Senator Avakian and Members of the Committee:

I am presenting testimony in support of Senate Bill 20, the School Buffer Bill, which restricts spraying of pesticides near school property and roads servicing school property in order to protect our most vulnerable citizens - our children.

A colleague recently sent me an article in which contained three of the most meaningful statements pertinent to Senate Bill 20:

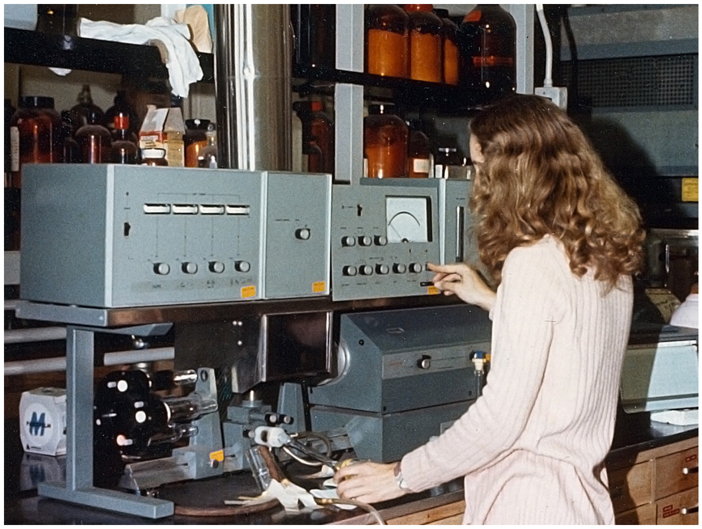
- **"Children are our most important national resource".**
- **"Children's brains, development, and behavior are important to their health, to their ability to contribute to society throughout life, and to the well-being of future generations."**
- **"Children cannot make choices about their environment; it is up to adults to make the right decisions to ensure that they are protected".**

The authors Lynn R. Goldman and Sudha Koduru from John Hopkins University in Baltimore Maryland deserve our thanks for helping us focus on what is important and most precious to us. The article Chemicals in the Environment and DevelopmentalToxicity to Children: A Public Health and Policy Perspective was published in Environmental Health Perspectives, Vol. 108, Supplemental 3, in June 2000.

It may be found at the website below:

http://www.pubmedcentral.nih.gov/picrender.fcgi?artid=1637825&blobtype=pdf

For many years I was involved in diverse fields of scientific research. My first research was in Air Pollution inquiries with Dr. T. J. Chow at Scripps Institute of Oceanography and Dr. Clair Patterson showing that the lead in the environment came from the lead additive in gasoline, which ultimately resulted in the ban on leaded gasoline. I moved to Oregon to set up the lab at the University of Oregon for Dr. Gordon Goles in preparation for analysis of the lunar samples.



Following that, I worked with a team of scientists conducting research on Nitrogen Cycling in the Canopy of Old-Growth Douglas Fir at the H. J. Andrews Experimental Forest in Blue River Oregon. I assisted with analysis of samples in the lab and also participated in some field work.

In my many scientific pursuits I gained an appreciation for the delicate balance between humankind and the environment. Because humankind has the capabilty of destroying the environment, we also have the enormous responsibility of making sure we DO NOT destroy the environment!

For the nearly 30 years I have engaged in organic/no spray farming, and forestry. My experience in forestry research combined with my experience with organic farming and forestry convinces me that **man-made pesticides are not necessary for either farming or forestry.**

I have farmed organically in the Willamette Valley in Coburg, Junction City, and Elmira, and in the Coast Range in several locations.

All our farms have been maintained organically and without pesticides. The riparian forest my husband and I own is managed without chemicals. We grow vegetables, orchard fruit, cane berries, strawberries, blueberries, grapes, pasture, sheep for wool, and timber.

All food and fiber crops can be grown successfully without use of pesticides. Oregon has one of the highest numbers of organic farms in the nation, and a significant number of non-chemically managed timberlands/woodlots as well.

Although the farmers and foresters who oppose the School Buffer will claim they will go out of business, it is simply not true. Restricting pesticide use will not force them out of the business of agriculture (growing crops) or forestry (growing trees).

Present day agriculture has been hijacked by the chemical companies and turned into a "chemiculture".

For the sake of our children, Senate Bill 20 requires farmers and foresters to restrict pesticide use within a buffer zone around schools.

The "crop" we really need to concentrate on growing is not chemically produced timber or chemically produced food. The crop we need to grow is our children - for our children truly are our most important resource and our only future.

It is up to adults to make the right decision.

I urge you to protect our children from exposures to pesticides by supporting and voting for Senate Bill 20 providing a Pesticide-Free Buffer around the Schools.

Sincerely

[redacted]