6-1996

MR399: Forestry-Related Nonpoint Source Pollution in Maine: A Literature Review

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CFRU Information Report 38

Forestry-Related Nonpoint Source Pollution in Maine: A Literature Review

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Miscellaneous Report 399

June 1996

Acknowledgements

The authors wish to acknowledge the contributions of Dr. John Moring, who provided advice at the outset of this project and reviewed an earlier version of this paper. We also thank Mr. John Soules, Maine DEP, who provided advice as well as access to several agency papers and reports. This project was funded by Maine DEP through the US Environmental Protection Agency under Section 104b(3) of the Federal Clean Water Act. Financial support was also provided by the Cooperative Forestry Research Unit, which donated the time of R. Briggs, and by the University of Maine, which contributed indirect costs to the project.

Introduction

Maine is rich in renewable natural resources, which provide the basis for a vigorous forest industry. The 16.8 million acres of commercial forest land (Seymour and Lemin 1989) cover more than 90% of the state. Maine is the most forested state in the Northeast. Forests occupy 78% of Maine's surface (Maine Department of Environmental Protection 1994a). Forestry, supported by this resource base, is the dominant component of the state economy. Maine's forestry sector generated a total value of $5.076 billion in 1990 (Field 1995).

Maine has abundant, high-quality water resources, due to the combination of location in the Northeast and the preponderance of forested watersheds. Precipitation in the Northeast exceeds evapotranspiration and is evenly distributed throughout the year, generates numerous streams, ponds, lakes, and wetlands that provide high-quality water, aquatic habitat, and recreational opportunities. Water quality and forest productivity both depend directly on the physical integrity of the watersheds, which are inextricably linked to the soil.

Forest management operations associated with timber harvesting have the potential to disturb soil, leading to erosion and sedimentation. The consequences can be both acute (decreased water quality) and chronic (reduced long-term site productivity). Erosion and sedimentation control, a primary focus of watershed management (Satterlund and Adams 1992), can be used effectively to minimize soil disturbance. The set of practices that have been designed to minimize or eliminate soil displacement and transport are collectively referred to as best management practices (BMPs).

Most states individually have developed a set of BMPs for use in forest management. Aust (1994), summarizing the BMPs for forested wetlands in the southern Appalachian region of the U.S., noted that BMPs vary widely in definition of wetland types, streamside management zones, road construction, detail of the BMP manuals, and timber removal activities. Maine's BMPs are detailed in a Department of Conservation publication (Maine Bureau of Forestry 1991).

A great deal of research effort has been devoted to the topic of erosion and sedimentation from forest lands. This report reviews the literature dealing with the potential sources of forestry-related nonpoint source pollution and impacts on surface water, with particular emphasis on those studies conducted in Maine and in the Northeast. This review is organized along the following topics: sediment, temperature, nitrate, phosphorus, acidity, and herbicides.

Sediment

Sediment concentrations in rivers and streams draining forested watersheds in the United States are substantially lower than those draining watersheds that are heavily influenced by nonforest land uses. A survey of available sediment and erosion data from 812 plots and forested watersheds by Patric et al. (1984) revealed mean sediment concentrations ranging from 4–12 ppm for forested watersheds. The reported mean for the Penobscot River in ME (5 ppm, range 1–9 ppm) was among the lowest of the values reported. Corresponding sediment concentrations for nonforest land uses ranged from 40–1298 ppm. The data examined by Patric et al. (1984) provide the most comprehensive examination of the relative purity of water generated from forested watersheds, many of which were actively managed for forest products.

In New England, soil erosion from the undisturbed forest floor is rare because of a natural protective layer of surface litter and high soil infiltration capacity (Martin and Hornbeck 1994). Under undisturbed conditions in the eastern forest, soil erosion is primarily due to stream bank channel erosion (DeHart 1982). Erosion in undisturbed New England forests averages approximately 30–40 kg/ha/yr (Martin and Hornbeck 1994). Physical disturbance of the landscape near water bodies often increases sediment input. Erosion caused by forestry activities is most likely to occur in association with skid trails and haul roads in steep terrain and when soils are saturated, because of the combined potential for soil disturbance and for overland flow.

Water turbidity is often used as a measure of sedimentation. Several methods exist for measuring water turbidity, thus making direct comparison among various studies difficult. Jackson Turbidity Units (JTU), Nephelometric Turbidity Units (NTU), mg/L solids, and ppm solids are commonly used to measure turbidity. Turbidity units measured in ppm and mg/L solids are interchangeable assuming a density factor of one. Otherwise, no standardized conversion among turbidity units exists (Watson and Der 1986).

Forestry sources

One of the most famous studies examining the effects of clearcutting and herbicide application in a northern hardwood system was done at the Hubbard Brook Experimental Forest, New Hampshire (Likens et al. 1970). All vegetation growing on an entire watershed was felled and left in place. Vegetative regrowth was prevented by the repeated application of herbicides. No road building, skidding, or timber removal occurred in the watershed. Increases in stream water turbidity following treatment were negligible. The Hubbard Brook work provides the most direct evidence that cutting of trees *per se* is not the primary cause of erosion.

The many studies done prior to and after the Hubbard Brook devegetation experiment clearly document the overwhelming contribution of haul roads and skid trails as primary sources of erosion and sedimentation. Borne (1979) reported that one-third of the erosion problems in Maine were caused by skidding alone, while approximately 25% of the problems were attributed to truck roads. Log yards accounted for only a small portion of the erosion.

The closer the proximity of heavy equipment operation to streams, the greater the potential for persistent sedimentation problems. The removal of a large volume of trees (which generates greater skid and haul traffic) was associated with an increased likelihood of erosion. Patric (1976) also identified logging roads as the primary source of most soil lost from forest land in the East. More recently, King (1989) and Martin and Hornbeck (1994) reasserted that the major causes of erosion problems in New England are improperly designed landings and truck roads.

This association is a common theme throughout the literature dealing with this topic in the context of forestry and watershed management. The statement made by Hartung and Kress (1977: 6) captures the essence of the literature: "In the eastern woodlands, 99 percent of sediment originates on logging roads."

Many of the erosion problems with roads and skid trails originate at the point of intersection with running water. Problems associated with stream crossings can be very persistent. In New Brunswick and Nova Scotia, Grant et al. (1986) found increased siltation at both of the bridge crossings that were investigated, even though one of the crossings had been inactive for four years. No increases in sedimentation were noted in streams draining clearcut areas, probably because of the small size of the clearcuts and the time elapsed since cutting (greater than one year in five out of seven sites).

A survey of the effects of timber harvesting on erosion and sedimentation in New Hampshire illustrates the importance of conducting logging operations properly. DeHart (1982) found evidence of sedimentation in 13 of the 27 streams and ponds that were evaluated. All erosion and sedimentation problems were associated with the failure to properly follow one or more BMPs.

Even with careful harvesting, logging can cause increases in erosion and sedimentation if buffer strips are not used along waterways. Garman and Moring (1991) reported the effects of clearcutting along the East Branch of the Piscataquis River, Maine, associated with a fourth-order boreal river. A commercial clearcut removed 90% of the timber from the study area during January, leaving no buffer strips. Soil disturbance was minimal and the river channel was undisturbed. Timber was skidded to yarding areas and haul roads that were at least 500 m from the river. Suspended solids in the river were significantly higher after logging compared to before logging.

Even when buffer strips are utilized, loss of trees to windthrow may contribute to sedimentation. Lynch et al. (1985) reported the results of timber harvesting in a Pennsylvania watershed using BMPs, including a 30 m buffer strip in which very selective removal of high-value trees was permitted. Small increases in turbidity were caused by windthrow along an unprotected intermittent stream channel. Foster (1988) reported softwoods and older trees as being generally more susceptible to wind damage in central New England.

Hornbeck et al. (1987) reported the effects of strip and block clearcutting in the northern hardwoods of Hubbard Brook. The strip cut watershed had a 15 to 25 m variable width buffer strip, while no buffer strip was left during the block clearcutting. The control and both streams in the cut watersheds usually had turbidity levels below 1 JTU. However, the control stream never exceeded 5 JTU, while both harvested watersheds had readings above 11 JTU. High turbidity readings in the streams draining the cut watersheds were more frequent during storm events. Sediment accumulation behind the weir was similar for the cut and control watersheds. Peak flows during spring runoff were generally not increased, probably due to differences in timing of snow melt in cut and uncut areas.

The effects of whole-tree clearcutting for five sites in New England (including the Hubbard Brook site in NH and the Weymouth Point site in Piscataquis County, ME) were reported by Pierce et al. (1993). Whole-tree harvesting caused only minor changes in stream turbidity; the usual drinking water turbidity standard (10 JTU) was rarely exceeded. High turbidity levels for two samples at the fairly level Maine site (12 and 17 JTU) were observed during the harvest operation from the section of stream that was not protected with a buffer strip; maximum turbidity for the stream draining the control watershed was 3 JTU.

The hillier New Hampshire site was protected with a buffer strip and half was harvested in the winter. Turbidity values of 2,200 and 3,300 JTU were associated with a failure of a skid road culvert in the NH site. Otherwise, turbidity values were less than 1 JTU in the control and harvested watersheds. The authors concluded that whole-tree harvesting does not necessarily generate large increases in sedimentation so long as sufficient precautions are taken with stream crossings during logging. This finding supports the efficacy of BMPs.

In summary, the primary sources of nonpoint source (NPS) pollution from forestry activities (haul roads and skid trails) are widely recognized in the published literature. The problems occur where roads and trails intersect with, or channel runoff directly into, streams. Efforts to control NPS pollution should be directed specifically towards construction and maintenance of haul roads and skid trails.

Impacts

Turbidity and sedimentation caused by logging are major water quality concerns for forested watersheds. The effects of sediment on fish are primarily caused by reductions in stream bed gravel permeability, resulting in reduced embryo survival, habitat loss due to the filling of pools, and reduced food supply due to the impacts of sediments on lower trophic levels (MacDonald et al. 1991). Although sediment-caused decreases in invertebrates are most often attributed to reduction in habitat availability, abrasion and clogging of respiratory systems by suspended particles also can be detrimental, especially to filter-feeding functional groups such as Trichoptera (Garman and Moring 1987). Logging operations can degrade lakes and reservoirs by filling them with sediment transported by streams and by increasing turbidity.

