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NATIONAL COUNCIL OF THE PAPER INDUSTRY FOR AIR AND STREAM IMPROVEMENT, INC., 260 MADISON AVENUE, NEW YORK, N.Y. 10016

FORESTS AS NONPOINT SOURCES OF POLLUTION, AND EFFECTIVENESS OF BEST MANAGEMENT PRACTICES

TECHNICAL BULLETIN NO. 672

JULY 1994

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Significant progress has been made in reducing the impacts of forest management operations on water quality and fish habitat. The National Council and its member companies have played important roles in the development, testing, and implementation of effective Best Management Practices (BMPs) for water quality protection in managed forests.

Nevertheless, the public remains concerned about effects of forest management on water resources. States participating in the federal Coastal Zone program are in the process of developing new nonpoint source control programs for forestry and other activities. Proposed amendments to the federal Clean Water Act would impose significant new requirements on forest management operations and other nonpoint sources throughout the country.

This report reviews the literature on (a) the quality of water in forests, (b) effects of forest management on water quality, and (c) the effectiveness of BMPs for water quality protection. It is shown that the quality of water draining from forested watersheds is normally the best in the nation (relative to other land uses) whether the forests are left untouched or intensively managed. Forestry is a relatively small contributor to the nation's overall nonpoint source problem in terms of both quantity and quality of discharge. Local and watershed-scale impacts on water quality can be severe, but these can be controlled effectively in most circumstances by use of basic BMPs such as stream bank protection. Additional control measures beyond the basic BMPs may be needed where there is greater than normal risk of water quality impact.

Sedimentation is the most important water quality problem associated with forest management operations. As might be expected, many published studies were conducted in areas with steep terrain where sedimentation was perceived to be a problem. Studies showing absence of sedimentation problems in gentle terrain are relatively scarce, perhaps because investigators and funding agencies have allocated limited resources to other topics. With very few exceptions, nutrients and pesticides have not been associated with water quality problems in managed forests.

This report was prepared by Dr. Dan Binkley and Dr. Lee MacDonald of Colorado State University. It is the second in a series of forestry-related bulletins developed to provide an information base for use by the industry during Clean Water Act reauthorization. The first in the series was Technical Bulletin No. 660 "*Benefits and Costs of Programs for Forestry Nonpoint Pollution Control in Washington and Virginia*" (April 1994).

Your comments and suggestions on this report are solicited. They should be directed to Dr. Alan A. Lucier, Program Director – Forest Environmental Studies, at this office (212/532-9251 ext. 237).

Very truly yours,

Isaiah Gellman

Dr. Isaiah Gellman
President

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FORESTS AS NONPOINT SOURCES OF POLLUTION,
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ABSTRACT: This report synthesizes available knowledge on the effects of forest practices on water quality by examining six propositions:

(1) *The quality of water draining forested watersheds is normally the best in the nation whether the forests are left untouched or intensively managed.* Available information supports this proposition.

(2) *Properly implemented Best Management Practices (BMPs) effectively control nonpoint sources of pollution from forestry operations under most circumstances.* This proposition is generally true, but in some unstable areas conventional BMPs for road construction and forest harvest may not be sufficient to prevent adverse effects on stream channels and fish habitat.

(3) *Effects of present day management activities on water quality are usually transient and are rarely severe enough to cause significant damage to fish populations.* Negative impacts of forest practices should be uncommon where conventional BMPs are used. Exceptions to this conclusion would include (a) unstable areas or areas with highly erodible soils; (b) the combination of management activities with extreme storm events and (c) possibly downstream depositional areas where there is potential for cumulative effects.

(4) *The potential for water quality impacts from forest management depends upon factors such as climate, soil types, topography, etc. Special control measures can be targeted where there is greater than normal risk for short- or long-term adverse effects (such as areas with high risk for landslides).* This proposition is generally true, although we note that extreme storm events can have unavoidable effects on both managed and unmanaged landscapes and streams. The cost of special control measures may in some situations be prohibitive relative to the value of the reduced risk of water quality degradation.

(5) *The water draining forest watersheds is not "toxic" (as defined by the Clean Water Act and elsewhere) unless affected by spills or other unusual events.* Water quality issues for forest streams do not involve "toxins," with the possible exceptions of high nitrate concentrations and high aluminum concentrations. Both of these water quality parameters very rarely exceed toxic thresholds in forest streams.

(6) *The direct effects of forest practices on water quality within a small watershed may be different than the cumulative effects of the same practices on many small watersheds within a basin.* This proposition may be generally true, but very little comprehensive information is available. Use of conventional BMPs at the site level to maintain water quality may be effective at minimizing cumulative effects within a basin. More research is needed to fully evaluate this proposition.

KEYWORDS: Water quality, nonpoint source pollution, cumulative effects, impacts of forest practices

RELATED NCASI PUBLICATIONS:

"*Benefits and Costs of Programs for Forestry Nonpoint Pollution Control in Washington and Virginia*," NCASI Technical Bulletin No. 660 (April 1994).

"*Status of NCASI Cumulative Watershed Effects Program and Methodology*," NCASI Technical Bulletin No. 634 (June 1992).

"*The Effectiveness of Buffer Strips for Ameliorating Offsite Transport of Sediment, Nutrients, and Pesticides from Silvicultural Operations*," NCASI Technical Bulletin No. 631 (June 1992).

"*The Use of Environmental Audits and Assessments by the Forest Products Industry for Timberland Operations*," NCASI Technical Bulletin No. 574 (October 1989).

"*Long-Term Broad-Scale Water Quality Planning and the Use of Environmental Audits in Forest Management Programs*," NCASI Technical Bulletin No. 541 (February 1988).

"*Procedures for Assessing the Effectiveness of Best Management Practices in Protecting Water and Stream Quality Associated with Managed Forests*," NCASI Technical Bulletin No. 538 (January 1988).

"*Industry, State, and Federal Programs Designed to Assess and Protect Water Quality Associated with Managed Western Forests*," NCASI Technical Bulletin No. 466 (July 1985).

FORESTS AS NONPOINT SOURCES OF POLLUTION,
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I INTRODUCTION

Forests cover about 1/3 of the conterminous United States, and most of the headwaters of major rivers and streams arise in forested catchments. There is considerable public concern over the effects of forest management on water quality, and this stems in large part from perceptions that:

- forests yield water of exceptional quality;
- forested watersheds are major sources of high-quality domestic water supplies; and
- populations of desired fish species depend on high-quality water in forests.

Most forest management activities are considered potential nonpoint sources of water pollution, and nonpoint sources are now regarded as the primary cause of degraded water quality in the U.S. (USEPA 1990). Although forestry is not as important a source of nonpoint pollution as other land uses, such as agriculture (USEPA 1990), it attracts a large share of the public interest.

In this paper, we synthesize available knowledge on the effects of forest practices on water quality. This synthesis examines six basic propositions:

(1) The quality of water draining forested watersheds is normally the best in the nation whether the forests are left untouched or intensively managed.

(2) Properly implemented Best Management Practices (BMPs) effectively control nonpoint sources of pollution from forestry operations under most circumstances.

(3) Effects of present day management activities on water quality are usually transient and are rarely severe enough to cause significant damage to fish populations.

(4) The potential for water quality impacts from forest management depends upon factors such as climate, soil types, topography, etc. Special control measures can be targeted where there is greater than normal risk for short-term or long-term adverse effects (such as areas with high risk for landslides).

(5) The water draining forest watersheds is not "toxic" (as defined by the Clean Water Act and elsewhere) unless affected by spills or other unusual events.

(6) The direct effects of forest practices on water quality within a small watershed may be different than the cumulative effects of the same practices on many small watersheds within a basin.

These propositions are examined in relation to the following water quality indicators: temperature, sediment, dissolved oxygen, nutrient concentrations, pesticide concentrations, stream channel conditions, and populations of fish and invertebrates.

II BACKGROUND ON WATER QUALITY PARAMETERS AND INDICATORS

A. Temperature

Stream temperatures depend on a variety of energy transfer processes, including incoming radiation, evaporation, and the temperature of water entering streams from subsurface flow (Beschta *et al.* 1987, Sullivan *et al.* 1990). Seasonal variations in stream temperatures generally follow the seasonal pattern of air temperature, but with a lag of a month or so. The metabolic rate of fish and other stream organisms may double with a rise in temperature from 5 to 15°C, while the same change will reduce the saturation concentration for dissolved oxygen by about 20%. Forest practices influence stream temperatures primarily by altering the canopy cover that intercepts solar radiation, and secondarily by altering: (a) the size and shape of stream channels, and (b) the volume of low flows.

B. Sediment

Erosion and sedimentation are major concerns in forest management. Large inputs of fine sediments can impair habitat for fish and other stream organisms. Mechanisms of impact may include (a) lower permeability of streambed gravels (which reduces water/gas exchange and can prevent winter refuge) (Chapman and McLeod 1987, MacDonald *et al.* 1991); (b) burying of gravels (which can inhibit or prevent the movement of organisms and materials between the stream channel and the hyporheic or river-influenced groundwater zone) (Stanford and Ward 1992); and (c) filling of pools (with loss of rearing habitat for fish) (Bisson *et al.* 1992b). Sediments also contain nutrients and organic matter, and can affect rates of in-stream ecological processes such as oxygen consumption.

Direct measures of stream sediments include turbidity (light absorbing and scattering properties), concentrations of suspended sediment (typically in mg of suspended particles per liter), and bedload (material rolled along the streambed). Indirect measures (related primarily to bedload sediment) include channel characteristics such as the ratio of stream width to depth, bank erosion, depth or volume of pools, pool filling by fine sediments, and the particle size distribution of streambed materials (Lisle 1982, Lisle and Hilton 1992). The impacts of increased stream sediments vary greatly among locations and situations.

C. Dissolved Oxygen

Fish and many other stream organisms rely on dissolved oxygen for metabolism. The maximum concentration of dissolved oxygen depends on temperature and air pressure (elevation). At sea level, water at 5°C can contain up to 12.8 mg of dissolved oxygen per liter, compared with 10.1 mg/L at 10°C. Every 1000 m increase in elevation decreases oxygen solubility by about 12% (Golterman *et al.* 1978). The actual concentration of oxygen in streamwater depends also on the rate of oxygen depletion from chemical processes, respiration by animals and micro-organisms, and oxygen replenishment by both aquatic plant photosynthesis and gas exchange with the atmosphere. In general, concentrations of oxygen greater than 8 mg/L are optimal (USEPA 1986, Chapman and McLeod 1987).

D. Nutrient Concentrations

Increases in nutrient concentrations in streams can increase stream productivity, increase diurnal fluctuations in stream oxygen concentrations, and decrease or increase species diversity. In some cases, increases in nutrient concentrations may increase fish production by increasing the food base. The effects and desirability of an increase in nutrient concentrations depend on the status of other limiting factors, the magnitude of the nutrient change, and the desired characteristics of the affected and downstream water bodies. Nitrogen (usually as nitrate) and phosphorus (as phosphate) are the elements that most often affect ecological processes in streams and lakes. High levels of nitrate degrade the quality of water for human consumption; the recommended water quality criterion is 10 mg of nitrogen as nitrate per liter (USEPA 1986).

E. Pesticide Concentrations

In forests, herbicides and pesticides are applied to control undesirable plants, insects, and microorganisms. Some of these chemicals may reach streams either through direct deposition from aerial applications, by surface runoff, or by subsurface flow. Pesticides are used infrequently on any given area, and affect a very small portion of forest lands in any given year. The impacts of pesticides at concentrations typically observed in streams are thought to be very low (Norris *et al.* 1991, Oregon Dept. of Forestry 1992).

F. Stream Channel Conditions

Forestry operations may alter the characteristics of stream channels through direct inputs of sediments in run-off and landslides, through direct encroachment, by altering the long-term recruitment of large woody debris into the stream channel, through reduction in the integrity of streambanks, and through indirect effects of altered hydrology patterns (particularly the size and duration of peak flows). To the best of our knowledge there are no state water quality criteria for stream channel conditions, but changes in some of these parameters may impair water quality by harming designated beneficial uses such as coldwater fisheries. Many National Forests use some index of stream channel condition to indicate the scope for continued management (Parrott *et al.* 1989).