Decreases in salmonid populations have been linked to forestry-related sediment inputs in Nova Scotia and New Brunswick (Grant et al. 1986). Total salmonid biomass (brook trout [*Salvelinus fontinalis*] plus Atlantic salmon [*Salmo salar*]) was significantly lower downstream from both bridge crossings investigated, most likely because of increased siltation. At the forestry sites where no increases in sediment were noted, total salmonid biomass was not altered.

Decreases in invertebrate populations and fish habitat degradation resulting from turbidity and sedimentation have been documented in Maine. Taylor (1989) found that artificial sediment additions increased invertebrate drift, particularly among chironomids and simuliids. Decreased invertebrate density occurred in low gradient stream sections where sand buried the more favorable habitat. After logging, Garman and Moring (1991) found higher levels of fine sediment in gravel pools of the East Branch of the Piscataquis River.

Transport

Downstream transport of sediments as the result of forestry activities has been observed (Lyons and Beschta 1983; Haines et al. 1990). Transport of sediment is primarily controlled by current velocity and sediment size. Conditions such as high discharge, high gradient, and channel confinement increase current velocity, and subsequently increase the transport of sediments. Smaller sediment particles require less energy to move, causing them to be more prone to long-range transport. When streams flow into lakes and reservoirs, the lack of current causes the deposition of sediment into these environments.

Effects of BMPs on sedimentation

Of all the pollutants resulting from land use activities, sediment is probably the major water quality concern (Binkley and Brown 1993a). It is not surprising that the majority of BMPs are targeted at controlling sedimentation (Maine Forest Service 1994). Natural levels of sediment in forest streams are highly variable. Across North America most forest streams average < 5mg/L suspended sediments, although some exceed 20 mg/L (Binkley and Brown 1993a). Virtually all studies have shown that best management practices are very effective at reducing or eliminating the transport of sediments into watercourses (Lynch and Corbett 1990; Commerford et al. 1992; Binkley and Brown 1993a; Farrish et al. 1993; Martin and Hornbeck 1994). The most effective factor for preventing sediments and nutrients from reaching a watercourse is a vegetative buffer strip (Gilliam 1994). The first step in preventing sedimentation in the Northeast is minimizing the area of disturbed forest floor (Bormann et al. 1974; Patric 1976; Megahan 1981; Hornbeck et al. 1987). Next is preventing the movement of sediments across the landscape. Preharvest planning of skid trails, yard areas, and truck roads are of paramount importance. Gravel and grass surfaces were found to reduce soil loss in a study of the effects of different road surfaces on reducing soil loss (Swift 1984; Megahan et al. 1992).

Harvesting

It is important to select logging methods and equipment to minimize soil disturbance. This should include proper layout of roads and skid trails (Hornbeck et al. 1986). Heavy equipment or skidding activities should not be allowed in or across drainage ditches (Askew and Williams 1984). Winter logging should always be considered as an option for protecting fragile sites from erosion (Hornbeck et al. 1986).

Maine's Land Use Regulation Commission conducted a survey of 405 logging operations to determine the extent of sedimentation and the effectiveness of current regulations (Borne 1979). They found that nearly all the erosion problems, other than haul roads and yards, were associated with skid trails. About two-thirds of the gully erosion was caused by skid trails and one-third by yards and landings. Surprisingly, the length of gullies was not associated with slope; long gullies occurred on gentle as well as steep slopes. Seventy-five percent of the cases where erosion persisted occurred within 250 feet of streams.

As noted previously, the Hubbard Brook experiment, where all the trees in one watershed were felled and left on the ground, showed that the actual felling of trees did not cause significant increases in sedimentation (Likens et al. 1970; Hornbeck et al. 1987). However, three to five years after cutting, stream sediments increased. This increase was attributed to the decomposition of woody debris in the stream resulting in greater stream velocities and subsequent bank erosion (Bilby and Likens 1980; Martin and Hornbeck 1994).

In Pennsylvania's Leading Ridge Experimental Watershed, harvested areas showed no increase in sediment levels until two to three years after harvest (Lynch and Corbett 1990). These increases were attributed to three factors: (i) windthrow in the buffer strip that exposed mineral soil; (ii) 1,500 feet of intermittent stream was exposed without a buffer strip; and (iii) an intermittent stream became perennial after harvest (Lynch and Corbett 1990).

It is important that BMPs be implemented over the entire harvest area since runoff is regulated by natural processes throughout the watershed. For example, a study of 1,584 clearcut sites in Louisiana found that if runoff was allowed to channelize before reaching a buffer strip the channel would likely reach the watercourse (Farrish et al. 1993). Farrish also emphasized the effectiveness and importance of installing water bars on skid trails and ensuring herbaceous revegetation once harvesting is completed.

Site preparation

Since sediment loss is a function of soil disturbance one would expect intensive site preparation to result in sediments moving into watercourses. Any site preparation such as “windrowing” that causes heavy soil disturbance should be avoided (Farrish et al. 1993).

A 5,900-acre natural hardwood forest in South Carolina was converted to a loblolly pine plantation and monitored at 20 sampling points after 14 rainstorms over a period of two years. Overall water quality was not adversely affected when compared to the hardwood control as long as the appropriate BMPs were implemented (Askew and Williams 1986).

However, drainage ditches have been installed on one million hectares in southern wetlands. Now the pattern is going to "minor drainage" (Williams 1994).

A study in Oregon found that slash burning for site preparation was followed by the largest increase in sediment loading (Fredrickson et al. 1975). Similar effects of slash burning have not been detected in the East probably due to differences in soil physical properties and more gentle topography and riparian zones (Borg et al. 1988).

Operations in wetlands

There has been little work on the effects of silvicultural activities in wetlands on water quality in the Northeast. However, studies from a spruce swamp in Michigan have shown immediate increases in NO_3 and NH_4, then they begin to decrease in concentration. However, NO_3 and NH_4 were still elevated over levels in a control stand five years after harvesting (Trettin et al. 1996). Nitrate concentrations five years after harvesting were well below the drinking water standard of 10 ppm established by EPA. Similar results occurred in a spruce peatland in the subarctic region of Quebec (Moore 1987). The decline in forest cover in riparian areas has been associated with declining water quality, including increased sedimentation and nutrient loading (Childers and Gosselink 1990).

Suggestions

Since cut and fill slopes are often primary causes of sedimentation (Swift 1986) innovative techniques to mitigate such sources of sedimentation should be implemented. A variety of materials have been used to slow water velocity, allowing particles to settle from moving water. One such technique is the utilization of slash for the construction of filter windrows on fill slopes. Cook and King (1983) found filter windrows to be very effective (75%–85%) in preventing material from leaving fill slopes in a study conducted in Idaho. The effectiveness of straw mulch for controlling erosion from disturbed surfaces has also been demonstrated. Higher straw levels resulted directly in less soil loss (Zuzel and Pikul 1993). Different combinations of rocks and mulch on disturbed surfaces were examined in Colorado to determine the most effective mix to prevent soil loss from disturbed sites. A combination of 50% rock and 50% mulch/litter minimized soil loss; rocks alone were not as effective as litter (Benkobi et al. 1993).

One of the earliest and best data sets demonstrating the effectiveness of BMPs in reducing sedimentation was reported by Reinhart and Eschner (1962) and cited by the Society of American Foresters (1995). Water samples were collected from four watersheds in the Fernow Experimental Forest in West Virginia logged during 1957 and 1958. Samples showed a marked reduction in turbidity with increased level of care and planning for skid trails and haul roads (Table 1). As experimental erosion control measures improved (no control on skidding and road construction on grades up to 40%, to no channel skidding, water bars, vegetation stabilization, and restriction of roads to grades under 20%), the distribution of turbidity measurements became indistinguishable from those obtained from unharvested control areas.

Temperature

Sources

Any forestry practice that reduces canopy shading of water bodies has the potential to increase water temperatures. Sullivan et al. (1990) reported that removal of the canopy could increase stream temperatures by 5°C or more. Others have reported increases in stream temperatures from 2-10°C after clearcutting depending on latitude, altitude, remaining vegetation and amount of logging debris (White and Krause 1993; Lynch and Corbett 1990). Brown (1985) presented a model to predict maximum temperature increases as a result of canopy removal. The model predicts that streams with low discharge and shallow stream beds are more susceptible to temperature increases. The strength of this general relationship is further reinforced by White and Krause’s (1993) review. They noted that small streams are much more subject to heating than large ones.

Increases in stream temperature can be reduced or entirely prevented by leaving a protective tree cover along streams. In most cases the retention of buffer strips limits the maximum stream temperature increase to less than 2°C (Binkley and Brown 1993b) and increases have generally been linked to the absence of buffer strips. Garman (1984) linked increased summer water temperatures and larger diurnal temperature fluctuations to the removal of 80% of the riparian vegetation on the East Branch of the Piscataquis River, Maine, a fourth-order boreal river. Noel et al. (1986) reported that cut over streams in New England, particularly those lacking a buffer strip, had higher maximum temperatures during the summer. A maximum increase of 7°C was found at a Vermont site with no buffer strips. One year after logging, Burton and Likens (1973) reported a 4-5°C increase and larger diurnal temperature variation in unbuffered stream sections at Hubbard Brook. Two years after the cut, temperature differences between the buffered and unbuf-

Table 1. Frequency distribution of turbidity[1] samples during logging[2] on watersheds roaded under increasingly stringent erosion control measures (Society of American Foresters 1995, based on data of Reinhart and Eschner [1962]).

	Frequency Distribution for Turbidity (JTU) of stream samples				
	No. of Samples by Turbidity Class				
Experimental Erosion Control Measure[3]	0–10	11–99	101–999	1000+	Max. JTU
Channel skidding permitted. No control over road location, grades up to 40%, no water bars, 7.3% of watershed roaded.	126	40	24	13	56000
Channel skidding permitted. No control over road location, grades up to 30%, water bars at 35 ft intervals, 6.2% of watershed roaded.	171	17	8	7	5200
No channel skidding. No control over road location, grades up to 20%, water bars as needed, 5.8% of watershed roaded.	195	8	0	0	210
No channel skidding or wet weather logging. Roads located about 40 ft from stream, water barred, and seeded to grass after logging. Road grades up to 20%. 1.9% of watershed roaded.	201	2	0	0	25
Permanently forested.	202	1	0	0	15

[1]Turbidity expressed as JTU was assumed to approximate parts per million of sediment.
[2]Turbidity returned close to levels in the permanently forested watershed two years after logging ceased.

fered sections were reduced (2°C), probably due to partial regrowth of streamside vegetation.