G. Populations of Fish and Invertebrates

In general, forestry operations do not reduce water quality enough to warrant concern from the standpoint of human water consumption. Forest management activities may have substantial effects on aquatic biota, principally through increased sediment loading. Macroinvertebrates are particularly susceptible, and they are both a major food source for fish and a controlling factor on other key processes such as litter decomposition. Monitoring macroinvertebrates may provide a sensitive indicator of changing water quality, but high variation in macroinvertebrate populations over time and across locations means that small changes are difficult to detect and a substantial amount of background information is needed before unusual changes can be discerned (MacDonald *et al.* 1991, Rosenberg and Resh 1992). Coldwater fisheries are often regarded as the designated beneficial use most sensitive to forest management activities, and these fisheries are of great public concern and economic importance.

III DISCUSSION OF PROPOSITIONS

Proposition 1: The quality of water draining forested watersheds is normally the best in the nation whether the forests are left untouched or intensively managed.

A. Background

A wide range of studies have validated this proposition (summarized in Binkley and Brown 1993). In the 1982 National Fisheries Survey (Judy *et al.* 1984), silvicultural activity was estimated to produce adverse effects on fish in about 7.5% of the river and stream kilometers that were assessed, compared with

29.5% for agricultural lands, and 6.7% for cities. In the 1988 Water Quality Survey (USEPA 1990), silvicultural activities were estimated to reduce water quality in 2.4% of river and stream kilometers that were assessed, compared with 15.3% for agriculture.

Omernik (1977) estimated the concentrations of total N and P in streams draining large areas of differing land use. Streams draining forested areas had concentrations of N (0.6 mg/L) and P (0.02 mg/L) that were an order of magnitude lower than streams draining agricultural areas (5.4 mg N/L, 0.2 mg P/L). Nitrate-N was also lowest in streams draining forests (Figure 1).

Forests typically occur on the steepest portions of a landscape, yet annual sediment yields from forested lands are lower than any other rural land use. Average sediment yields reported by Gianessi *et al.* (1986) were: forests 0.2 Mg/ha; rangeland 0.3 Mg/ha; pasture 0.3 Mg/ha; and cropland 0.9 Mg/ha.

A summary of water quality from the U.S.G.S.'s Hydrologic Bench Mark Streams showed that "natural" watersheds (primarily managed and unmanaged forest) averaged 0.06 mg-NO_3-N/L compared with 0.3 mg-NO_3-N/L for other streams (Biesecker and Leifeste 1975).

Patric *et al.* (1984) contrasted sediment yields from forested watersheds (< 5 km^2) to sediment yields from watersheds with other land uses (primarily agriculture). In the eastern U.S. the average annual sediment yield for managed and unmanaged forested watersheds was about 0.15 Mg/ha as compared to 0.35 Mg/ha for the other watersheds. In the West, forested watersheds again yielded about 0.15 Mg ha^{-1} yr^{-1} (1 Mg ha^{-1} yr^{-1} = 890 pounds per acre per year), as compared to 0.42 Mg ha^{-1} yr^{-1} for other land uses.

Patric *et al.* (1984) also compared the concentrations of suspended sediment in major rivers draining mostly forested areas with rivers draining areas with other land uses (Table 1). The concentration of suspended sediment was about 10 times greater in rivers draining non-forested areas.

Yoho (1980) compiled reported erosion rates for a variety of land uses on small watersheds in the South. Intact pine forests yielded the lowest quantities of sediment (from 0 to 0.2 Mg ha^{-1} yr^{-1}), with carefully clearcut forests (presumably applying BMPs) yielding only moderately more (0.1 to 0.4 Mg ha^{-1} yr^{-1}) over one to several years. Pastures produced 0.9-4.5 Mg ha^{-1} yr^{-1}, and carefully cultivated agricultural fields were found to yield from 0.9 to 16.8 Mg ha^{-1} yr^{-1}. Harvesting and site preparation without use of BMPs yielded between 3 and 14 Mg ha^{-1} yr^{-1} for one to several years, temporarily matching the annual erosion rate from carefully cultivated fields.

FIGURE 1 Average streamwater nitrate concentrations for large watersheds of different major land uses (from Omernik 1977) demonstrate that water from forests has the lowest concentrations.

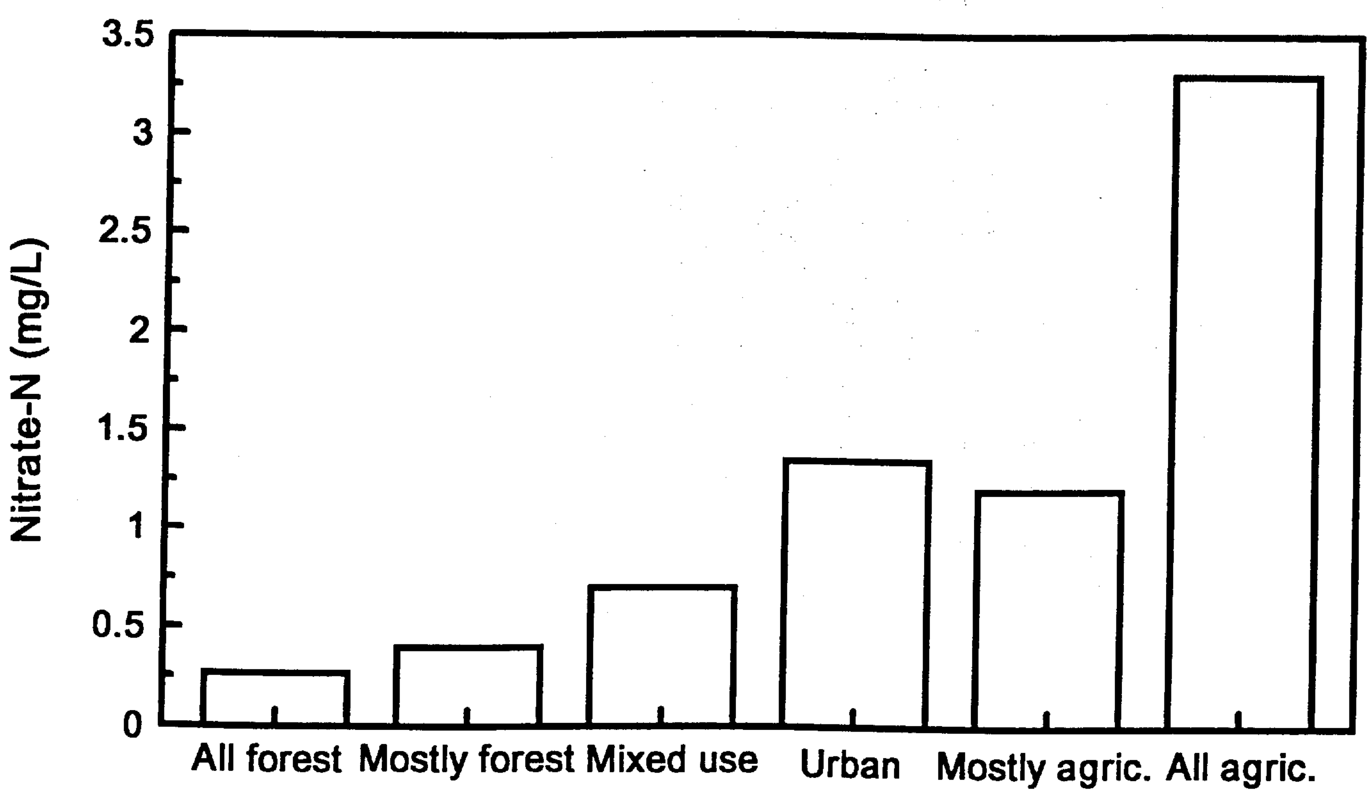


TABLE 1. Suspended sediment concentrations in major rivers draining forested and non-forested watersheds (from Patric et al. 1984).

River and State	Average sediment concentration (mg/L)
Predominantly forested	
Penobscot, ME	5
Blackwater, VA	10
St. Mary's, FL	5
Ford, MI	5
St. Croix, WI	9
Mokelumne, CA	8
Elwha, WA	6
Pend Oreille, ID	5
Umpqua, OR	12
Predominantly other land uses	
Susquehanna, PA	121
Pee Dee, SC	40
Alabama, AL	75
Ohio, IL	78
Platte, NE	723
Brazos, TX	774
Salinas, CA	1090
Yakima, WA	89

B. Conclusion on Proposition 1

We conclude that water draining forested watersheds is indeed among the best in the country, whether the forests are undisturbed or managed. Forest management practices have the potential to degrade water quality, particularly where landslides or other types of mass movements may alter channel conditions, but with few exceptions the overall level of water quality remains very high.

Proposition 2: Properly implemented BMPs effectively control nonpoint sources of pollution from forestry operations under most circumstances.

BMPs for forestry operations have been under informal and formal development for more than two decades, and the definition, design, and implementation of BMPs continue to evolve. BMPs differ among types of forests, states, and climatic regions, as do regulations governing their implementation (Binkley and Brown 1993). For the purposes of this proposition, we assume that the definition of BMPs generally matches current knowledge relative to protection of water quality within a given forest type or region.

A. Temperature

A variety of studies have documented rises in average summer temperatures following removal of forest canopies and occasional depressions in average winter temperatures. Increases in summer temperatures following complete canopy removal range from about 2 to 12°C across the United States (Table 2). For example, a commercial clearcut at the Leading Ridge watershed in central Pennsylvania used BMPs to minimize impacts on streams (Lynch *et al.* 1985). Stream temperatures increased by about 1 to 2°C during the summer months, and decreased by about the same amount in winter (Figure 2). In contrast, removal of riparian vegetation in another watershed allowed stream temperatures to increase by 5 to 11°C in summer months.

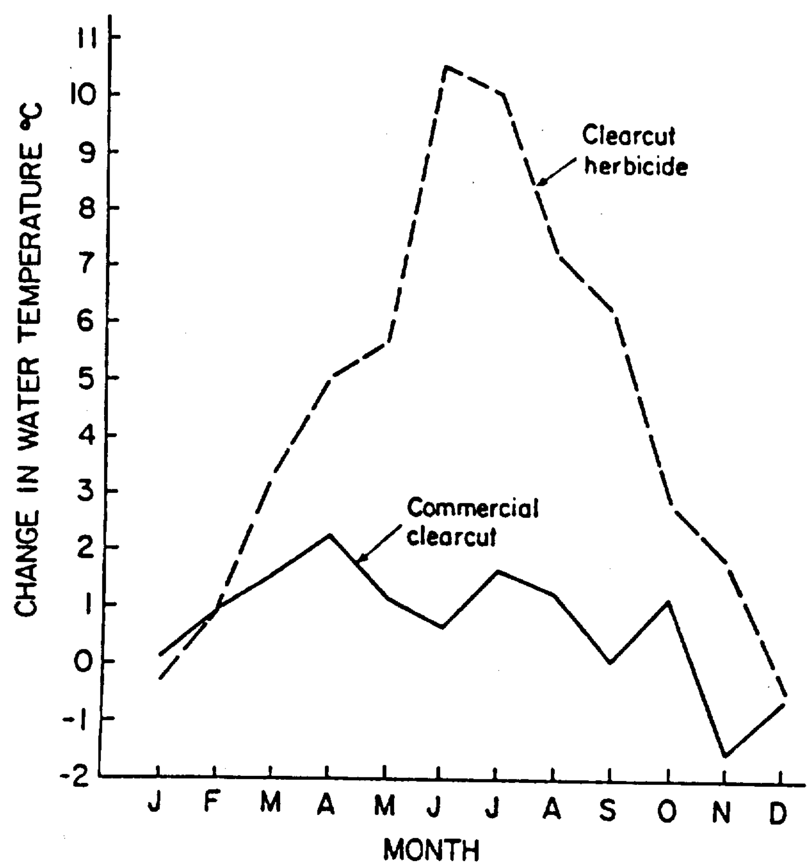
BMPs typically call for the retention of vegetation in riparian areas. Retention of canopy cover generally keeps changes in stream temperatures to less than 2°C (Beschta *et al.* 1987). A recent study in Washington indicated that small streams can be adequately protected by requiring forest operators to retain a specified amount of stream shading rather than retaining a specified proportion of the existing stream shading (Sullivan *et al.* 1990).