A stream within a whole-tree clearcut in Maine with no buffer strips frequently had stream temperatures 4°C higher than the stream in the uncut control, with an average increase of 2°C (Pierce et al. 1993). In contrast, a stream in a whole-tree clearcut in New Hampshire with buffer strips never was more than 2°C warmer than the uncut control, and usually within 1°C (Pierce et al. 1993). A study that had no uncut buffer strips, but reduced shading by only 5%, resulted in no significant temperature changes in a small tributary of the Aroostook River, Maine (Mullen and Moring 1988).

Small increases in stream temperature may occur if intermittent streams are not protected. Lynch et al. (1985) harvested a Pennsylvania watershed using best management practices, including a 30 m buffer strip in which a very selective removal of high-value trees occurred. Slight increases in summer water temperatures (1-2°C) were linked to intermittent streams that had become perennial after logging because of reduced evapotranspiration. The warmer water from the previously intermittent stream heated the water of the receiving stream.

Impacts

The narrow width of small streams allows for a high degree of canopy shading, resulting in cool summer water temperatures. Higher temperatures reduce the amount of oxygen that can dissolve in water, and can be stressful or lethal to cold-water biota. Temperatures below 20°C are generally considered suitable for salmonids (Wallace 1993). Prolonged temperature exposure above 25°C is lethal to brook trout (Stolz and Schnell 1991), and temperatures above 27°C are lethal to Atlantic salmon (Elliot 1991). Protecting small streams from thermal degradation is especially important when they provide temperature retreats from the higher summer temperatures in larger streams. In river systems with Atlantic salmon, protecting small streams is important because first-order streams constitute 20%–40% of the available habitat (Haines and Akielaszek 1984).

Increased temperature has been linked to changes in stream biota in the Northeast. Garman (1984) reported the higher summer water temperatures and larger diurnal water temperature fluctuations after clearcutting along the East Branch of the Piscataquis River. Increases in water temperatures were believed to be responsible for the disappearance of brook trout. At Hubbard Brook, a 4-5°C increase in stream temperature and increases in organic debris were believed to be responsible for the loss of the larval two-lined salamanders (*Eurycea b. bislineata*) from the stream (Burton and Likens 1973). Noel et al. (1986) surveyed clearcuts in New England. Greater abundances of algae in streams draining clearcut watersheds were attributed to higher temperatures and greater light availability. Increased temperature, algal concentrations, and detritus were believed to be responsible for the higher abundance of invertebrates in clearcut streams.

When harvesting reduced shading by only 5%, no significant temperature changes occurred in a small tributary of the Aroostook River, Maine (Mullen and Moring 1988). Correspondingly, no significant differences in periphyton or brook trout populations were noted after cutting.

Transport

Increased water temperatures are most likely to persist at downstream locations when water temperatures are increased on very cold streams (Binkley and MacDonald 1993). Low volume streams and streams with shallow streambeds are the most susceptible to temperature increases, but should dissipate excess heat more rapidly. The work done by Burton and Likens (1973) at Hubbard Brook suggests that small headwater streams (July discharge = 0.567 l/s) return rapidly to unshaded temperatures. Pierce et al. (1993) reported that temperature quickly decreases to pre-cutting levels when streams flow into shaded areas or regrowth provides new shade.

Effects of BMPs on temperature

Results from the Leading Ridge Watershed in Pennsylvania showed that debris dams in streams contribute to higher temperatures despite the benefits of such dams in controlling sediment transport (Lynch and Corbett 1990). A buffer strip of trees as wide as one tree height on the south side of streams is most effective in preventing stream temperature increases (White and Krause 1993). The previously cited literature discussed above clearly shows that buffer strips are effective in minimizing impacts of harvesting on stream temperature.

Nitrate

Nitrogen, an essential plant nutrient, is cycled through forested systems in two inorganic forms: ammonium (a cation attracted to colloidal exchange complexes) and nitrate (an anion repelled from colloidal exchange complexes). Consequently, when nitrogen is lost from forested systems it is lost as nitrate. Under undisturbed conditions, nitrogen cycles through the forest ecosystem efficiently. The result is low nitrate concentrations in forested streams, particularly during the peak of the growing season. Forest removal alters the nitrogen cycle, which can result in higher nitrate concentrations in nearby streams, particularly in northern hardwood watersheds. For consistency, all nitrate concentrations are reported as nitrate-N.

Forestry sources

Martin and Pierce (1980) investigated the effects of clearcutting on nitrate concentrations in the White Mountains of New Hampshire. Twenty-one of the 23 sites examined were dominated by northern hardwoods. In the uncut reference watersheds, the mean annual nitrate concentration was 0.4 mg/L. In the entirely clearcut watersheds, the mean annual nitrate concentration the first year after harvesting averaged 2.4 mg/L, and ranged from 3.6 to 5.9 mg/L. During the second year, the mean annual nitrate concentration was 4.0 mg/L, and ranged from 4.5 to 6.3 mg/L. By the end of the fourth year after harvesting, nitrate levels had returned to reference levels in eight out of nine entirely clearcut streams. The partially clearcut watersheds followed a pattern similar to the entirely clearcut watersheds, but the average nitrate concentrations were lower. Clearcutting the upper slopes of the watershed increased nitrate concentrations in stream water, while buffer strips appeared to reduce nitrate concentrations. The lowest nitrate concentrations occurred in a progressive strip cut with a buffer strip.

In New England locations outside the White Mountains, concentrations of nitrate in stream water increase less dramatically, if at all, in response to cutting. Martin et al. (1984) conducted a study of clearcutting (stem only and whole-tree) that was designed to detect obvious changes in stream chemistry at 15 locations in New England, including one hardwood and four conifer sites in north-central and eastern Maine. In the northern hardwoods of Maine and Vermont, nitrogen (nitrate plus a very small amount of ammonium) concentrations ranged from 0.1 to 0.7 mg/L in the reference streams as well as in the streams draining partially cut and clearcut areas. At the conifer sites in Maine, clearcut and reference streams both had nitrogen concentrations of 0.1 mg/L or less. Martin et al. (1984: p. 210) offered the following explanation for their observations:

> Clearcutting of entire watersheds tends to be rare in New England; it was difficult for us to locate such areas to sample. Stream water from uncut portions of partially clearcut watersheds may also have diluted any nutrient-rich water from clearcut areas. Therefore, partial cuts, advanced regeneration, rapid revegetation, buffer strips, and extended duration of cutting may have all helped to reduce the nutrient concentrations in streams draining cut over areas.

Only in northern hardwoods of the White Mountains were large increases in nitrogen detected by Martin et al. (1986). Streams draining the uncut forest had total nitrogen concentrations less than 0.5 mg/L, while those draining clearcut watersheds averaged 2.0 mg/L.

Increases in stream water nitrate due to clearcutting northeastern forests are not isolated to the White Mountain region. Nitrate concentrations were monitored in a clearcut and uncut watershed in central New Brunswick (Krause 1982). The clearcut watershed contained approximately 51% softwoods and was logged between May 1978 and February 1979. Nitrate export from the clearcut watershed increased 9% in 1978, 236% in 1979, 368% in 1980, and 329% in 1981. The nitrate concentrations were generally highest in the winter and spring, particularly in the clearcut watershed. The maximum concentration of nitrate in the stream of the main channel for the clearcut watershed was 1.6 mg/L, while that for the control site peaked at 0.2 mg/L. In the clearcut watershed, concentrations up to 2.5 mg/L were observed in one of the tributaries from a predominantly hardwood drainage.

Pierce et al. (1993) reported the effects of whole-tree clearcutting in New England, including the Weymouth Point site in Maine and a site in New Hampshire. The Maine site was a "fairly level" spruce-fir forest. The mean monthly nitrate concentration of the uncut stream averaged 0.05 mg/L, and peaked at 0.12 mg/L. In contrast, the whole-tree harvested site had maximum mean monthly nitrate concentrations of 0.9 mg/L in the first year after cutting and 0.7 mg/L in the second. The New Hampshire site was northern hardwoods, containing some slopes exceeding 10%. A buffer strip was left along the stream channel, and the site was harvested in two cuts separated by half a year. The mean monthly nitrate concentration peaked at 1.5 mg/L in the first season after harvesting was complete, compared to 0.8 mg/L for the control. Two years after the cutting ended, the control and harvested sites had similar nitrate concentrations.

Herbicide application can increase nitrate concentrations in streams. Triclopyr BEE was applied by helicopter at 1.9 kg ae (acid equivalent)/ha to a regenerating whole-tree spruce-fir harvest in Maine in August 1985 (Smith et al. 1988). Concentrations of nitrate reached a maximum of 1.6 mg/L in November 1985. In British Columbia, significant increases in stream water nitrate occurred for four years when vegetation cover in a watershed was reduced 43% by glyphosate application, but no significant nitrate increase occurred with a 4% vegetation reduction (Feller 1989).

Impacts

Nitrate is the form of elemental N that is most important in limiting phytoplankton growth in estuaries, although phosphorus may be important under some conditions (Kennish 1992). Nitrogen inputs to estuaries may increase phytoplankton growth, leading to eutrophication. The extent to which forestry practices affect coastal eutrophication and the extent of coastal eutrophication are not known for Maine, although anecdotal evidence suggests nutrient enrichment in estuaries may be a problem (Maine Department of Environmental Protection 1994a).

Per unit area, streams draining forested watersheds have the lowest nitrate concentrations (Omernik 1976). However, forested areas constitute 78% of the surface of Maine, whereas cropland and pasture make up 2.8% and 0.8% respectively (Maine Department of Environmental Protection 1994a). Even if nitrate concentrations in stream water are unaffected by forestry activities, nitrate export can increase because of the higher volume of water from cut watersheds. However, research at the Weymouth Point Watershed shows that this increase could be very small. Nitrate export from 1991 through 1994 from the 120 ac watershed clearcut in 1981 ranged from 0.07-0.16 kg/ha/yr (Briggs and Hornbeck 1995). That number is significantly lower than nitrate inputs in rain (11.1 kg/ha/yr).

The EPA limit for nitrate nitrogen in drinking water is 10 mg/L. Although nitrate concentrations may increase as a result of forest harvest and herbicide application, the EPA limit was not exceeded in any of the studies reviewed. Based on the work of Kincheloe et al. (1979), the EPA limit should be adequate to protect the sensitive stages of most salmonids, although data on the majority of species found in Maine are lacking. Work done by Camargo and Ward (1992) indicates concentrations of nitrate below 10 mg/L should not be harmful to aquatic invertebrates.

Transport

Nitrate transport is higher at low temperatures because of reduced rates of denitrification and immobilization (Perrin et al. 1984). Nitrate concentrations are typically highest during the dormant season, particularly during periods of high flow. Therefore, the greatest flux of nitrogen to downstream reaches should occur during periods of high discharge and low temperatures.

Martin et al. (1986) reported concentrations of nitrate at various locations downstream of a clearcut. Water samples were collected from streams that drained a completely clearcut area and at downstream points where the clearcut area represented 63%, 17%, 13%, and 6% of the cut area. Nitrate concentrations were highest at the 100% location and decreased at the 63% location and again at the 17% location. At the 13% and 6% locations, the concentrations of nitrate were not different from the uncut reference. It is unclear from this study whether the reductions in the nitrate concentrations were caused by dilution alone, or by a combination of dilution and nitrate removal from the stream.

The rate of nitrogen removal in small agricultural streams was investigated in Norway by Faafeng and Roseth (1993). Nitrate was added to the streams at varying rates, and concentrations of nitrogen (nitrate plus nitrite) were monitored at downstream locations. They found the higher the concentrations of nitrate, the greater the rate of removal from stream water. Therefore, large increases in nitrate export should not persist for great distances, but small increases may. This study suggests that if nitrate concentrations are rapidly reduced by dilution, the rate of removal may also decrease, thereby increasing transport.

Phosphorus

Phosphorus is an essential plant nutrient, and is very strongly conserved in forest ecosystems (Wood et al. 1984). The annual weighted concentration of dissolved phosphorus + fine particulate phosphorus in an undisturbed stream at Hubbard Brook was 1.18 µg/L (Hobbie and Likens 1973). Total phosphorus in water is made up of both dissolved (primarily phosphate) and particulate forms, with particulate forms constituting the bulk (Hobbie and Likens 1973; Wetzel 1975). Thus, total phosphorus tends to be coupled with suspended sediments (Verhoff et al. 1979; Childers and Gosselink 1990).

Forestry sources

Few studies have documented the impact of forestry on phosphorous export in the Northeast. Work done by Wood et al. (1984), and Hornbeck et al. (1987) estimated that strip and block clearcutting did not increase losses of phosphorus in streams at Hubbard Brook. The low rates of erosion in these cuts helped prevent increases in total phosphorus export.

Across North America, changes in phosphate concentrations in streams due to timber harvest are uncommon (Binkley and Brown 1993b). In Ontario, small increases in phosphate and total phosphorus in filtered stream water were recorded one year after strip cutting an upland black spruce forest, but not in subsequent years (Nicolson 1988). Knighton and Stiegler (1981, as cited by Verry 1986) found that clearcutting a black spruce bog increased phosphorus export from 0.33 to 0.89 kg/ha in the first year after harvest. Within three years, export had returned to preharvest levels. Replanting after harvesting in spruce swamps may actually show a decrease in PO_4 concentrations because of vegetative uptake (J.W. McLaughlin, pers. comm.).

Impacts

Phosphorus is the nutrient that limits algal growth in fresh water (Hecky and Kilham 1988), and increased phosphorus loads can cause eutrophication. In lakes, coldwater fishes are particularly sensitive to phosphorus inputs because of the resulting anoxia in the deep, coldwater habitat. A model to predict the sensitivity of a lake to phosphorus inputs is available through the Maine Department of Environmental Protection (1987). The predictions of this model, by county, are presented in Table 2.

In Maine, timber harvesting is thought to cause, or contribute to, nutrient problems in a few lakes, although agriculture and shoreline development are bigger problems (Maine Department of Environmental Protection 1994b). Assuming that increases in dissolved phosphorus concentrations in streams due to timber harvest are uncommon in Maine, increases in phosphorus as a result of timber harvest should be restricted to sediment bound inputs, although the findings of Knighton and Stiegler (1981) suggest further work on forested wetlands is warranted.

Most of the phosphorus in sediments is not bioavailable (Dorich et al. 1985; DePinto et al. 1981). The Maine Department of Environmental Protection (Noel et al. 1992) assumes that 25% of the particulate phosphorus and 100% of the dissolved phosphorus will become available for uptake by algae. By adequately controlling erosion, phosphorus inputs associated with sediments should be avoidable.

Transport

Transport of particulate phosphorus is coupled with the transport of sediments (Verhoff et al. 1979; Childers and Gosselink 1990). Sediment transport is discussed in a separate section.

Effects of BMPs on nutrients

The impacts of different harvesting regimes on the cycling of nutrients in northern hardwood forests has been intensively studied at the Hubbard Brook watersheds. Increases in nitrate of up to 2.0 mg/L were observed after a complete harvest and no increase if <70% is harvested (Martin et al. 1984). Nitrate responses were lower in the strip cut treatments than in the clearcut (Martin and Pierce 1980). In no cases did nitrate levels at Hubbard Brook exceed 10 mg/L (potable standard) (Hornbeck et al. 1987).

Table 2. Percentage of lakes predicted by Maine DEP to be extremely or highly vulnerable to phosphorus imbalance.

County	% Vulnerable	County	% Vulnerable
Androscoggin	69	Oxford	19
Aroostook	5	Penobscot	25
Cumberland	84	Piscataquis	1
Franklin	7	Sagadahoc	100
Hancock	28	Somerset	7
Kennebec	61	Waldo	28
Knox	53	Washington	0
Lincoln	60	York	8

Similar results were observed at the Leading Ridge watershed in Pennsylvania where modest increases in stream nutrient levels were recorded the first year after harvest but remained well below drinking water standards. After four years there were no differences in water chemistry relative to preharvest levels (Lynch and Corbett 1990).

Vegetation, especially that with an extensive root network, is important in a buffer strip since 73% of nitrogen exported from a site enters a buffer strip through subsurface flow (Peterjohn and Correll 1984). The importance of the role of vegetation in the sequestration of nutrients is obvious.

Acidity

The influence of timber harvest on soil pH depends on the balance between the release of base cations (Na^+, K^+, Ca^{+2}, Mg^{+2}) and the acidity generated by nitrification and sulphur oxidation, with nitrification being far more important (White and Krause 1993). Depending on the balance, forestry practices can raise or lower the soil pH, subsequently influencing the pH of streams (White and Krause 1993).

Sources

Martin et al. (1984) conducted a study of clearcutting (stem only and whole-tree) that was designed to detect changes in stream chemistry at 15 locations in New England, including one hardwood and four conifer sites in north-central and eastern Maine. In the coniferous forest at Ragmuff, Maine, an average pH drop of 0.8 was detected in streams draining watersheds with more than 30% of the area clearcut. In one entirely clearcut watershed in the northern hardwoods of Vermont, the pH was 0.5 units lower than the reference. At all other locations investigated, the pH was either unchanged or less acidic in the streams draining the cut watersheds.

The effects of clearcutting on stream chemistry in the White Mountains of New Hampshire were reported by Martin et al. (1986). In the first two years after harvesting, the average pH in the streams draining the cut and uncut watersheds was similar, although two streams were more acidic after cutting, and one stream was less acidic. However, in the third and fourth years after timber harvest, the streams in the cut watersheds had an average pH of 6.0 while the controls averaged 5.4.

At Hubbard Brook, Hornbeck et al. (1987) reported that strip clearcutting reduced the pH of stream water from 5.7 to 5.3 and increased the variability in the second and third years after cutting began. By the fourth year, the pH had returned to pre-harvest values. Block clearcutting decreased the pH from 5.0 to 4.8 in the first two years after cutting. After the second summer, the pH began to rise. During the fourth through tenth years after harvesting, the block cut watershed averaged approximately a half a unit higher than pre-cut values.

Garman and Moring (1991) reported the effects of clearcutting along the East Branch of the Piscataquis River, Maine, a fourth-order boreal river. A commercial clearcut removed 90% of the timber from the study area during January, leaving no buffer strips. No significant changes in pH were noted in the river one year after logging.

Pierce et al. (1993) discussed the potential for whole-tree clearcutting to acidify soils and streams. Growing trees take up nutrient cations at a greater rate than nutrient anions. To maintain the charge balance, trees release H^+ ions to the soil. Under undisturbed conditions, decomposition of organic matter and nutrient mineralization return base cations to the soil, presumably buffering against increased acidity. However, soil acidification is associated with aggrading systems because of the rapid uptake of base cations from the exchangeable pool coupled with production of organic acids as forest floor decomposition proceeds. Shepard et al. (1990) documented the increase in soil acidity associated with aggrading red pine plantations growing on a K deficient outwash plain in northern New York.

Whole-tree harvesting may contribute to soil acidification by removal of base cations in forest products that would otherwise become components of the forest floor. This is of particular concern on poor quality sites, where whole-tree harvesting and short rotations should be avoided. Pierce et al. (1993) estimated that the soil acidification produced by their whole-tree clearcuts was the equivalent of 50 years of pH 4.0 precipitation. Long-term data needed to evaluate the extent of acidification caused by biomass removal are lacking. In the short term, decreases in acidity may be neutralized by increased weathering and decreased sulfate leaching, unless nitrification rates are high. Pierce et al. (1993) found that whole-tree clearcutting did not affect the pH of streams at a spruce-fir site in Maine, or a northern hardwood site in New Hampshire. At Hubbard Brook (which has high rates of nitrification), Lawrence et al. (1987) reported that the pH of the stream draining the cut watershed increased slightly relative to the control immediately after harvesting, and then dropped, averaging approximately 0.3 units lower than the control in the following eight months.

Impacts

In most studies, forestry practices have not been shown to degrade water quality with regards to acidity. In northern hardwood sites with high rates of nitrification, small decreases in stream water pH may occur. After the initial drop, however, the pH typically returns to preharvest or even less acidic levels. In softwoods, the only pH decrease detected was at Ragmuff, Maine. At all other sites, pH either remained unchanged or increased as a result of forest harvest. Concerns have been raised that intensive biomass removal will eventually lead to soil and stream acidification, especially in areas containing soils with low buffering capacity (Pierce et al. 1993). With the spread of intensive forestry, this may become an important issue in the future.

The detrimental effects of low pH on aquatic systems have been widely documented, and the forestry operations that decrease stream pH may adversely affect aquatic biota. Decreased pH in water bodies is usually associated with elevated aluminum (Haines and Akielaszek 1983; Driscoll 1985) and large increases in aluminum have been documented at forestry operations that decrease stream pH (Likens et al. 1970; Lawrence et al. 1987). Decreased pH combined with elevated aluminum is highly toxic to fish (Baker and Schofield 1982; Buckler et al. 1987). Periods of high discharge usually have the lowest pH levels and highest aluminum concentrations (Haines et al. 1990). Acid-aluminum toxicity is reduced by dissolved organic carbon (DOC) (Peterson et al. 1989) and calcium (Ingersoll et al. 1990).