TABLE 2: Summary of temperature changes from forest practices and devegetation experiments (expanded from Beschta *et al.* 1987). Retention of buffer strips generally kept temperature changes to < 2°C.

Location	Treatment	Stream temperature variables	Temperature change (°C)	Reference
Georgia	Clearcut with partial buffer strip	Avg. June-July maximum	+6.7	Hewlett and Fortson 1982
Maryland	Riparian harvest up to 40 m from channel	Avg. summer maximum minimum	 +4.4 to 7.6 +0.6 to 1.1	Corbett and Spencer 1975
New Jersey	Riparian herbicide application	Avg. summer maximum minimum	 +3.3 0	Corbett and Heilman 1975
North Carolina	Deadened cove vegetation Clearcut Understory cut	Avg. summer maximum	+2.2 to 2.8 +2.8 to 3.3 0 to 0.3	Swift and Messer 1971
Pennsylvania	Riparian harvest	Avg. summer maximum minimum	 +3.9 0	Lynch *et al.* 1975
	Clearcut with buffer strip	Avg summer maximum minimum Avg. winter maximum minimum	 +0.6 to 1.6 0 to 0.6 -0.7 to +0.9 -0.5 to +1.0	Rishel *et al.* 1982
	Clearcut with herbicide devegetation	Avg summer maximum minimum Avg. winter maximum minimum	 +10 to 10.5 +1.7 to 1.8 -0.5 to +0.9 -1.4 to +0.2	Rishel *et al.* 1982
New Hampshire	Clearcut with herbicide devegetation	Avg. summer maximum	4	Likens *et al.* 1970
West Virginia	Clearcut without buffer strip	Avg. summer maximum	+4.4	Kochenderfer and Aubertin 1975

Location	Treatment	Stream temperature variables	Temperature change (°C)	Reference
Alaska	Clearcut and natural openings, no buffer strips	Change in temperature/ 100 m	0.1 to 1.1	Meehan 1970
British Columbia (Vancouver Island)	Clearcut, no buffer strip	Avg. daily temperature range in summer	0.5 to 1.8 beyond control	Hotlby and Newcombe 1982
	Clearcut and burned, no buffer strip	Avg. daily temperature range in summer	0.7 to 3.2 beyond control	Holtby and Newcombe 1982
British Columbia	Clearcut, no buffer strip	Maximum observed increase	3.8	Feller 1981
	Clearcut and burned, no buffer strip	Maximum observed increase	3.3	Feller 1981
Oregon (Cascades)	Clearcut, no buffer strip	Avg. summer maximum	4.4 to 6.7	Levno and Rothacher 1967
	Clearcut and burned, no buffer strip	Avg. summer maximum	6.7 to 7.8	Levno and Rothacher 1969
Oregon (Coast Range)	Clearcut, no buffer strip	Avg. summer maximum	2.8 to 7.8	Brown and Krygier 1970
	Clearcut and burned, no buffer strip	Avg. summer maximum	9 to 10	Brown and Krygier
Oregon (Cascades)	Mixed clearcut and forested reaches	Change in temperature/ 100 m	0 to 0.7	Brown *et al.* 1971
	Tractor stripped riparian zone	Change in temperature/ 100 m	15.8	Brown *et al.* 1971
Oregon (Cascades)	Shelterwood cut	Avg. summer maximum	0	Adams and Stack 1989
	Patch cuts	Avg. summer maximum	0	Adams and Stack 1989
	Clearcut	Avg. summer maximum	3 to 8	Adams and Stack 1989

FIGURE 2 Commercial clearcutting with retention of a buffer strip allowed stream temperatures to increase by about 1°C in an experiment in Pennsylvania. Complete devegetation of the riparian zone allowed temperatures to increase by up to 10°C (from Lynch *et al.* 1985).



B. Sediment

Forest streams vary dramatically in sediment concentrations over time, within the stream cross-section, and between streams. For example, some streams draining intact forests in the Southeast show high concentrations of suspended sediments during storm events, with maximum concentrations on the order of 80 to 300 mg/L (e.g., Schreiber and Duffy 1982, McClurkin *et al.* 1985). Some areas, such as the Upper Coastal Plain in Mississippi, may have suspended sediment concentrations during storms of more than 2000 mg/L even in undisturbed forests (Beasley 1979). High natural variability can make it difficult to quantify effects of forest practices on sedimentation.

Important sources of fine sediment include roads and ditches, cutbanks, slope failures and debris flows, streambank erosion and channel scour, and the diversion of streams at road crossings. In most cases, increases in sediment concentrations from forest activities derive mainly from road construction activities, and decline within a few years as road surfaces stabilize. In some cases, sediment concentrations in road ditches may remain elevated as long as the road is in use (W. Megahan, personal communication). Reid and Dunne (1984) reported sediment concentrations in ditches in excess of 31,000 mg/L during road use by logging trucks, and Sullivan and Duncan (1981) reported concentrations between 1,000 and 18,000 mg/L.

Implementation of BMPs generally minimizes impacts on sediment concentrations. For example, a watershed study in Pennsylvania tested whether BMPs could prevent substantial impacts on water quality (Lynch *et al.* 1975, 1985). As part of the BMP approach, only 43% of the watershed was harvested; 30-m buffer strips were retained along streams; performance of the contractor was monitored intensively; locations of roads and skid trails were determined in advance by a forester; and all roads and trails were rehabilitated after logging. Sediment concentrations in the first year after harvest averaged 1.7 mg/L for the control watershed, and 5.9 mg/L for the harvested watershed. Windthrow of some trees in the streamside buffer strip increased average sediment concentrations to 9.3 mg/L in the second post-harvest year. Sediment concentrations remained slightly elevated relative to the control watershed for at least 10 years after harvest. These researchers concluded that use of BMPs did not completely prevent impacts on water quality, but that the impacts were relatively small and of no concern relative to water quality guidelines. [Note that some states allow only a 10%, 20%, or even no increase in sediment concentrations above background, so even insignificant changes can violate this type of numeric criteria. It may not necessarily follow that the designated beneficial use associated with that criteria has been adversely affected.]

One of the best studies of BMP effectiveness comes from California, where Knopp *et al.* (1987) followed the adequacy of BMPs for a 10-year period in the Six Rivers National Forest. Comparisons of landslides before and after the implementation of BMPs showed a 50% reduction in harvest-related slides and an 85% reduction in the number of landslides associated with roads. The percent of fine materials in the stream bottoms declined from 20-25% before BMP implementation to about 13-18% after implementation.

In a similar survey of 40 projects across Idaho (Harvey *et al.* 1988), operations that complied with BMPs led to stream sedimentation problems in less than 10% of the cases, whereas situations that did not comply with BMPs led to sedimentation problems in 70% of the cases.

In Georgia, compliance with BMPs is voluntary, yet compliance averaged about 85% for the 345 investigated operations (Georgia Forestry Commission 1991). About 95% of the streambanks and channels remained intact, although longer-term monitoring would be necessary to see if subsequent storms affect stream channels.

C. Dissolved Oxygen

Current forest practices generally do not substantially deplete oxygen levels in streams. A few studies have documented that adding large amounts of fine litter to small, low-turbulence streams can deplete stream oxygen. One study from Oregon reported that large additions of fine litter materials into a low-turbulence stream depleted oxygen concentrations to near zero until slash was pulled from the stream (Hall *et al.* 1987). Similar results were reported for a stream in Quebec (Plamondon *et al.* 1982). BMPs, particularly use of buffer strips, should preclude excessive input of fine organic materials in streams and ensure that dissolved oxygen levels in streamwater remain high (MacDonald *et al.* 1991).

Effects of forest practices on dissolved oxygen in water within streambed sediments is less clear. Increased sedimentation may lead to decreased permeability of streambeds and depressed oxygen concentrations within the streambeds (Everest *et al.* 1987). Proper design and implementation of BMPs should greatly reduce any increase in fine sediments in the stream channel and thereby minimize the reduction in dissolved oxygen in the gravel beds.

D. Nutrient Concentrations

Most forest harvesting studies in the United States have documented increased concentrations of nitrate following harvest (Binkley and Brown 1993), but in almost all cases these increases have remained well below the 10 mg-N/L drinking water standard. Three exceptions are worth noting (Binkley and Brown 1993).

(1) Some streams in the Pacific Northwest draining undisturbed forests of red alder (a nitrogen fixing species) average more than 1 mg-N/L with maximum concentrations above 10 mg-N/L. After harvesting, some sites show lower nitrate export, and some higher. More information is needed before generalizations can be made.

(2) High concentrations of nitrate have been observed in waters draining from high-elevation forests in the southern Appalachian mountains. For example, some streams draining watersheds with old forests of red spruce had average concentrations of 5 mg-N/L, with higher maximum values (Silsbee and Larson 1982). Factors contributing to elevated nitrate concentrations may include high rates of atmospheric nitrogen deposition and low rates of nitrogen uptake by the forest. Rates of nitrogen uptake may be affected by harvesting, and by changes in vigor that have been attributed to stand maturation or regional air pollution.

(3) In New Hampshire, high concentrations of nitrate have been observed in harvesting experiments with northern hardwoods in the White Mountains. Average post-harvest concentrations of nitrate in streamwater have exceeded 3 mg-N/L in some studies (Hornbeck *et al.* 1986, 1987). In other harvesting studies with northern hardwoods and other forests types in the Northeast, nitrate concentrations were much lower (Martin *et al.* 1984).

Forest fertilization typically increases streamwater nutrient concentrations, but most studies have shown that these increases are too small to degrade water quality (Binkley and Brown 1993). A few exceptional cases have been reported. In the Fernow Experimental Forest in West Virginia, several fertilization studies found nitrate concentrations in excess of 10 mg-N/L up to several months after application of 225 to 340 kg-N/ha (Kochenderfer and Aubertin 1975, Helvey *et al.* 1989, Edwards 1991). Fertilizer was applied carefully by hand, with no application directly to streams. Fertilization is an uncommon practice for hardwood forests in the East. Fertilization is a routine practice on many intensively managed pine forests in the Southeast, but few studies have examined fertilization effects on water quality (Shepard 1994). Campbell (1989) found peak stream concentrations of 1.2 mg/L of nitrate-N after fertilization of a loblolly pine forest on the Coastal Plain of North Carolina, similar to results at another site (Herrmann and White 1983, Fromm 1992). The high nitrate concentrations in the hardwood fertilization studies do not appear to apply to pine forests in the Southeast, but more investigation may be warranted.

In the Pacific Northwest only one of several dozen forest fertilization studies (Fredriksen *et al.* 1975, Meehan *et al.* 1975, Tiedemann *et al.* 1978, Bisson 1982, 1988, Hetherington 1985, Bisson *et al.* 1992a) found nitrate concentrations that approached the drinking water standard. Hetherington (1985) found peak nitrate concentrations of 9.5 mg-N/L about 2 months following a commercial fertilization operation (with application directly over streams) on Vancouver Island.

A few harvesting studies have shown slight increases in phosphate concentrations after logging (Salminen and Beschta 1991), but these increases were far too slight to degrade water quality, although some increase in stream productivity may have resulted.