Acid damage to biota is most likely to occur at locations where the pH is already low. Sixty of the 1,005 Maine lakes assessed for acid damage are affected by high acidity (Maine Department of Environmental Protection 1994a). Small headwater lakes and low-order streams in New England, particularly in areas underlain by bedrock low in buffering capacity, are generally more acidic and have lower calcium concentrations than do higher order streams (Haines and Akielaszek 1983). Therefore, acid-sensitive biota inhabiting these waters, particularly if they are low in DOC, should be the most susceptible to increased acidity.

In New England, small streams constitute 20%-40% of the available habitat for Atlantic salmon (Haines and Akielaszek 1984). Atlantic salmon toxicity occurs at pH levels below 5.0-5.5 (Jagoe and Haines 1990). In Maine, the levels of pH and aluminum concentrations found in first-order streams may be toxic to sensitive life history stages of Atlantic salmon (Haines and Akielaszek 1984). The acid-aluminum conditions present in larger streams generally do not appear to be hazardous to Atlantic salmon (Haines and Akielaszek 1984). Brook trout are generally more tolerant of acidic conditions than are Atlantic salmon (Rosseland et al. 1986; Haines et al. 1990). Furthermore, brook trout have been documented to migrate to avoid acidic conditions (Gagen et al. 1994). Striped bass (*Morone saxatilis*) are very sensitive to acidic conditions, and rivers with an average pH of 6.3 during the spawning season experienced reproductive failure in the Chesapeake Bay system (Hendrey 1987). These levels of acidity are also toxic to blueback herring (*Alosa aestivalis*) (Hendrey 1987).

Transport

The transport of acidity in streams depends on the characteristics of the downstream drainages. The acidity will be transported until the downstream reaches have the ability to increase basic cation concentrations, relative to acidic anions (Lawrence et al. 1987).

Herbicides

In Maine, glyphosate and triclopyr are used in intensive forestry to reduce competition from commercially undesirable species. Glyphosate is available as an isopropylamine salt of glyphosate in Roundup, Accord, Rodeo, and Vision. Triclopyr is available as a triethylamine salt (TEA) in the Garlon 3A formulation, and as a butoxyethyl ester (BEE) in the Garlon 4 and Release formulations. For additional information on forest chemicals, comprehensive reviews are available by Norris et al. (1991), Shipp et al. (1986), Grossbard and Atkinson (1985) and a collection of abstracts (Sullivan and Sullivan 1993). The effect of herbicides on nitrate concentration is covered in the nitrate section.

Glyphosate

Forestry sources

Newton et al. (1984) conducted a study of glyphosate in an Oregon watershed. Glyphosate was applied at 3.3 kg acid equivalent per ha (ae/ha), by helicopter, with no buffer to protect the stream. Glyphosate residues in stream water peaked at 0.27 mg/L shortly after spraying began, and declined exponentially to near zero mg/L within 10 days. Direct application was the main source of glyphosate to the stream, as glyphosate is immobile in soil (Norris et al. 1991; Tortennson 1985). Concentrations of glyphosate in the sediments peaked slightly above 0.50 mg/L 14 days after spraying, and occurred at approximately 0.10 mg/L after 55 days. Neither glyphosate nor aminomethylphosphonic acid (the major metabolite of glyphosate) were detected in fish.

Impacts

Toxicity data indicate Roundup is moderately toxic, with greater toxicity at higher temperatures (Folmar et al. 1979; Atkinson 1985). Early life history stages of rainbow trout (*Oncorhynchus mykiss*) and channel catfish (*Ictalurus punctatus*) were more susceptible to Roundup (Folmar et al. 1979).

The highest concentration of glyphosate in stream water observed by Newton et al. (1984) was 0.27 mg/L. This concentration barely exceeded the NOEC (no observed effect concentration) for rainbow trout swim-up fry (0.24 mg/L), and did not exceed the NOEC for rainbow trout fingerlings (0.83 mg/L) or *Daphnia* (0.30 mg/L) (Norris et al. 1991). Because direct application was the main source of glyphosate to the stream, concentrations would have been much lower if buffers were used. Therefore, with the use of buffers and proper application procedures, the potential for glyphosate toxicity would be minimized (Tooby 1985; Chapman 1989). The maximum concentration of glyphosate in stream water reported by Newton et al. (1984) did not exceed the 0.7 mg/L drinking water standard set by the EPA. Under a realistic worst-case scenario, estimated buffer strips of 25–30 m would limit mortality of salmonids and aquatic invertebrates to less than 10% (Payne et al. 1989, 1990).

Transport

Glyphosate has been documented to travel long distances when it reaches moving water. Residues in stream water dissipate rapidly with concentration below detection limits (0.1 ppb) within 9bh of the application (Feng et al. 1989). In Washington State, Comes et al. (1976) added glyphosate to flowing canal water and found 70%-72% of the herbicide was present after 1.6 km of travel. After 8 km in one canal and 14 km in another, 57% and 58% of the glyphosate, respectively, was still present. However, the presence of sediment with water has been known to greatly reduce glyphosate efficacy; glyphosate breaks down when it comes into contact with soil (Rueppel et al. 1977; Torstennson 1985; Bronstad and Friestad 1985; Feng and Thompson 1989). Applicators know that care must be exercised in drawing sediment-free water from streams that are used to mix with the chemical in the tanks for application to forest land.

Triclopyr

Forestry sources

Thompson et al. (1991) investigated the environmental fate of triclopyr in a boreal watershed in northern Ontario. Garlon 4 was applied at 3.84 kg ae/ha by helicopter, with no buffer to protect the stream. The highest concentrations of triclopyr BEE in stream water (up to 0.35 mg/L) were detected during and immediately after applications. Triclopyr acid (a less toxic breakdown product of triclopyr BEE) reached a maximum of 0.14 mg/L six hours after spraying had commenced. A rainfall event two days after spraying increased stream water concentrations of triclopyr. Concentrations of triclopyr in stream water fell below detection (0.001 mg/L) within 72 hours. The main source of triclopyr to the stream was direct application, with small amounts being washed off the vegetation in association with a rainfall event soon after the application. Neither long-range over-land flow nor leaching were believed to be significant sources of triclopyr to the stream. Triclopyr has low mobility in the soil, especially in soils of high organic matter content (Norris et al. 1991). Triclopyr concentrations in the stream sediments were generally below detection limits. Concentrations of total triclopyr in fathead minnows (*Pimephales promelas*) reached a maximum of 43.5 g/g on day 0, with residue levels declining exponentially on day 1 (3.82 µg/g) and on day 3 (0.08 µg/g).

In Maine, triclopyr residues were monitored at a commercial spruce-fir operation by Smith et al. (1988). Triclopyr BEE was applied by helicopter at 1.9 kg ae/ha to a regenerating whole-tree clearcut. In the stream section with no buffer, triclopyr residues peaked at 0.056 mg/L the day of spraying. A second peak of 0.048 mg/L was associated with a rain event six days after spraying. Below an unsprayed, forested buffer the maximum concentration of triclopyr in stream water was 0.011 mg/L. The higher maximum concentration of triclopyr observed by Thompson et al. (1991) may be attributed to the higher herbicide application rates in combination with more intensive stream water sampling.

Impacts

Limited toxicological data on invertebrates and fishes indicates Garlon 3A is only slightly toxic to nontoxic (Norris et al. 1991), whereas Garlon 4 is highly toxic to juvenile coho salmon (*Oncorhynchus kisutch*) (Johansen and Green 1990), rainbow trout, and bluegills (*Lepomis macrochirus*) (Norris et al. 1991), but only slightly toxic to aquatic insects (Kreutzweiser et al. 1992).

The maximum concentration of triclopyr BEE (0.35 mg/L) in stream water reported by Thompson et al. (1991) was well above the NOEC calculated by Norris et al. (1991) for rainbow trout (NOEC = 0.074 mg/L) and bluegill (NOEC = 0.087 mg/L). However, the short duration of exposure reduced the likelihood of adverse effects. *In situ* tests with caged yellow perch, fathead minnows, crayfish, and caddis flies generally did not cause mortality. The use of buffer strips should further reduce the likelihood of mortality.

Based on the watershed study done by McKellar et al. (1982) in West Virginia and the risk assessment conducted Norris et al. (1991), concentrations of Garlon 3A typically associated with forestry operations should not approach levels toxic to fish.

Transport

Several studies suggest that triclopyr transport in streams is limited. Smith and McCormack (1988) measured triclopyr concentrations above and below a 450-m-long unsprayed buffer strip. Above the buffer, concentrations in the stream ranged from 0.056 mg/L to below detection (0.005 mg/L), and averaged 0.031 mg/L in the first eight days following application. Below the buffer, however, triclopyr levels were always below detection except for one sample of 0.011 mg/L. Although the relative importance of dilution vs. removal in reducing triclopyr concentrations was not evaluated, triclopyr photodegrades rapidly in water (Woodburn et al. 1993), which should limit transport.

Buffer Strips

Effectiveness of buffer strips in controlling sedimentation is dependent on slope; length of slope (Bauer et al. 1972), soil stability (Fredricksen et al. 1975), type of vegetation present (Schlosser and Karr 1981), condition of fine roots especially three to five years after cutting, and hydraulic conductivity of the soil (Phillips 1989). Lynch and Corbett (1990), working in Pennsylvania, recommend buffer strips at least 1½ times the height of trees. They also found it necessary to maintain buffer strips on intermittent as well as perennial streams. Forest litter protects filter strip soils from erosion (Swift 1986). Roads should be no closer than 100 m from streams (Emaruchi 1991).

The study by Angel (1982) questioned whether a 75-ft stream buffer zone was adequate to prevent sedimentation from reaching streams. Other than problems at stream crossings, all erosion and sedimentation problems arose from sources further than 75 ft from the water body affected. Eighty-eight percent of these originated between 75 and 250 ft from the water body. The majority of erosion problems beyond 250 ft did not result in sedimentation problems (Angel 1982). Other recommended buffer widths are 15 m (Phillips 1989; Nieswand et al. 1990), and 15 to 30-m buffer strips on either side of all streams (Hornbeck et al. 1986).