E. Pesticide Concentrations

Pesticides may have direct and indirect effects on aquatic biota (for an excellent summary see Norris *et al.* 1991). Toxic effects result from direct exposure to the pesticides in the streamwater or through food intake. Indirect, non-toxic effects may result from changes in the riparian vegetation, increased inputs of organic matter to streams, and changes in physical properties of streams (through secondary effects such as an increase in the sediment load or an increase in temperature). The major pathway for pesticides to enter streams is by direct application, including drift from nearby treatment areas. The relative absence of articles in the literature that document impacts of pesticides on water quality suggests that adverse impacts are minimal (MacDonald *et al.* 1991, Norris *et al.* 1991).

Herbicide spraying for hardwood control in conifer forests typically results in streamwater concentrations of less than 0.1 mg of herbicide/L of streamwater, well below minimum concentrations typically needed to affect stream plants (Norris *et al.* 1991). Indirect effects of herbicides may result from effects on riparian vegetation, including increased nitrate inputs (Neary 1988), decreased slope stability, and altered food web structure in streams. However, critical indirect effects have not been documented for any normal forest use of herbicides, and BMPs typically restrict herbicide application to non-riparian zones.

Insecticide spraying may kill both terrestrial and aquatic insects, temporarily increasing the "drift" of dead insects in streams (Norris *et al.* 1991). This may dramatically increase the availability of food for drift-feeders such as trout, and if the insecticide has no effect on the fish, a brief increase in fish production may result. Given the short life cycle of insects, insect communities should recover within a few years.

Retention of untreated buffer strips along streams reduces the input of pesticides into streams. For example, the Forest Practices Program of the Oregon Dept. of Forestry (1992) examined the concentrations of herbicides in streams following aerial application on 50 watersheds. Over 80% of the streams sampled had no detectable herbicide, and those with detectable levels remained 2 to 30 times below the accepted standards. The report concluded that retention of 20-m wide unsprayed buffer strips was adequate for stream protection in most cases. The amount of herbicide reaching the stream channel also depends on factors such as the wind speed, wind direction, and type of carrier for the herbicide, and these can be regulated through the BMP process.

F. Stream Channel Conditions

This category includes a wide range of physical properties of streams, including stream cross sections, width-to-depth ratios, composition of streambed materials (especially the proportion of fine sediments), amount of coarse woody debris, pool depths, and bank stability. These conditions have major effects on habitat suitability for fish and other aquatic organisms.

Forest practices, particularly road construction and harvesting, may substantially alter stream channel conditions. Such changes can range from subtle changes in the bed material particle size to the dramatic effects of landslides and debris flows (Benda 1990). Most studies on the effects of forest practices on stream channels have been conducted in the Pacific Northwest and northwestern California, where steep slopes, erodible soils, and high precipitation produce a geomorphically active landscape. Despite widespread concern over the effects of forest practices on stream channel conditions, only a few studies have documented the full chain of cause-and-effect from forest practices to resultant impact on fish populations (Sullivan *et al.* 1987). The following paragraphs illustrate the variability in stream channel effects and recovery rates, as well as the importance of extrinsic factors.

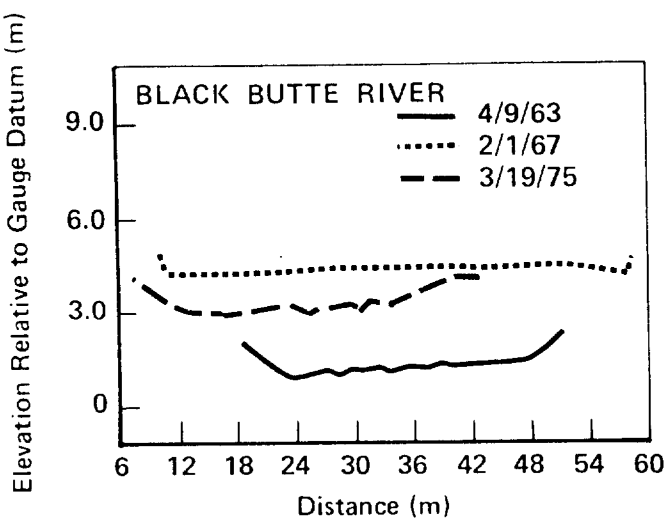
Sullivan *et al.* (1987) summarized several case studies. From 1950 through 1970, extreme storm events combined with extensive logging to substantially alter stream channel conditions in parts of northern California. The beds of some streams were raised by as much as 4 m, and stream widths doubled. Particle sizes of streambeds decreased and stream channels shifted or became braided. Pools were filled in, and riffles were less pronounced. Summer flows were reduced as more water flowed through the deposited materials within the stream channel. The shifting and widening of the channels degraded riparian vegetation and eroded streambanks. The effects on fish populations were poorly documented, but populations declined over this period. Subsequent recovery of the stream channels varied after these large floods. Stream depths generally increased and stream widths generally decreased, but most streams channels had not returned to pre-flood conditions within 10 years (Figure 3). The impacts of these large flows were dramatic, but the contribution of forest practices to the impacts of the floods could not be separated from the storm effects.

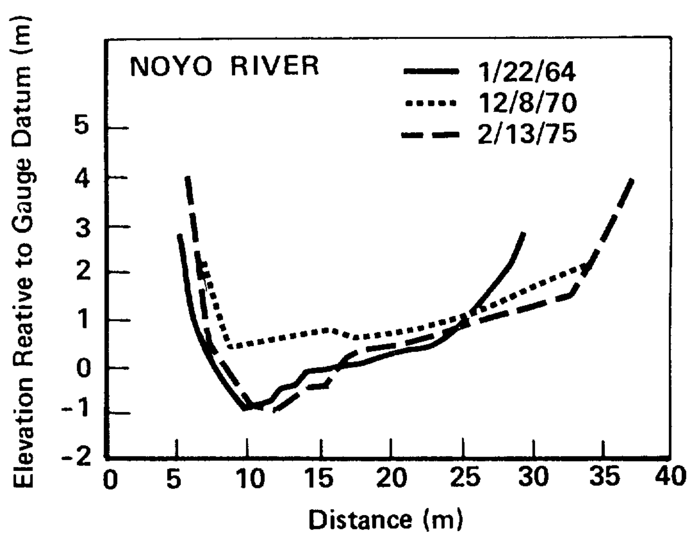
Another example comes from the Middle Fork of the Willamette River in Oregon. In 1964 a major flood substantially widened the stream channel. Sediment loads increased from bank erosion and landslides, which were often associated with logging roads (Grant 1986, Sullivan *et al.* 1987). Debris flows in headwater streams scoured channels to bedrock. Twenty years later the effects of this flood were very evident; the channel contained bars of cobbles that were too large for moderate flows to move, and the riparian vegetation was dominated by willows and red alder rather than conifers. Although logging roads accounted for a disproportionate number of landslides, the contribution of roads to the overall effects of this large flood are unclear.

The South Fork of the Salmon River showed relatively rapid recovery to the combination of logging and large storms. This 1000 km^2 watershed has highly erodible granitic soils (Sullivan *et al.* 1987, Megahan *et al.* 1980), and between 1950 and 1965 about 15% of the basin was logged. The combination of logging and approximately 1000 km of road construction, when combined with a series of severe storms, more than tripled the amount of stored sediment in the stream channels. Following a 1966 moratorium on logging, sediment deposits were reduced and the percentage of fine materials on the surface of the spawning beds declined from 30% in 1966 to 8% in 1979. Reductions in sediment inputs after the logging moratorium apparently allowed the high-energy streams to remove the excess sediments in a relatively short time.

The Clearwater River on the western slope of the Olympic Mountains in western Washington has been the site of studies on the effects of forest practices on stream channel characteristics and populations of coho salmon (Cederholm and Reid 1987). About 60% of the watershed had been logged at least once, with a road density of about 3 km/km^2. Recent practices have included buffer strips of old forest (10 m to 100 m wide) on harvest units of state-owned land. Stream conditions had been substantially altered in some reaches from old harvesting operations that had included limited protection of streams, and from some more recent harvests on private land. Undisturbed stream channels in nearby Olympic National Park show that fines comprise less than 10% of streambed materials, compared with 10-15% for moderately impacted

FIGURE 3 Channel cross sections for the Black Butte River (upper) and the Noyo River (lower) in northwestern California, before and after a major flood in 1964 (Lisle 1982, cited in Sullivan *et al.* 1987). Immediately after the flood, aggradation of the streambed made the stream shallower and wider; only partial recovery occurred in the following years.





watersheds, and 15-20% for heavily impacted watersheds within the Clearwater River drainage. The percent of fine materials in the streambeds correlated with the percent of the basin in roads (Figure 4); landslides associated with roads were identified as the key sediment sources. Despite these impacts, Cederholm and Reid (1987) concluded that if the Clearwater River received sufficient spawners, the coho salmon population would not be adversely affected by forestry activities alone. Overfishing and degraded habitat combined to substantially reduce the coho population.

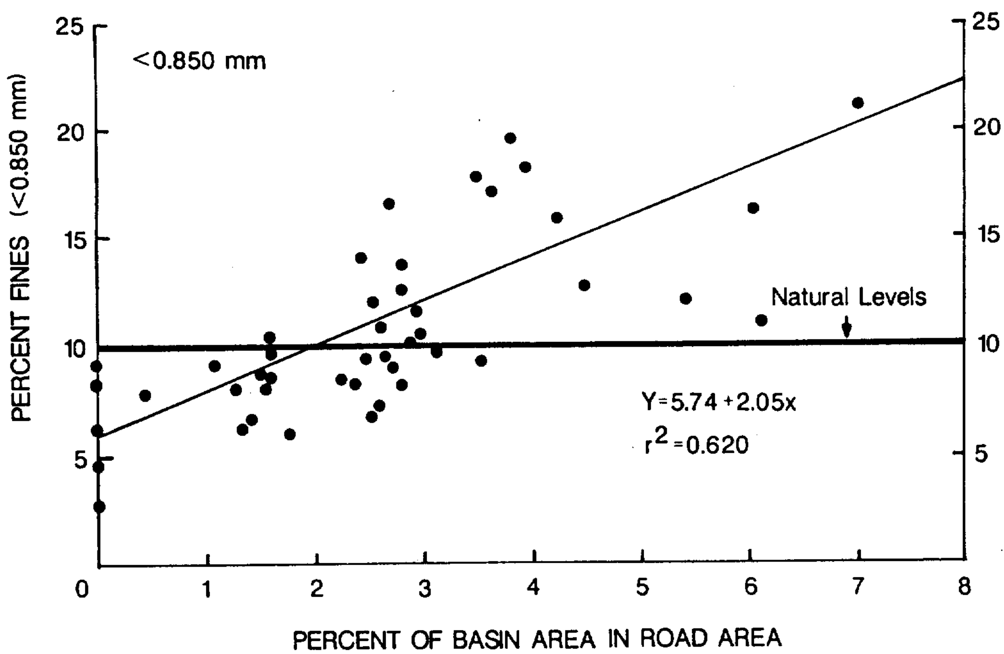
The complex interactions between management practices and storm events have been documented for the 10 km² Carnation Creek drainage on the west side of Vancouver Island (Hartman *et al.* 1987, Powell 1988, Scrivener 1988, Hartman and Scrivener 1990). In the late 1970's three harvesting treatments were applied along different sections of Carnation Creek:

(1) Maximum disturbance of the riparian zone (removal of all trees and shrubs, removal of woody debris from the stream, and some yarding of logs across the stream);
(2) Careful logging with retention of streamside vegetation but no extensive buffer strip, and no physical disturbance of the stream during logging;
(3) Careful logging with a buffer strip (from 1 to 70 m wide) along the stream.

The intensive treatment reduced the amount of coarse woody debris in the stream channel by 25 to 75%, but the other two treatments showed no significant reduction. Where the buffer strip was retained, the average stream width increased by 0.6 m in 7 years, giving an annual rate of bank loss of 93 m²/km of stream length, and an annual erosion rate of 0.5 m³/m of stream. In the carefully logged area without a buffer strip (treatment 2), the stream width increased by more than 8 m in 6 yr, and about 1650 m² of streambank was lost annually per km of stream. Maximum pool depths declined after logging by about 5 to 15 cm per year in all three treatments. Seven years after the intensive logging treatment, the quantity of fine materials in the streambed was continuing to increase. More recent observations indicate accelerating changes in the stream channel as large storm events mobilize the sediment that was delivered to the channel during logging. Apparently, the initial impacts were minimized by the absence of large flow events (S. Chatwin, B.C. Ministry of Forests, personal communication 1992).