BMP Compliance Versus Effectiveness

Most of the research dealing with evaluation of BMPs has focused on compliance monitoring and evidence of soil movement. Compliance surveys have been done for several states at varying degrees of intensity (Adams et al. [1995], South Carolina; Brynn and Clausen [1991], Vermont; Schultz et al. [1992], Montana). A survey by Maine DEP for 23 (of a total of 6115) sites harvested in organized towns during 1993 indicated that 30% exhibited serious potential for erosion and sedimentation problems (Maine Department of Environmental Protection 1994a). The sample was small and statistical reliability was not assessed. In general, the results were similar to those obtained in Minnesota, where a sample of 48 harvested sites indicated an overall BMP compliance rate of 79% (Rossman and Phillips 1992). The utility of compliance studies depends directly on the assumption that compliance equates to effectiveness. The studies that have examined sedimentation and BMPs strongly support this assumption. Very few studies have evaluated the relationship between BMP compliance and quality of habitat for aquatic biota, particularly fishes.

The only study that we are aware of linking BMP compliance with biological effectiveness was done in South Carolina by Adams and Hook (1993). BMP compliance was assessed for 177 sites in five categories: road system, road stream crossings, streamside management zones, harvest operations, and log decks. Compliance was rated as excellent, adequate, or inadequate. Each site was subsequently given a pass/fail rating. Of the 57 sites associated with perennial streams, 27 of those that were unaffected by uses other than forestry were sampled intensively for stream habitat assessment. Streams ranged in size from 2-ft wide with 3-ft deep runs to 25-ft wide with 3-ft runs and 4-ft pools. BMP compliance was 84% statewide compared to 67% for the smaller subset selected to include perennial streams for habitat evaluation.

The EPA rapid bioassessment protocol was used by Adams and Hook (1993) for habitat assessment. A numerical rating was based on the following categories: macroinvertebrate habitat availability, cobble embeddedness (sediment interferes with development of young), logging slash in streams (decomposition uses up oxygen), pool riffle/bend ratio, canopy cover, dominant streamside cover, and bank vegetation stability. The downstream score was compared to a reference point above the harvest site or to a similar stream in an unharvested area. Three of the 27 sites were unaffected, scoring 89%-130% of the reference site. Five sites were rated as supporting, scoring 76%-84% of reference, five sites were rated partially supporting (60%-70% of reference site), and five sites were rated as nonsupporting (below 60% of reference values). When BMP compliance was inadequate, sediment and woody debris negatively affected streams. The big effects were due to introduction of large woody debris, increased cobble embeddeness, and loss of macroinvertebrate habitat and riparian canopy.

Examination of the data by redefining unaffected sites as those scoring 77% or better of the reference downstream condition in conjunction with assessing BMP compliance showed that 88% of the sites that were unaffected complied with BMPs. Similarly, for the affected sites, only 20% complied with BMPs. This is the most direct evidence to date that compliance with BMPs is strongly associated with maintenance of stream biologic quality.

Research Needs

Information on the effects of timber harvest on water quality in the Northeast is conspicuous by its absence in several key topic areas: wetlands, boreal forests, and impacts on stream biota. Unlike the Southeast (e.g., Childer and Gosselink 1990) information on the impacts of timber harvest on Northeastern forested wetlands is currently not available. This is particularly noteworthy because forested wetlands constitute 18% of Maine's forest (Maine Department of Environmental Protection 1994a). A study performed in the southern U.S. concluded that due to low relief, wetland surface waters lack the energy to transport sediments and that wetland water quality is less affected by silvicultural practices than upland activity (Shepard 1994). It is reasonable to expect that this may be the case in our region. Information on the effects of timber harvest on water quality in boreal ecosystems is rare, and very few studies have evaluated the effects on forestry related water quality degradation on aquatic biota, particularly fishes.

Finally, effort should be expended on refinement of existing BMPs, many of which date back to the early 1960s and have not been critically analyzed or improved since that time. Brynn and Clausen (1991) pointed out that overly restrictive or unnecessary erosion control measures may cause operators to question BMPs in general. For this reason, it is important to continually question and revise current practices to eliminate those of limited effectiveness and to actively promote those that work.

Literature Cited

Adams, T., and D. Hook. 1993. Implementation and effectiveness monitoring of forestry best management practices on harvested sites in South Carolina. Best Management Practices Monitoring Report No. BMP-1, South Carolina Forestry Commission, Columbia, SC.

Adams, T.O., D.D. Hook, and M.A. Floyd. 1995. Effectivness monitoring of silvicultural best management practices in South Carolina. S.J.A.F. 19:170–176.

Angel, G. 1982. 1980-81 survey of logging operations in the plantations and unorganized townships of Maine. Land Use Regulation Commission, Maine Department of Conservation. Augusta, ME.

Askew, G.R., and T.M. Williams. 1986. Water quality changes due to site conversion in coastal South Carolina. Southern J. Appl. For. 10:134–136.

Askew, G.R., and T.M. Williams. 1984. Sediment concentrations from intensively prepared wetland sites. Southern J. Appl. For. 8(3): 152–157.

Atkinson, D. 1985. The toxilogical properties of glyphosate—a summary. pp. 127–133. *In* E. Grossbard, and D. Atkinson (eds.) The Herbicide Glyphosate. Butterworths, Boston.

Aust, W.M. 1994. Best management practices for forested wetlands in the southern Appalachian region. Water, Air and Soil Pollution 77:457–468.

Baker, J.P., and C.L. Schofield. 1982. Aluminum toxicity to fish in acidic waters. Water, Air, and Soil Pollution 18:289–309.

Bauer, L.D., W.H. Gardner, and W.R. Gardner. 1972. Soil Physics. John Wiley and Sons, New York. 498 p.

Benkobi, L., M.J. Tilica, and J.L. Smith. 1993. Soil loss as affected by different combinations of surface litter and rock. J. Environ. Quality 22(4): 657–661.

Bilby, R.E., and G.E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61(5):1107–1113.

Binkley, D., and T.C. Brown. 1993a. Forest practices as nonpoint sources of pollution in North America. Water Resources Bull. 29(5):729–740.

Binkley, D., and T.C. Brown. 1993b. Management impacts on water quality of forests and rangelands. USDA For. Ser. Gen. Tech. Report RM-239.

Binkley, D., and L.H. MacDonald. 1993. Forests as non-point sources of pollution, and effectiveness of best management practices. Nat. Coun. Air Str. Imp., 260 Madison Ave. New York.

Borg, H., A. Hordacre, and F. Batini. 1988. Effects of logging in stream and river buffers on watercourses and water quality in the southern forests of western Australia. Australian Forestry. 51(2): 98–105.

Bormann, F.H., G.E. Likens, T.G. Siccama, R.S. Pierce, and J.S. Eaton. 1974. The export of nutrients and the recovery of stable conditions following deforestation at Hubbard Brook. Ecological Monographs 44(3):255–277.

Borne, J. 1979. A survey of erosion and sedimentation problems associated with logging in Maine. A Water Quality Planning Report Submitted to the Maine Department of Environmental Protection. Maine Land Use Regulation Commission. Augusta, ME.

Bronstad, J.O. and H.O. Friestad. 1985. Behaviour of glyphosate in the aquatic environment. pp. 200–205. In E. Grossbard and D. Atkinson (eds.) The Herbicide Glyphosate. Butterworths, Boston.

Brown, G.W. 1985. Forestry and Water Quality. Oregon State University Book Stores, Inc. Corvallis, Oregon.

Briggs, R.D., and J.W. Hornbeck. 1995. Impacts of precommercial thinning on surface water chemistry and crop tree growth at Weymouth Point. Final Report to USDA CSRS. In preparation.

Brynn, D.J., and J.C. Clausen. 1991. Postharvest assessment of Vermont's silvicultural management practices and water quality impacts. Northern J. of Appl. For. 8:140–144.

Buckler, D.R., P.M. Mehrle, L. Cleveland, and F.J. Dwyer. 1987. Influence of pH on the toxicity of aluminum and other inorganic contaminants to East Coast striped bass. Water, Air, and Soil Pollution 35:97–106.

Burton, T.M., and G.E. Likens. 1973. The effect of strip-cutting on stream temperatures in the Hubbard Brook Experimental Forest, New Hampshire. BioScience 23:433–435.

Camargo, J.A., and J.V. Ward. 1992. Short-term toxicity of sodium nitrate ($NaNO_3$) to nontarget freshwater invertebrates. Chemosphere 24:23–28.

Chapman, P.M. 1989. Salmonid toxicity studies with Roundup. *In*: P.E. Reynolds (ed) Proceedings of the Carnation Creek Herbicide Workshop. Forest Pest Management Institute, Forestry Canada, Saulte Ste. Marie, Ontario P6A 5M7. 349 p.

Childers, D.L., and J.G. Gosselink. 1990. Assessment of cumulative impacts to water quality in a forested wetland landscape. J. of Environ. Quality 19:455–464.

Comes, R.D., V.F. Bruns, and A.D. Kelley. 1976. Residues and persistence of glyphosate in irrigation water. Weed Science 24:47–50.

Commerford, N.B., D.G. Neary, and R.G. Mansell. 1992. The effectiveness of buffer strips for ameliorating offsite transport of sediment, nutrients, and pesticides from silvicultural operations. Nat. Coun. Air Str. Impr. Tech. Bull. No. 631.

Cook, M.J., and J.G. King. 1983 Construction costs and erosion effectiveness of filter windrows on fill slopes. USDA, Forest Service Research Note INT-335, Intermountain Forest and Range Experiment Station, Ogden, UT, 5 p.

Corbett, J.A., and E.S. Lynch. 1985. Management of streamside zones on municiple watersheds, p. 187-190. *In*: Riparian ecosystems and their management: reconciling conflicting uses. First Amer. Riparian Conf. USDA Gen. Tech. Report RM-120.

DeHart, D.B. 1982. The effects of timber harvesting on erosion and sedimentation in New Hampshire. New Hampshire Division of Forests & Lands, Department of Resources & Economic Development.

DePinto, J.V., T.C. Young, and S.C. Martin. 1981. Algal-available phosphorus in suspended sediments from lower Great Lakes tributaries. J. Great Lakes Research 7:311–325.

Dorich, R.A., D.W. Nelson, and L.E. Sommers. 1985. Estimating algal available phosphorus in suspended sediments by chemical extraction. J. Environ. Quality 14:400–405.

Driscoll, C.T. 1985. Aluminum in acidic surface waters: chemistry, transport, and effects. Environmental Health Perspectives 63:93–104.