In contrast, Sullivan (1985) examined the cumulative, 9-year effects of road construction and logging in an 8000-ha block along the Middle Santiam River in Oregon. About 3400 ha were harvested using high-lead cable systems in units of about 20 ha

FIGURE 4 For the Clearwater River area in northwestern Washington, the percent of fines in streambeds correlated with the percent of the basin area in roads (from Cederholm and Reid 1987).



each, and a 180 km of roads were carefully built. No long term increases in suspended sediment concentrations were detected. Annual sediment yield averaged 1.3 Mg/ha for the study site as compared to 1.6 Mg/ha for the upstream river. It was concluded that the cumulative impacts on water quality of road construction and logging – conducted in accordance with Oregon's forest practices regulations – appeared negligible.

G. Populations of Fish and Invertebrates

Populations of fish and invertebrates depend on a large variety of stream features, including temperature, light, nutrients, physical characteristics (including coarse woody debris and pools), and hydrology. The characteristics of communities in streams are very dynamic over time, showing both short-term fluctuations and long-term trends (Gregory *et al.* 1987). Short-term fluctuations may result from unusual floods or droughts, and long-term trends may result from successional development of riparian vegetation.

Road construction and logging may impair fish habitat if excessive sedimentation reduces the permeability of streambed gravels, or if channel structure is drastically altered (such as a reduction in depth or number of pools). If adequate shade is not retained during forest harvest, increased temperatures may also be a problem. Temperature effects typically last for only one or two years (sometimes up to 10 years) within a single stream, but the cumulative effects within an entire basin may be more persistent (Gregory *et al.* 1987).

Many studies have documented increased numbers and sizes of fish following logging (Hall and Lantz 1969, Aho 1977, Murphy and Hall 1981, Murphy *et al.* 1981, Hawkins *et al.* 1983, Tschaplinski and Hartman 1983, Bilby and Bisson 1987, other references cited in Gregory *et al.* 1987 and Bisson *et al.* 1992b). Logging may substantially alter the foodwebs of streams, especially if light penetration through the surrounding canopy increases (Gregory *et al.* 1987). Increased light may lead to increased production by benthic algae, and herbivorous insects may respond to the increase in available food. Moderate increases in stream temperature may favor fish growth, especially at high elevation and in northern latitudes where low stream temperatures often limit productivity. Altered stream temperatures may also shift the dominance of fish species (Reeves *et al.* 1987).

Removal of the plant canopy is likely to decrease inputs of detritus and coarse woody debris to streams, and detritivore populations may decline. However, canopy removal often increases the overall population of macroinvertebrates. For example, Newbold *et al.* (1980) estimated the population of macroinvertebrates for a stream in northern California. The total density of macroinvertebrates in a control reach and in reaches bordered by buffer strips of vegetation averaged about $1500/m^2$, as compared with 4500 organisms/m^2 for a logged reach with no buffer strip. In another California study, however, macroinvertebrates densities were lower 10 years after clearcutting with no buffer strips than in control streams (Erman and Mahoney 1983), and this was attributed to altered channel characteristics.

Increased food supplies and increased stream temperatures may increase survival of young fish. In Carnation Creek, warmer temperatures after logging were expected to allow earlier emergence of fry and earlier smolt migration, perhaps doubling the biomass of coho salmon smolts for several years (Holtby 1988a,b). An increase in the population of one species may come at the expense of a reduction in some other species (Bisson *et al.* 1992b). Increased (or decreased) survival of young fish in summer may or may not affect overall populations of fish. Other factors may be critical, such as habitat during winter floods in Alaska (Koski *et al.* 1984), fishing pressure, dams, and ocean survival rates.

Logging operations may also influence habitat; blowdown of trees within buffer strips can temporarily increase the amount of woody debris in streams and improve salmonid habitat. In the absence of a buffer strip, inputs of coarse woody debris will decline. A reduction in coarse woody debris can substantially degrade fish habitat (Maser *et al.* 1988, Bisson *et al.* 1992b). Many states and national forests are developing and implementing forest practice rules and guidelines to retain trees along streams for future recruitment of coarse woody debris.

Application of insecticides may reduce insect populations near streams, and some studies have concluded that short-term (one-to-two year) reductions in fish populations may occur (e.g., Reed 1966). Too few studies have been conducted to generalize the impacts of insecticides on fish populations in forested streams.

Gregory *et al.* (1987) developed a generalized pattern of the course of stream productivity after logging for streams in the Pacific Northwest. According to their expectations, logging should increase primary production and sustain higher macroinvertebrate populations for several years. Stream productivity and macroinvertebrate populations may show a later decline if a uniform canopy of conifers develops over the stream. Long-term impacts may depend heavily on the effects of logging on the short- and long-term abundance, type, and size of coarse woody debris in streams, and on sedimentation of the stream channel (Everest *et al.* 1987).

In the larger picture, over 200 native stocks of anadromous Pacific salmon may be under moderate to severe risk of extinction in the Northwest (Nehlsen *et al.* 1991), and habitat loss is a major factor behind the declines of these stocks. Unfortunately, Nehlsen *et al.* (1991) could not separate habitat loss into contributions from forest practices and other factors, such as hydroelectric dams.

In summary, past logging practices have often resulted in a simplification of stream channels. Sedimentation and the removal of large debris reduces the diversity of habitats, particularly the size and frequency of pools (Bisson *et al.* 1992b). Bisson and Fransen (cited in Bisson *et al.* 1992b) found that the diversity of all fish species increased as the variation in stream velocity increased in Huckleberry Creek in Washington.

Logging typically increases the productivity of streams, more than doubling the population of benthic invertebrates in some cases (Erman *et al.* 1977). At the same time that overall populations increase, the diversity of invertebrates (Erman *et al.* 1977) and fish species may decrease. Reeves *et al.* (cited in Bisson *et al.* 1992b) found that salmonid species diversity in logged watersheds was only about half that of unlogged watersheds along Oregon's coast.

H. Conclusion on Proposition 2

As a general rule, we conclude that Proposition 2 is true. If properly designed or implemented, BMPs can largely eliminate the adverse effects of forest management activities on stream temperature, dissolved oxygen, and the concentration of nutrients and pesticides (Commerford *et al.* 1992).

However, a number of case studies – primarily from the Pacific Northwest – show that road construction and forest harvest can substantially alter sediment yields and stream-channel conditions. These in turn can affect aquatic life in both the stream channel and interconnected groundwater zone. Although there are remarkably few detailed studies documenting the effectiveness of specific BMPs, the proper application of BMPs, such as appropriate road construction and retention of buffer strips, substantially reduces these impacts. However, the application of BMPs will not prevent all changes in stream channel conditions in all areas. Slope failures and landslides do occur with greater frequency on logged units in some regions (see Proposition 4), and there also is potential for adverse effects further downstream (see Proposition 6). Rates of degradation and recovery are complex functions of stream type, amount and size of sediment supply, amount of coarse woody debris, and the timing and magnitude of storm events. These points indicate that we need to develop and test more stringent BMPs in some locations, improve our capability to evaluate risk, and focus future research on erosion, sedimentation, and effects on stream channels and fish habitat.

Proposition 3: Effects of present day management activities on water quality are usually transient and are rarely severe enough to cause significant damage to fish populations.

The protection of water quality during forest operations has improved substantially over the past two decades. Generalizations about impacts of forestry from the 1960's cannot be directly applied now. For example, Knopp *et al.* (1987) compared the impacts of forest activities on landslides in the Fox Planning Unit of the Six Rivers National Forest in California. The unit receives about 3000 mm/yr of precipitation, and has steep, erosion-susceptible slopes. The authors examined patterns in landslides for the pre-1976 (pre-BMP) period with later impacts from improved practices (including narrower, high gradient roads with outsloped drainage in upper slope positions; uphill cable yarding; and exclusion of geologic hazard areas and streamside zones from harvesting). As noted in the discussion on Proposition 2, landslides associated with harvesting activities dropped by about 60%, and those related to roads declined by about 85%. These reductions are substantial, though it is difficult to determine the proportional reduction in impacts on fish populations, as the latter would depend on a variety of site-specific factors.

A. Temperature

Present day management activities, including retention of buffer strips along streams, should not result in severe changes in streamwater temperatures. As noted above, changes in water temperature following harvesting are typically less than 2°C where buffer strips are retained. These smaller increases in temperature may stem from several sources. If the buffer strips are narrow or windthrow causes gaps, there will be an increase in solar radiation to the stream surface. In some cases, formerly intermittent channels may become perennial after harvesting, and water in these channels may heat up from direct sunlight before entering the perennial streams with buffer strips (Rishel *et al.* 1982). Only one study has reported major changes in stream temperatures after harvest when a buffer strip was retained (Hewlett and Fortson 1982). In this case maximum summer temperatures increased about 11°C in a harvested unit, despite the retention of a 12-m, uncut buffer strip.

B. Sediment

The effects of sediments on fish derive primarily from impacts on the permeability of streambed gravel and resultant success of reproduction, habitat loss due to filling of pools, and reduction in food supply due to sediment effects on lower trophic levels (MacDonald *et al.* 1991). Adult fish may adapt well to suspended sediment concentrations of over 1000 mg/L

(Everest *et al.* 1987). Controlled experiments under laboratory conditions have documented clear relationships for certain salmonid species between the proportion of fine sediments (and hence permeability) in gravel beds and survival until emergence, but it has proven very difficult to extrapolate such studies to the variable conditions of real streams and rivers (Everest *et al.* 1987).

Massive inputs of sediment, from either forest activities or natural mass movements, may reduce the reproductive success of salmon (e.g., Platts and Megahan 1975, Everest and Meehan 1981), but most studies are unable to separate the effects of forest practices and increased sediment production from the confounding impacts of fish harvesting and mortality from other sources, such as migration through dams (Cederholm and Reid 1987, Everest *et al.* 1987).

In Alaska, logging along Harris River and Twelvemile Creek increased the concentration of fine sediments in gravel beds for about five years, and this appeared to reduce the survival rate of eggs and fry. However, the density of live alevins increased during this period, perhaps due to a confounding increase in the number of spawning fish (Salo 1967, Gibbons *et al.* 1987). Gibbons *et al.* (1987) noted that the high year-to-year variation in sediment yield in both the logged and unlogged watersheds made it difficult to determine whether logging activities increased sedimentation, and thus whether logging activities did impair the salmon fisheries. The Alaska Division of Fish and Game has abandoned efforts to monitor any impacts on fish from sediments from logging, and shifted their emphasis to improved land use planning and proper application of BMPs (Gibbons *et al.* 1987).

Documentation of the transient effects of management practices on sediment is difficult because of the high temporal variability in sediment concentrations within storms, between storms, and from year to year (e.g., Walling and Webb 1981, Meade *et al.* 1990, MacDonald 1993). Relatively few studies have evaluated the full cycle of predisturbance sediment yields, sediment yields during management, and post-disturbance recovery rates. In many cases sediment production decreases to undetectable levels within a few years, but appears to vary by site and intensity of treatment (e.g., Marion and Ursic 1993). The continuing presence and use of unpaved roads typically precludes recovery to pre-disturbance levels (USFS 1981, C. Troendle, USFS, pers. comm.). Changes in flow can also increase sediment concentrations if large amounts of sediment are stored in the stream channels (Marion and Ursic 1993).

C. Dissolved Oxygen

Large inputs of fine organic debris may reduce dissolved oxygen concentrations in streamwater (e.g., Hall and Lantz 1969, Hall *et al.* 1987), but high turbulence in most forested streams rapidly replenishes dissolved oxygen concentrations (Ice 1978). Current forest practices do not generally result in high inputs of fine organic debris, and use of BMPs should prevent any oxygen depletion problems after forest harvesting. Problems with low dissolved oxygen concentrations may occur where logs are stored in rivers or lakes (NCASI 1992).

D. Nutrient Concentrations

Any effect of forest practices on nutrient concentrations should be limited to modest increases in stream productivity; the literature contains no examples of damage to fish populations from nutrient concentrations following forest harvest or fertilization.

E. Pesticide Concentrations

Many chemicals used to control vegetation and insects have impaired the development of fish in laboratory studies, including the herbicides picloram and 2,4-D, and the insecticides malathion and carbaryl. Norris *et al.* (1991) summarized the state-of-knowledge and used field data to calculate exposures that might result from forest applications of pesticides. Relative to critical concentrations determined in laboratory studies, these calculated exposures should show no effect on fish. Table 3 provides their estimates of the likely maximum exposure fish would receive from operational application of several pesticides, based on the observed peak concentration (in mg/L) times a maximum exposure period of 48 hrs (giving [mg/L]*h). These exposures range from about 0.1 [mg/L]*h for the herbicide 2,4-D to 1.7 [mg/L]*h for the insecticide acephate. The ratio of the exposure needed to produce a minimum effect on fish to the exposure that would result from forest operations was defined as a "margin of safety." A margin of 1 would mean the minimum exposure necessary to have an observable impact would match the expected exposure; a margin of 10 would mean that the field exposure reached only 1/10 of the level necessary for an observable impact. For most pesticides, the margin of safety exceeded one, and in many cases exceeded 100. Contamination of streams by pesticides is very unlikely except for accidental spills or direct application to streams. Use of BMPs, such as retention of unsprayed buffers along streams and avoiding application during windy periods, should prevent any direct toxicity to fish (MacDonald *et al.* 1991).

F. Stream Channel Conditions

The impacts of road construction and forest harvest may be slight, or may substantially increase the fine sediments in gravel beds, widen stream channels, or trigger debris flows that scour the stream bottom. A variety of studies have implicated

TABLE 3: Integral of concentration-time curves for 48 hours for several pesticides after aerial application, based on highest observed peak concentration (Column 3), expected maximum exposure that could occur if untreated buffer strips were retained along streams (Column 4), the minimum exposure needed for an observed effect on fish (Column 5), and the margin of safety (Column 6) (from Norris *et al.* 1991).

1. Chemical	2. Actual peak concentration (mg/L)	3. Integral for actual peak concentration [(mg/L)*hr]	4. Short term exposure estimate for peak 0.02 mg/L [(mg/L)*hr]	5. No observed effect limit for fish [(mg/L)*hr]	6. Margin of safety (ratio column 5:4)
2,4-D	0.014	0.116	0.334	48	144
Amitrole	0.110	0.498	0.091	--	--
Dicamba	0.037	0.310	0.167	--	--
Malathion	0.040	0.074	0.037	0.082	2.2
Acephate	0.471	1.708	0.072	48	666
Carbaryl	0.121	0.343	0.057	2.08	36

roads and logging as the causes of increased erosion and mass movements that have degraded stream channels (e.g., Kelsey 1980 for northern California, Megahan *et al.* 1980 for the South Fork of the Salmon River, Grant et al. 1984 for the Middle Fork of the Willamette, Hartman *et al.* 1987 for Carnation Creek, Cederholm and Reid 1987 for the Clearwater River). In general, the largest impacts resulted from the combination of unusually intense storms with an increased vulnerability due to changes in soil moisture and drainage from roads or logging.

In the South past agricultural practices have led to the aggradation of many stream channels. Current increases in the size or duration of peak flows that might result from forest practices could remobilize this sediment and alter stream channels (Murphey and Grissinger 1985). Mechanical site preparation is the activity most likely to trigger large and sustained increases in sediment production (Marion and Ursic 1992). Detailed studies on the Hubbard Brook experimental watershed suggest that erosion and sedimentation can be controlled if BMPs are adhered to during and after logging (Martin and Hornbeck 1993).

The use of appropriate BMPs generally minimizes the impacts of forest harvest on stream channel conditions. Past studies suggest that road construction and drainage, avoidance of failure-prone slopes, and the establishment of buffer strips along channels are critical to protecting water quality and aquatic resources (see Section F under Proposition 2). Effective BMPs in one area may not be effective in other, more sensitive areas. In particular, standard BMPs may not be sufficient to prevent large changes in stream channel condition in areas with a high susceptibility to mass movements (see Proposition 4 below). It also is important to note that downstream depositional areas may be more affected by an increase in sediment loads than higher-gradient source areas. Recovery rates can be slow (Section F, Proposition 2), and this is particularly true for the replenishment of large woody debris (Sedell *et al.* 1988).

G. Populations of Fish and Invertebrates

The potential for forest practices to degrade streams and reduce populations of fish is well established – increased sedimentation or degradation of stream channel conditions clearly can degrade fish habitat. Aside from extreme events, such as landslides and debris torrents triggered by unusually intense storms (Sullivan *et al.* 1987), it has proven difficult to connect any decrease in fish populations with the effects of management activities. As noted earlier, studies on the effects of logging on fish populations in Alaska were generally inconclusive because of large natural variations in stream bed conditions and high year-to-year variation in numbers of spawning fish (Gibbons *et al.* 1987). In the intensively studied Clearwater River basin, forest practices increased the sediment concentrations in gravel beds and may have reduced habitat quality for fish; however, excessive fishing of the coho stocks also resulted in too few spawning fish (Cederholm and Reid 1987).

Conceptually, one would expect the recovery of fish populations to be related to the recovery rate of the perturbed habitat factor, but with a lag appropriate to rebuilding the population of concern. Invertebrate populations will recover much more rapidly than fish populations, due to both their shorter life cycle and the regular drift of potential colonizers.

Stream shading will generally recover in a few years if the stream channel has not been severely altered. Replacement of the large woody debris, which is essential to habitat complexity and channel morphology, occurs on a time scale of decades to centuries. Decreased habitat quality may persist for 5 to 10 years following sedimentation, or for much longer if the stream bed was scoured by debris flows or severely altered (Sullivan *et al.* 1987). Bisson *et al.* (1992b) discuss the recovery rate of fish populations following disturbance, and caution that with continuing fishing pressure recovery will be measured over decades.

H. <u>Conclusion on Proposition 3</u>

We conclude that severe impacts on fish populations should be rare where carefully-designed BMPs are implemented during and after logging. Exceptions to this generalization may occur in areas where root cohesion is critical to slope stability, in areas with highly erodible soils, and possibly in downstream depositional areas. Extreme storm events also can greatly affect erosion rates from forestry activities. Where adverse effects do occur as a result of changes in stream channel condition, these effects will persist over time periods ranging from several years to more than a century. Data on past practices may not be an appropriate indicator of present impacts, and fish populations are subject to numerous confounding factors.

Proposition 4: The potential for water quality impacts from forest management depends upon factors such as climate, soil types, topography, etc. Special control measures can be targeted where there is greater than normal risk for short-term or long-term adverse effects (such as areas with high risk of landslide).

A. <u>Background</u>

The impacts of forest practices on some parameters of water quality are relatively consistent across regions and landform, including temperature, dissolved oxygen, nutrient concentrations, and pesticide concentrations. Substantial differences among regions and landforms are likely for the effects of forest practices on stream flow, sediment concentrations, stream channel conditions, and therefore on populations of fish.

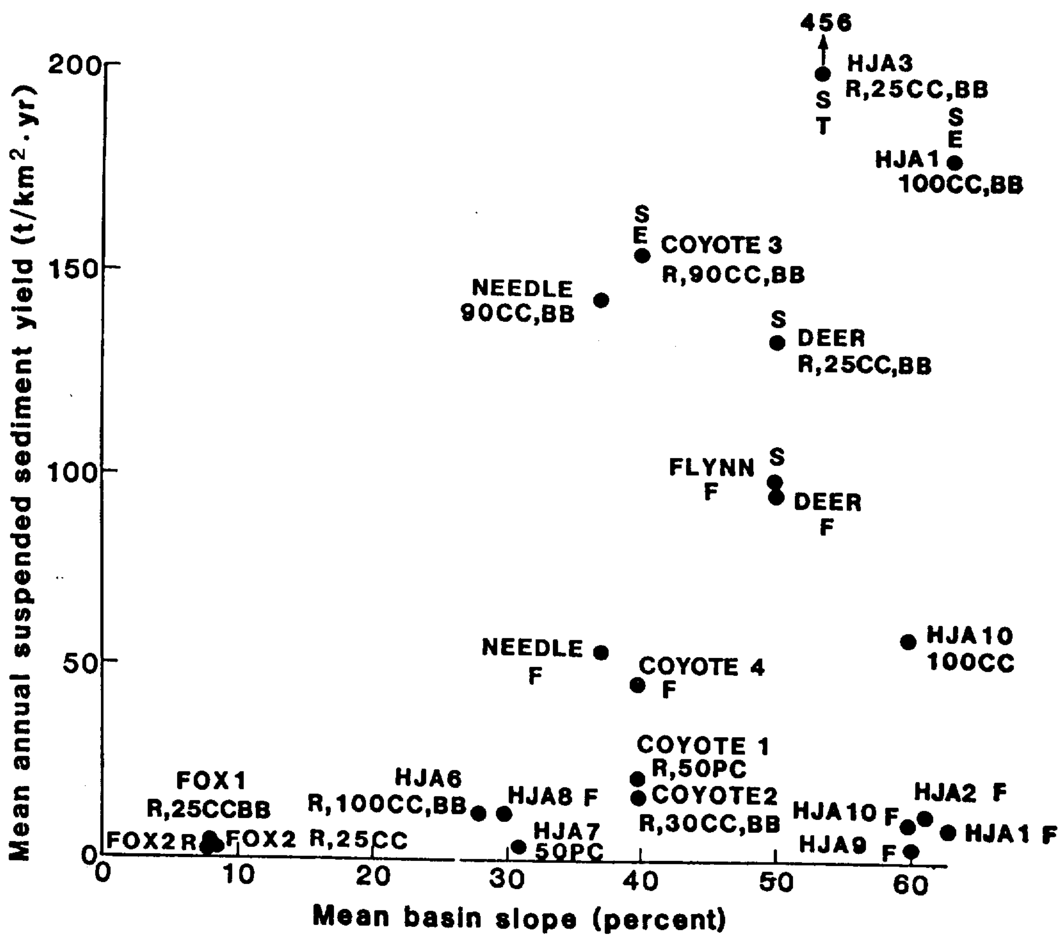
Detailed sediment budgets from four sites in California, Oregon and Washington illustrate the diversity of erosion patterns in steep, high-precipitation environments (summarized by Swanson *et al.* 1987). At the Van Duzen River, rapid debris slides move about 500 Mg km^{-2} yr^{-1}, and earth flows (rapid slumps of hillslopes) add another 150 Mg km^{-2} yr^{-1}; however, the majority of erosion results from gullies, and this was estimated at over 2000 Mg km^{-2} yr^{-1}. At Redwood Creek, rapid debris slides move about 1200 Mg km^{-2} yr^{-1}, compared with 2500 Mg km^{-2} yr^{-1} for gullies. Erosion rates are much lower for the H.J. Andrews Experimental Forest in Oregon and the Clearwater River Basin in Washington, with less than 80 Mg km^{-2} yr^{-1} of erosion from all sources. Soil creep (slow, downhill movement of intact soil) accounts for a substantial component of erosion at these two sites, but is unimportant in the California sites.

Within a single precipitation regime, sediment yield depends heavily on factors that influence erosion, including basin size, slope and the nature of soils. Swanson *et al.* (1987) compiled information on suspended sediment yields (Mg km^{-2} yr^{-1}) for over 20 small watersheds in western Oregon (Figure 5). The watersheds receive between 1200 and 2600 mm/yr of precipitation, and range from 9 to 300 ha in size. The maximum observed rates of erosion increased with increasing slope, with little response to management until slopes exceed 35%.