Elliot, J.M. 1991. Tolerance and resistance to thermal stress in juvenile Atlantic salmon, *Salmo salar*. Freshwater Biology 25:61–70.

Emaruchi, B. 1991. Riparian strip widths for sediment control. Faculty of Forestry, University of New Brunswick. M.S. Thesis.

Faafeng, B.A., and R. Roseth. 1993. Retention of nitrogen in small streams artificially polluted with nitrate. Hydrobiologia 251:113–122.

Farrish, K.W., J.C. Adams, and C.V. Thompson. 1993. Soil conservation practices on clearcut forests in Louisianna. J. Soil and Water Conservation 48(2): 136–139.

Feller, M.C. 1989. Effects of forest herbicide applications on streamwater chemistry in southwestern British Columbia. Water Resources Bulletin 25:607–616.

Feng, J.C., D.G. Thompson, and P.E. Reynolds. 1989. Fate of glyphosate in a Canadian forest stream ecosystem. pp. 45–64. *In:* P.E. Reynolds (ed) Proceedings of the Carnation Creek Herbicide Workshop. Forest Pest Management Institute, Forestry Canada, Saulte Ste. Marie, Ontario P6A 5M7.

Feng, J.C. and D.G. Thompson. 1989. Persistance and dissipation of glyphosate in foliage and soils of a Canadian coastal forest watershed. pp. 65–87. *In:* P.E. Reynolds (ed) Proceedings of the Carnation Creek Herbicide Workshop. Forest Pest Management Institute, Forestry Canada, Saulte Ste. Marie, Ontario P6A 5M7.

Field, D.B. 1995. Micro-IMPLAN analysis of 1990 data. Unpublished report, Dept. of Forest Management, Univ. of Maine, Orono, Feb. 10.

Folmar, L.C., H.O. Sanders, and A.M. Julin. 1979. Toxicity of the herbicide glyphosate and several of its formulations to fish and aquatic invertebrates. Archives of Environ. Contamin. and Toxicol. 8:269–278.

Foster, D.R. 1988. Species and stand response to catastrophic wind in central New England, U.S.A. J. Ecol. 76:135–151.

Fredricksen, R.L., Moore, D.G., and L.H. Norris. 1975. Impact of timber harvest, fertilization and herbicide treatment on streamwater chemistry in the Douglas-fir region, pp. 283–313. *In*, B. Bernier and C.H. Winget (eds.) Forest Soils and Land Management. Proc. Fourth N.A. Forest Soils Conf., Les Presses de Universite Laval, Quebec.

Gagen, C.J., W.E. Sharpe, and R.F. Carline. 1994. Downstream movement and mortality of brook trout *(Salvelinus fontinalis*) exposed to acidic episodes in streams. Can. J. of Fish. and Aquatic Sci. 51:1620–1628.

Garman, G.C. 1984. Initial effects of deforestation on community structure and function of the East Branch Piscataquis River, Maine. Ph.D. Thesis. Department of Zoology, University of Maine.

Garman, G.C., and J.R. Moring. 1991. Initial effects of deforestation on physical characteristics of a boreal river. Hydrobiologia 209:29–37.

Garman, G.C., and J.R. Moring. 1987. Impacts of sedimentation on aquatic systems: a literature review and proposed research agenda. Report prepared for the Maine Land Use Regulation Commission, Committee on Stream Research.

Gilliam, J.W. 1994. Riparian wetlands and water quality. J. Environ. Quality 23(5): 896–900.

Grant, J.W., J. Englert, and B.F. Bietz. 1986. Application of a method for assessing the impact of watershed practices: effects of logging on salmonid standing crops. North American J. Fish. Man. 6:24–31.

Grossbard, E., and D. Atkinson (eds.) 1985. The herbicide glyphosate. Butterworths, Boston. 490 p.

Haines, T.A., S.A. Norton, J.S. Kahl, C.W. Fay, S.J. Pauwels, and C.H. Jagoe. 1990. Intensive studies of stream fish populations in Maine. U.S. Environmental Protection Agency, EPA/600/3-90/043.

Haines, T.A., and J. Akielaszek. 1983. A regional survey of chemistry of headwater lakes and streams in New England: vulnerability to acidification. U.S. Fish and Wildlife Service, FWS/OBS-80/40.15.

Haines, T.A., and J.J. Akielaszek. 1984. Effects of acidic precipitation on Atlantic salmon rivers in New England. U.S. Fish and Wildlife Service, FWS/OBS-80/40.18.

Hartung, R.E., and J.M. Kress. 1977. Woodlands of the northeast erosion and sediment control guides. USDA Soil Conservation Service Northeast Tech. Service Center, Broomall PA. 26 p.

Hecky, R.E., and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment. Limnology and Oceanography 33:796–822.

Hendrey, G.R. 1987. Acidification and anadromous fish of Atlantic estuaries. Water, Air, and Soil Pollution 35:1–6.

Hobbie, J.E., and G.E. Likens. 1973. Output of phosphorus, dissolved organic carbon, and fine particulate carbon from Hubbard Brook watersheds. Limnology and Oceanography 18:734–742.

Hornbeck, J.W., C.W. Martin and C.T. Smith. 1986. Protecting forest streams during whole-tree harvesting. Northern J. Appl. For. 3:97–100.

Hornbeck, J.W., C.W. Martin, R.S. Pierce, F.H. Bormann, G.E. Likens, and J.S. Eaton. 1987. The northern hardwood forest ecosystem: ten years of recovery from clearcutting. USDA Forest Service, Research Paper NE-RP-596.

Ingersoll, C.G., D.R. Mount, D.D. Gulley, T.W. LaPoint, and H.L. Bergman. 1990. Effects of pH, aluminum, and calcium on survival and growth of eggs and fry of brook trout (*Salvelinus fontinalis*). Can. J. of Fish. and Aquatic Sci. 47:1580–1592.

Jagoe, C.H., and T.A. Haines. 1990. Morphometric effects of low pH and limed water on the gills of Atlantic salmon (*Salmo salar*). Can. J. Fish. and Aquatic Sci. 47:2451–2460.

Johansen, J.A., and G.H. Green. 1990. Sublethal and acute toxicity of the ethylene glycol butyl ether ester formulation of triclopyr to juvenile coho salmon (*Oncorhyncus kisutch*). Archives of Environ. Contamin. and Toxicol. 19:610–616.

Kennish, M.J. 1992. Ecology of Estuaries: Anthropogenic Effects. CRC Press, Boca Raton, FL.

Kincheloe, J.W., G.A. Wedemeyer, and D.L. Koch. 1979. Tolerance of developing salmonid eggs and fry to nitrate exposure. Bull. Environ. Contamin. and Toxicol. 23:575–578.

King, J.G. 1989. Stream flow responses to roadbuilding and harvesting: a comparison with the equivalent clearcut area procedure. USDA For. Ser. Res. Pap. INT-401.

Knighton, M.D., and J.H. Stiegler. 1981. Phosphorus release following clearcutting of a black spruce fen and a black spruce bog. *In*: Proceedings of the Sixth International Peat Congress, August 17–23, 1980, Duluth, Minnesota.

Krause, H.H. 1982. Nitrate formation and movement before and after clear-cutting of a monitored watershed in central New Brunswick, Canada. Can. J. For. Res. 12:922–930.

Kreutzweiser, D.P., S.B. Holmes, and D.J. Behmer. 1992. Ecotoxicol. and Environ. Safety 23:364–374.

Lawrence, G.B., R.D. Fuller, and C.T. Driscoll. 1987. Release of aluminum following whole-tree harvesting at the Hubbard Brook Experimental Forest, New Hampshire. J. Environ. Quality 16:383–390.

Likens, G.E., F.H. Bormann, N.M. Johnson, D.W. Fisher and R.S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecological Monographs 40:23–47.

Lynch, J.A., E.S. Corbett, K. Mussallem. 1985. Best management practices for controlling nonpoint-source pollution on forested watersheds. J. Soil and Water Conser. 40:164–167.

Lynch, J.A., and E.S. Corbett. 1990 Evaluation of best management practices for controlling nonpoint pollution from silvicultural operations. Water Resources Bulletin 26(1):41–52.

Lyons, J.K., and R.L. Beschta. 1983. Land use, floods, an channel changes: Upper Middle Fork Willamette River, Oregon (1936–1980). Water Resources Research 19:463–471.

MacDonald, L.H., A.W. Smart, and R.C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. U.S. Environmental Protection Agency, EPA/910/9-91-001.

Maine Bureau of Forestry. 1991. Erosion and sediment control handbook for Maine timber harvesting operations: best management practices. Augusta. 49 p.

Maine Department of Environmental Protection. 1987. Maine lakes classification project, report on vulnerability index. EPA Grant 3 66.405.

Maine Department of Environmental Protection. 1994a. State of Maine 1994 Water Quality Assessment.

Maine Department of Environmental Protection. 1994b. Appendices to the State of Maine 1994 Water Quality Assessment.

Maine Forest Service. 1994. Erosion control handbook for Maine timber harvesting operations best management practices. Department of Conservation Maine Forest Service, Augusta.

Martin, C.W., D.S. Noel, and C.A. Federer. 1984. Effects of forest clearcutting in New England on stream chemistry. J. Environ. Quality 13:204–210.

Martin, C.W., and J.W. Hornbeck. 1994. Logging in New England need not cause sedimentation of streams. Northern J. App. For. 11:17–23.

Martin, C.W., and R.S. Pierce. 1980. Clearcutting patterns effect nitrate and calcium in streams of New Hampshire. J. For. 78:268–272.

Martin, C.W., R.S. Pierce, G.E. Likens, and F.H. Bormann. 1986. Clearcutting affects stream chemistry in the White Mountains of New Hampshire. USDA Forest Service, Research Paper NE-579.

McKellar, R.L., O.E. Schubert, B.C. Byrd, L.P. Stevens, and E.J. Norton. 1982. Aerial application of Garlon 3A herbicide to a West Virginia watershed. Down to Earth 38:15–19.

Megahan, W.F. 1981. Effects of silvicultural practices on erosion and sedimentation in the interior west a case for sediment budgeting. *In* D.M. Baumgartner (ed.) Interior West Watershed Management. Symposium Proc., April 8–10, 1980. Spokane, Washington.

Megahan, W.F., S.B. Monsen, M.D. Wilson, N. Lozano, D.F. Haber and G.D.Booth. 1992. Erosion control practices applied to granitic roadfills for forest roads in Idaho: Cost effectiveness evaluation. Land Degradation and Rehabilitation 3:55–65.