Note that the relationship between slope and sediment yield may not be linear (Figure 5). Bourgeois (1978) found that erosion from roads on Vancouver Island was greater for slopes between 30 and 35° than for slopes greater than 35°, probably because greater care was taken in constructing roads on steeper slopes.

Figure 5 also shows that within a single slope class at a single area, such as the 60% slope class at the H.J. Andrews Experimental Forest in coastal Oregon, there is a tremendous range in sediment yield. Watersheds HJA1 and HJA2 both yielded less than 10 Mg km^{-2} yr^{-1} when undisturbed, but logging (without roads or streamside buffer) and burning of HJA1 raised the yield to about 170 Mg km^{-2} yr^{-1} in the first year. Sediment yields declined rapidly in later years. Another watershed in this group was partially cut and burned, and the inclusion of roads raised the yield to over 450 Mg km^{-2} yr^{-1}, with substantially increased sediment export occurring for more than 3 years. In contrast, logging of HJA10 increased sediment yields to just 55 Mg km^{-2} yr^{-1}.

FIGURE 5 Suspended sediment yield for experimental basins in the Fox (FOX), Coyote (COYOTE), Alsea (NEEDLE, FLYNN, DEER) and H.J. Andrews (HJA) areas of western Oregon. S = debris slides; T = debris flows; E = active earth flows; F = forested; R = roads; 50CC = 50% of watershed clearcut; BB = broadcast burning following logging; 50PC = 50% of stand removed in partial cut (from Swanson *et al.* 1987).



These large differences in erosion and sediment yield relate in part to parent material. Below about 900 m elevation in the H.J. Andrews Forest, the volcaniclastic parent materials are very susceptible to erosion, whereas soils at higher elevations are underlain by erosion-resistant, unaltered lava flows (Swanson and Dyrness 1975). For intact forests in the unstable, low-elevation zone, erosion averages about 2,000 m^3 km^{-2} yr^{-1}, compared with about 6,000 m^3 km^{-2} yr^{-1} for clearcut watersheds, and 65,000 m^3 km^{-2} yr^{-1} annually for clearcut watersheds containing roads. (note that current road construction standards are stricter than these older, sidecast designs). For the stable zone at higher elevation, erosion is near 0 for both forested and clearcut units, and even clearcut units with roads show erosion rates of less than 1,000 m^3 km^{-2} yr^{-1}.

How effective are special control measures (including BMPs) at minimizing the impacts of forest practices on sedimentation and stream channel characteristics? In the eastern U.S., implementation of BMPs for retaining buffer strips along streams and for improving road design and construction appear adequate. For example, research at the Coweeta Hydrologic Laboratory in western North Carolina has demonstrated the effectiveness of well-designed roads, including the use of broadbased, outsloping dips, seeding grass on roadways, and in some cases adding a gravel surface (Swift 1984, 1988, Kochenderfer and Helvey 1987).

Key problem areas within the United States appear to be restricted to portions of Idaho (locations with weathered, granitic rocks), northern coastal California, western Oregon and Washington, and southeast Alaska. These areas are distinguished by low slope stability and high risk of surface erosion (from concentrated flow) and debris flows.

Intensive research on appropriate harvesting methods and road designs has developed BMPs to greatly reduce impacts on water quality in unstable terrain. Sediment yield can be reduced by up to 90% in the Idaho Batholith with the intensive use of gravel and rocks on the road surface and ditches, and protection of cutslopes (Burroughs and King 1989). In the Redwood Creek Basin in California, much of the erosion associated with roads results from inadequate designs for accommodating runoff from large storms, and a lack of road maintenance (or retirement) after logging operations are finished (Weaver *et al.* 1987).

A variety of simple, preventative measures can greatly reduce the gully erosion problem for roads (Weaver *et al.* 1987): conducting long-term maintenance of all roads; designing stream crossings carefully to minimize diversion potential (see also Furniss *et al.* 1991); performing minor reconstruction on existing, poorly-designed stream crossings; and excavating stream crossings on abandoned roads with a high potential for diversion.

The appropriate BMPs for road construction depend on the nature of the soils and slopes. Roads in the Idaho Batholith may need to be surfaced with gravel to prevent erosion of coarse granitics that are susceptible to gully erosion, whereas the risks of gully erosion for Redwood Creek in California may depend more on diverted stream crossings and debris torrents. Surfacing roads with gravel will reduce but not eliminate fine sediment production, as road traffic causes a regular "pumping" of fines to the road surface. This largely explains why Reid and Dunne (1984) found fine sediment production from gravel-surfaced roads to be extremely sensitive to road use. Through paving and other practices roads can be engineered for minimal impacts on water quality, even in high hazard areas, but economic feasibility may be limiting. We also note that well-engineered BMPs designed for a 25- or 50-year storm may fail in more extreme events.

B. Conclusion on Proposition 4

The risk of degrading water quality from forest management practices varies greatly among forests, regions, and soils. Best Management Practices capable of preventing degradation of water quality in one situation may be inadequate in another. In most cases, BMPs have been developed that can minimize degradation of water quality to within generally acceptable limits. The greatest risks and challenges occur under intense storms on either unstable slopes or steep slopes with erodible soils. Our ability to identify hazardous areas is improving (Dietrich *et al.* 1992) and the application of special BMPs has generally been successful in the Idaho batholith. However, it is important to note that extreme storm events, such as those with a recurrence interval in excess of 50 years, can degrade water quality and stream habitat in both managed and unmanaged forests, despite the careful application of BMPs.

Proposition 5: The water draining forested watersheds is not "toxic" (as defined by the Clean Water Act and elsewhere) unless affected by spills or other unusual events.

A. Background

The definition of "toxic" typically refers to a poison or poisonous effect, with a poison defined as an agent that chemically destroys life or health upon contact with or absorption by an organism. Toxic pollutants are defined as "substances which, by themselves or in combination with other chemicals, are harmful to animal life or human health" (USEPA 1985, p. 45).

The water draining from forest watersheds does not meet this definition. The majority of water quality problems associated with forests deal with issues of fish habitat in relation to sedimentation and changes in stream channel characteristics. The sediments are not toxic, as they do not harm fish except at very high concentrations; normal sediments from forests are not poisonous.

We do not mean to minimize the impacts of excessive sedimentation on fish habitat; degradation of spawning beds clearly can reduce reproductive success. However, this should be viewed as habitat degradation along the lines of conversion of forests to agricultural lands. Agricultural land uses generally do not cause toxic effects in wildlife, but they may degrade habitat for some species and lower reproductive success. In this regard, we conclude that sedimentation aspects of water quality are very important, but do not warrant classification as a toxic problem.

Herbicides and pesticides are designed to be toxic to some species, but even aerial applications do not result in toxic concentrations in forest streams (see Proposition 3).

The few cases in which high nitrate concentrations have been observed in streams following forest management activities may be exceptions to the generalization that forest management does not cause toxicity problems. Some Northern Hardwoods forests in the White Mountains of New Hampshire have high concentrations of nitrate in streamwater (especially after harvesting). Complete deforestation with suppression of forest regrowth (for experimental purposes, not resembling normal forest operations; Likens *et al.* 1970) has led to excessively high concentrations (defined as 10 mg-N/liter for drinking water). Such high levels are not common for other areas in New England (Martin *et al.* 1984). In the Pacific Northwest, small streams draining watersheds that contain nitrogen-fixing red alder forests may show very high nitrate concentrations, perhaps exceeding the drinking water standard (Miller and Newton 1983). Small streams draining high-elevation forests in the Smoky Mountains may also contain high concentrations of nitrate (Silsbee and Larson 1982).

As noted in Proposition 2, pulses of high nitrate concentrations after fertilization have been documented in a few cases. Typically these pulses are short in duration and peak concentrations of nitrate-nitrogen are lower than the criteria for drinking water. In exceptional cases (e.g., repeated fertilization with a runoff event occurring shortly after application) this criteria has been exceeded, suggesting that more stringent controls should be applied when large proportions of catchments used for domestic water supply are to be fertilized. All other reports of streamwater chemistry from forested watersheds have indicated that nitrate concentrations are well below any threshold that would warrant concerns over toxicity.

Large storm events may cause turbidity levels to temporarily exceed the criterion for drinking water, but turbidity in itself is not toxic. Turbidity is important for aesthetic purposes, and increased turbidities require higher levels of chlorination for chemical control of microorganisms.

Aluminum may be an exception to our statement that water draining from forests does not meet toxicity definitions. Aluminum is a major component of all mineral soils, and aluminum concentrations increase as the acidity of soil solutions, streams, and lakes increase (= pH decrease). Extensive studies of acid deposition have documented that the movement of aluminum from soils into aquatic systems is one of the most dramatic effects of acid rain (Sullivan *et al.* 1990, Baker *et al.* 1990). Toxic effects of aluminum begin for some organisms at concentrations as low as 1.2 mg Al (Baker *et al.* 1990), but threshold levels vary among types of organisms and stages of development. For example, newly hatched tadpoles of leopard frogs had a lethal concentration (LC_{50} for 96 hr exposure) of 3 mg Al, compared with more than 12 mg Al for 3-week-old tadpoles.

Most studies on forestry impacts of water quality have not included aluminum for several reasons. First, the majority of forest streams have low enough acidity (high enough pH) that aluminum concentrations are too low to be of concern. Second, many of these water quality studies were designed to examine nutrient losses, and aluminum is not a nutrient. Finally, the large majority of these studies were initiated before acid rain concerns focused interest on streamwater aluminum. Binkley and Brown (1993) tabulated streamwater chemistry for studies that examined responses to forest practices in the U.S., and the small watershed studies at Hubbard Brook in New Hampshire included aluminum. The control watershed averaged about 0.25 mg Al; the devegetation treatment lowered stream pH and increased aluminum to 2.0 mg/L, whereas a conventional whole-tree harvest increased aluminum to just 0.8 mg/L. The increase in aluminum in the latter resulted from increased nitrate leaching and decreased pH, and probably was too low for any biologic impact.

In the absence of data on aluminum, inferences can be made from the response of stream pH to forest practices. Increases in pH should generally indicate decreases in aluminum concentration, and vice versa. In Binkley and Brown's (1993) summary, Hubbard Brook was the only location that showed lower stream pH after harvesting; all other sites showed no pH response or an increase in pH. Increases in soil temperature and decomposition after clearcutting may increase concentrations of bicarbonate in soil solutions, moderating stream pH. The exception to this pattern would be sites that have substantial nitrate production and leaching after harvest, such as Hubbard Brook.

B. Conclusion on Proposition 5

We conclude that further consideration of the effects of forests on streamwater aluminum concentrations may be warranted in areas with low pH, low dissolved organic carbon, and high nitrate leaching. However, no impacts of forest practices have been documented for aluminum-related impacts, and we do not think this should be a high priority in water quality issues.

Proposition 6: The direct effects of forest practices on water quality within a small watershed may differ from the cumulative effects of the same practices on many small watersheds within a basin.

A distinction is often made between direct on-site effects and cumulative off-site effects of forest practices on water quality. In practice, however, fewer management actions are being allowed in or immediately adjacent to stream channels, and water quality usually is evaluated at points in streams separated from management activities by buffer strips and/or a stream segment. In many cases, including the widespread practice of upstream-downstream comparisons, it is difficult to attribute an observed change in water quality to a particular activity at a specific location.

Clear examples of cumulative off-site effects of forest management on streams are difficult to demonstrate in all but the most severe cases (Bisson *et al.* 1992b). Evaluation of off-site effects requires consideration of all management activities and natural processes occurring upstream of the monitoring location. The inherent complexity of combining and routing these different inputs greatly hinders our efforts to define and analyze cumulative effects, despite a series of symposia addressing the concept (NCASI 1986, WRC 1986, American Society of Agronomy 1989, SAF 1990).