Moore, T.R. 1987. A preliminary study of the effects of drainage and harvesting on water quality in ambrotrophic bags near Sept-Isles, Quebec. Water Resour. Bull. 23:1–7.

Mullen, D.M., and J.R. Moring. 1988. Partial deforestation and short-term autochthonors energy input to a small New England stream. Water Resources Bulletin 24:1273–1279.

Newton, M., K.M Howard, B.R. Kelpsas, R. Danhaus, C.M. Lottman, and S. Dubelman. 1984. Fate of glyphosate in an Oregon forest ecosystem. J. Agric. and Food Chem. 32:1144–1151.

Nicolson, J.A. 1988. Alternate strip clearcutting in upland black spruce V. The impact of harvesting on the quality of water flowing from small basins in shallow-soil boreal ecosystems. For. Chron. 64:52–58.

Nieswand, G.H., R.M. Hordon, T.B. Shelton, B.B. Chavooshian, and S. Blarr. 1990. Buffer strips to protect water quality reservoirs: a model and recommendations. Water Resources Bulletin 26(6): 959–966.

Noel, J., J. Dennis, M.E. Dennis, C. Kuhns, D. Miller, and C. Eliot. 1992. Phosphorus control in lake watersheds: a technical guide to evaluating new development. Maine Department of Environmental Protection, Augusta.

Noel, D.S., C.W. Martin, and C.A. Federer. 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. Environ. Manage. 10:661–670.

Norris, L.A., H.W. Lorz, and S.V. Gregory. 1991. Forest chemicals. American Fisheries Society Special Publication 19:139–179.

Omernik, J.M. 1976. The influence of land use on stream nutrient levels. U.S. Environmental Protection Agency, EPA-600/3-76-014.

Patric, J.H. 1976. Soil erosion in the eastern forest. J. For. 74(10): 671–677.

Patric, J.H., J.O. Evans, and J.D. Helvey. 1984. Summary of sediment yield data from forested land in the United States. J. For. 82:101–104.

Payne, N., J.C. Feng, and P.E. Reynolds. 1989. Off-target deposit measurements and buffer zones required around water for various aerial applications of glyphosate. pp. 88–109. *In*: P.E. Reynolds (ed) Proceedings of the Carnation Creek Herbicide Workshop.

Forest Pest Management Institute, Forestry Canada, Saulte Ste. Marie, Ontatio P6A 5M7. 349 p.

Payne, N.J., J.C. Feng, and P.E. Reynolds. 1990. Off-target deposits and buffer zones required around water for aerial glyphosate applications. Pesticide Science 30:183–198.

Perrin, C.J., K.S. Shortreed, and J.G. Stockner. 1984. An integration of forest and lake fertilization: transport and transformations of fertilizer elements. Can. J. Fish. and Aquatic Sci. 41:253–262.

Peterjohn, W.T., and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: obsevations on the role of a riparian forest. Ecology 65(5): 1466–1475.

Peterson, R.H., R.A. Bourbonniére, G.L. Lacroix, D.J. Martin-Robichaud, P. Takats, and G. Brun. 1989. Responses of Atlantic salmon (*Salmo salar*) alevins to dissolved organic carbon and dissolved aluminum at low pH. Water, Air, and Soil Pollution 46:399–413.

Phillips, J.D. 1989 An evaluation of the factors determining the effectiveness of water quality buffer zones J. Hydrology 107:133–145.

Pierce, R.S., J.W. Hornbeck, C.W. Martin, L.M. Tritton, C.T. Smith, C.A. Federer, and H.W. Yawney. 1993. Whole-tree clearcutting in New England: manager's guide to impacts on soils, streams, and regeneration. USDA Forest Service, General Technical Report NE-172.

Reinhart, K.G., and A.R. Eschner. 1962. Effect on streamflow of four different forest practices in the Allegheny Mountains. J. Geophysical Research 67:2433–2445.

Rosseland, B.O., O.K. Skogheim, F. Kroglund, and E. Hoell. 1986. Mortality and physiological stress of year-classes of landlocked and migratory Atlantic salmon, brown trout and brook trout in acidic aluminum-rich soft water. Water, Air, and Soil Pollution 30:751-756.

Rueppel, M.L., B.B. Brightwell, J. Schaefer, and J.T. Marvel. 1977. Metabolism and degradation of glyphosate in soil and water. J. Agriculture and Food Chemistry 25(3): 517–528.

Satterlund, D.R., and P.W. Adams. 1992. Erosion and sedimentation. Chapter 10, p. 186–225. *In:* 1992. Wildland watershed management. Second Edition. John Wiley and Sons.

Schlosser, I.J., and J.R. Karr. 1981. Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. Environ. Manage. 5:233–243.

Schultz, S., R. Lincoln, J. Cauhorn, and C. Montagne. 1992. Quantification of erosion from a fire and subsequent rainfall event in the northern Rocky Mountains. Proc. Montana Academy of Sci. 52:143–152.

Seymour, R.S., and R.C. Lemin. 1989. Timber supply projections for Maine, 1980–2080. Maine Agric. Expt. Sta. Misc. Rept. 337. 39 p.

Shepard, J.P. 1994. Effects of forest management on surface water quality in wetland forests. Wetlands 14(1): 18–26.

Shepard, J.P., C.A. Nowak, D.C. LeBlanc, R.D. Briggs, and R.B. Downard. 1990. Influences of acidic deposition and forest development on conifers at Pack Forest, New York. p. 264–289. *In*: S.P. Gessel et al. (ed.) Sustained Productivity of Forest Soils. Proceedings of the 7th North American Forest Soils Conference, University of British Columbia, Faculty of Forestry Publication, Vancouver, BC.

Shipp, A.M., M.L. Hogg, K.S. Crump, and R.L. Kodell. 1986. Worst case analysis study on forest plantation herbicide use. Prepared for Forest Land Management Division, Department of Natural Resources, State of Washington by K.S. Crump and Company, Inc., Ruston, LA 71270. 367 p.

Smith, C.T., and M.L. McCormack, Jr. 1988. Watershed losses of triclopyr after aerial application to release spruce-fir. Proceedings Northeastern Weed Science Soc. 42:104–108.

Smith, C.T., J.W. Hornbeck, and M.L. McCormack, Jr. 1988. Changes in nutrient cycling following aerial application of triclopyr to release spruce-fir. Proceedings Northeastern Weed Science Soc. 42:94–99.

Society of American Foresters. 1995. Reauthorization of the clean water act. Forestry effects on water quality. SAF Task Force on reauthorization of the clean water act. Society of American Foresters, SAF 95-01. 24 p.

Stolz, J., and J. Schnell. 1991. Trout. Stackpole Books. Harrisburg, PA.

Sullivan, D.S., and T.P. Sullivan. 1993. Non-target impacts of the herbicide glyphosate (3rd ed) A compendium of references and abstracts. Mammal Pest Management Ltd., Langley, B.C. Canada V3A 4R1. 138 p.

Sullivan, K., J. Tooley, K. Doughty, J.E. Caldwell, P. Knudsen. 1990. Evaluation of prediction models and characterization of stream temperature regimes in Washington. Timber/Fish/Wildlife Report #TFW-WQ3-90-006. Washington Department of Natural Resources, Olympia.

Swift, L.W. 1984. Gravel and grass surfacing reduces soil loss from mountain roads. For. Sci. 30:657–670.

Swift, L.W. 1986. Filter strip widths for forest roads in the southern Appalachians. Southern J. Appl. For. 10(1): 27–34.

Taylor, P.N. 1989. Effects of controlled sediment additions on macroinvertebrates and water quality in small Maine streams. M.S. Thesis. Department of Entomology, University of Maine, Orono, Maine.

Thompson, D.G., B. Stanzik, D.D. Fontaine, T. Mackay, G.R. Oliver, and J. Troth. 1991. Fate of triclopyr ester (Release®) in a boreal forest stream. Environ. Toxicol. and Chem. 10:619–632.

Tooby, T.E. 1985. Fate and biological consequences of glyphosate in the aquatic environment. pp. 206–217. *In:* E. Grossbard, and D. Atkinson (eds.) The Herbicide Glyphosate. Butterworths, Boston.

Tortennson, L. 1985. Behaviour of glyphosate in soils and its degradation. pp. 137–150. *In:* E. Grossbard, and D. Atkinson (eds.) The Herbicide Glyphosate. Butterworths, Boston.

Trettin, C.C., M.F. Jurgensen, J.W. McLaughlin, and M.R. Gale. 1996. Functional responses to harvesting and site preparation in a coniferous swamp. *In:* Trettin, C.C. et al. (eds), Ecology and Management of Northern Forested Wetlands. Lewis Press, Boca Raton, FL. *In Press*.

Verhoff, F.H., D.A. Melfi, and S.M. Yaksich. 1979. Storm travel distance calculations for total phosphorus and suspended materials in rivers. Water Resources Research 15:1354–1360.

Verry, E.S. 1986. Forest harvesting and water: the lake states experience. Water Resources Bulletin 22:1039–1047.

Wallace, J. 1993. Environmental Considerations. *In*: K. Heen, R.L. Monahan, and R. Utter, eds. Salmon Aquaculture. Halsted Press, New York.

Watson, J.S., and A.T. Der. 1986. Turbidity: a literature review of its impacts on aquatic resources. Maryland Department of Health and Mental Hygiene, Office of Environmental Programs.

Wetzel, R.G. 1975. Limnology. W.B. Saunders Company, Philadelphia, PA.

White, J.B., and H.H. Krause. 1993. The impact of forest management on water quality and the establishment and management of protective buffer zones. Department of Forest Resources, University of New Brunswick.

Williams, T.M. 1994. Water quality changes associated with forest drainage. *In:* Water Management in Forested Westlands. Proc. Workshop on Water Management in Forested Wetlands. April 26–28, 1994, Atlanta, GA. USDA-Forest Service Techn. Rep. No. R8-TP20. 76–83 pp.

Wood, T., F.H. Bormann, and G.K. Voigt. 1984. Phosphorus cycling in a northern hardwood forest: biological and chemical control. Science 223:391–393.

Woodburn, K.B., F.R. Batzer, R.H. White, and M.R. Schultz. 1993. The aqueous photolysis of triclopyr. Environ. Toxicol. and Chemin. 12:43–55.

Zuzel, J.F., and J.L. Pikul. 1993 Effects of straw mulch on runoff and erosion from small agricultural plots in northeastern Oregon. Soil Science 156(2):111–117.