A number of studies have documented downstream impacts attributable to some combination of forest harvest, road-building, and storm events (Grant 1986, Lyons and Beschta 1983, Megahan *et al.* 1980, Megahan and Bohn 1989, Nolan and Marron 1985). From a management point of view it is important to distinguish whether such downstream impacts can occur despite effective protection of water quality through BMPs in upstream areas.

A. Temperature

The physical processes involved in temperature increases are probably better understood and more amenable to modeling than any other water quality parameter, particularly at the reach scale (Beschta *et al.* 1987, Sullivan *et al.* 1990, MacDonald *et al.*

1991). As noted earlier, summer water temperatures can increase as a result of forest harvest and this can adversely affect habitat suitability for coldwater fish. The basic physics of solar heating and efficient mixing provides the potential for additive, cumulative effects on stream temperature. Recent work in Washington under the Timber/Fish/Wildlife (TFW) Program suggests that downstream temperature increases can be minimized by instituting sufficiently stringent controls on riparian harvest within individual upstream reaches (Sullivan *et al.* 1990). Predictions of water temperature were much more successful on a reach scale (on-site) than on a basin scale (cumulative, off-site).

There is still a need to determine the likelihood that acceptable temperature increases in upstream locations will affect downstream temperatures. Such a situation might occur when the upstream waters are so cold that a substantial temperature increase is acceptable, or through a small but still acceptable temperature increase in a large number of upstream tributaries. The balance between these temperature increases and the other heat transfer processes could best be explored through a series of simulations using existing temperature models, and this would indicate under what circumstances reach-scale controls would be adequate to prevent adverse increases in downstream water temperatures.

B. Sediment

This is the pollutant of greatest concern and greatest complexity with regard to cumulative watershed effects. Certainly there is a strong physical basis to the transport and subsequent deposition of fine sediment to downstream reaches, and the downstream deposition of sediment has been observed as a result of management activities and natural events (Grant 1986, Lyons and Beschta 1983, Megahan *et al.* 1980, 1989, Nolan and Marron 1986). If we accept that low-order streams generally are higher-gradient and have a limited supply of sediment available for transport (i.e., supply-limited), whereas higher-order streams tend to be lower-gradient and sediment-saturated (i.e., transport-limited), then the first transport-limited reaches should be most susceptible to sediment accumulation and resultant changes in channel morphology and aquatic habitat. This conceptual view also suggests that increased sediment delivery may not necessarily affect lower-order channels, so BMP evaluations cannot focus solely on on-site effects.

Our ability to evaluate cumulative effects of sedimentation is severely hampered by the lack of data and reliable models on sediment transport and storage in the downstream direction. A simple assumption would be that sediment yields decline sharply with increases in drainage area, but consideration of the physical processes suggests that downstream sediment delivery will be controlled by a variety of factors, including the size and amount of the sediment being introduced to the stream channel, channel confinement, channel gradient, and the discharge regime. The tremendous spatial and temporal variability in the generation and transport of sediment in forested areas greatly exacerbates the problems of predicting and detecting any type of cumulative effect (Grant *et al.* 1984, MacDonald 1993). Scale considerations, sediment delivery to the stream channel, and the routing of sediment through the stream network have all been identified as high priority areas for research through NCASI's cumulative effects program (NCASIb 1992), and the work initiated under this program should clarify some of the key issues.

Even though we can't accurately predict downstream impacts, the literature clearly indicates a need to minimize the additional input of sediment due to forest management activities. As noted earlier, forest harvest and particularly forest roads can increase landslide frequency, induce surface erosion, and diminish streambank stability. Increases in sediment are easily associated with adverse impacts on a variety of designated beneficial uses (MacDonald *et al.* 1991), and this suggests that a prudent approach is best. Considerable additional work will be needed before we can begin to set thresholds based on a risk of downstream effects.

C. Dissolved Oxygen

The rapid reaeration rate of most forest streams, together with the naturally high dissolved oxygen content and limited effects of forest management, suggest that there is little potential for direct cumulative effects. However, intergravel dissolved oxygen is closely tied to gravel permeability and the hence the amount of fine sediment. This means that intergravel dissolved oxygen is inextricably linked to the amount of fine sediment, and the same concerns over fine sediment and cumulative impacts also apply here. Again the relationship between dissolved oxygen, gravel permeability, and fine sediments is a complex function of factors such as relative size distributions of the gravel and introduced sediment, timing and magnitude of storm events, water temperature, and the turbulence of flow.

D. Nutrients

Forest harvest has been shown to increase nitrate concentrations, and an increase in erosion rates can increase the amount of phosphorus in streams (Saliminen and Beschta 1991). Although these increases can be substantial in relative terms, the amounts usually are small in absolute terms and the observed values – with the few exceptions noted earlier – are well below water quality standards (MacDonald *et al.* 1991, Bisson *et al.* 1992b). In cases where there is a downstream sink (e.g., a lake, reservoir, or wetland), nutrients can accumulate and accelerate eutrophication, as may have happened at Lake Tahoe (Leonard *et al*. 1979, USFS 1985, Byron and Goldman 1986). For oligotrophic lakes with high scenic and recreational value, any increase in primary productivity may be undesirable. In such cases the existing numerical water quality standards may not be adequate, and there is considerable potential for adverse cumulative effects. These potential cumulative effects can be substantially reduced by careful management practices to minimize nutrient inputs through erosion, fertilization, recreation, and grazing.

In most cases the uptake and dilution of any increase in nutrients largely eliminates any potential cumulative effects problem. Many forest streams are nutrient-limited, and some forest management practices such as fertilization may increase aquatic productivity (Gregory *et al*. 1987).

E. Pesticide Concentrations

The processes of dispersion, dilution, uptake, and degradation suggest that the potential for cumulative effects of pesticide and herbicide applications are extremely limited, especially given the small proportion of a basin that would be treated within any time period (Proposition 3). Chemicals such as DDT posed problems in the past because of bioaccumulation, but most chemicals used today are not as persistent or subject to accumulation at higher trophic levels (Norris 1991).

F. Stream Channel Conditions

Stream channel conditions, at least with regard to cumulative watershed effects, are largely a function of changes in the balance between sediment load and discharge. Forest harvest and road-building can alter the size of peak flows, duration of peak flows, annual water yield, and the size of low flows (MacDonald *et al*. 1991). Forest management activities usually increase the size and duration of peak flows, and this generally increases the amount of sediment transport (Troendle 1993) and sediment concentrations (Marion and Ursic 1993). Similarly, forest harvest almost always increases annual water yield, and this historically has been considered as a benefit. The observed increase in low flows due to forest harvest can increase water supply at times when demand is high and increase available aquatic habitat.

The importance of changes in the size and magnitude of peak flows is being debated, but several arguments suggest the primary hazard for downstream areas is sediment rather than peak flows. Two of the key reasons for this statement include the dilution and dispersion effects on peak flows in the downstream direction, and the observation that, at least in rain-dominated areas, the greatest effect is on the smaller flows (e.g., Wright *et al*. 1990). In snow-dominated areas forest harvest can substantially increase peak runoff rates (e.g., Troendle and King 1985, King 1989), but the peak runoff rates are still very low compared to rain-dominated areas (Kattelmann 1990).

The effects of forest harvest on peak flows is of greatest concern in areas subject to rain-on-snow events. This concern stems from the fact that rain-on-snow events typically generate the largest flood events (Kattelmann 1990) and an analysis of the physical processes suggests that forest harvest could further increase the largest peak flows (Coffin and Harr 1991). These changes can increase channel erosion (Harr 1981) and impact designated beneficial uses.

Rain-on-snow as a cause of major flood events has been documented for northern Idaho, western Cascades, Sierra Nevada (Kattelmann 1990) and noted at Coweeta in the southeastern U.S. Unfortunately, most work has relied on plot studies (Beaudry 1984, Berris and Harr 1987, Coffin and Harr 1991).

Analysis of streamflow records suggests an increase in the size of peak flows due to forest harvest (Anderson and Hobba 1959, Harr 1986), although subsequent analysis may lead to a revision of these results (G. Grant, USFS, pers. comm.). Hence there is a need to document the frequency and magnitude of rain-on-snow events in different regions, and then to evaluate the effects of forest harvest on the size of peak flows of different recurrence intervals. Unfortunately, generalization about the more extreme events will be difficult because they are infrequent and little data will be available. NCASI's cumulative effects program recognizes the need to further evaluate the effects of forest harvest on peak flows in areas subject to rain-on-snow events, and this should help determine whether harvest thresholds are necessary to avoid cumulative increases in the size of peak flows.

The effects of forest harvest on peak flows can be ameliorated by BMPs that minimize the amount of roads and compacted areas. Such BMPs are likely to be most effective for smaller runoff events in rain-dominated areas. Road cuts which convert subsurface flow to surface flow should also be avoided when the size of peak flows is a concern. Full mitigation for the effects of forest harvest is not possible because the removal of the tree canopy reduces interception loss, accelerates snowmelt, and increases the likelihood of a rain-on-snow event.

G. Fish and Invertebrate Populations

Cumulative effects of forest practices on fish and invertebrate populations can be postulated through the downstream changes in temperature, nutrients, sediment, dissolved oxygen, and stream channel conditions discussed above. The mobility and temporal and spatial variability of aquatic organisms make it difficult to link changes in these controlling factors to measurements of aquatic life. Therefore, cumulative effects analyses typically focus on these controlling factors as the most sensitive (and measurable) indicators. Changes in physical habitat still need to be linked to water quality standards and goals, such as the protection and successful reproduction of aquatic life. In some cases it may be sufficient to monitor selected biota, and presume that in the absence of detectable change there are no on- or off-site management effects.

H. Conclusion on Proposition 6

The cumulative effects of multiple operations within a large basin may well differ from those that occur on-site within management units. At the present time the best protection against adverse off-site or cumulative effects is to utilize BMPs that minimize on-site effects. Areas of greatest concern and highest research priority include the downstream routing of sediment, effects of forest management on the size of rain-on-snow events, and nutrient loads in downstream oligotrophic lakes. This paper has identified some of the key gaps in knowledge with regard to cumulative effects, and many of these are being addressed through NCASI's cumulative effects program.

IV SUMMARY

The quality of water draining from forested watersheds is generally the best in the nation. Forest practices, particularly road construction and harvesting, have the potential to degrade water and stream quality primarily through increased sedimentation and changes in channel conditions. Intensive research projects have generally found that implementation of BMPs can prevent substantial degradation of water quality.

Three major areas warrant additional attention: high-nitrate systems, operational-scale assessments of the effectiveness of BMPs, and the cumulative effects of management practices within a basin.

High concentrations of nitrate in streamwater are currently found only in isolated situations, such as the White Mountains of New Hampshire, the Smoky Mountains of North Carolina and Tennessee, some watersheds containing nitrogen-fixing red alder in the Northwest, and transient peaks following fertilization in some cases. High rates of nitrogen deposition from the atmosphere in the Eastern U.S. have led to a variety of discussions and predictions relative to "nitrogen saturation" of forests (e.g., Aber *et al.* 1990, Johnson 1992). Current generalizations about low nitrate concentrations in streamwater from forests may change in the next few decades if high deposition of N from the atmosphere continues. Removal of nitrogen from forests by timber harvesting might reduce N saturation and lower N concentrations in streams (see Silsbee and Larson 1982).

The effectiveness of BMPs has been proven in research settings, but only a few studies have examined their effectiveness at an operational scale (e.g., SWRCB 1987, Knopp *et al.* 1987, Harvey *et al.* 1988). Assessment of the implementation and effectiveness of BMPs in operational settings is clearly a high priority. A given BMP may be very effective at protecting water quality in one situation, but be either unnecessary or inadequate in another. The economic costs and benefits of BMP implementation warrant more evaluation (Binkley and Brown 1993).

In many cases, public concerns about forest practices involve cumulative impacts within basins. In nearly all cases, dilution, dispersion, and storage reduce the impacts of management activities downstream of the managed watersheds. High nitrate concentrations in small streams may be diluted substantially downstream if only one or several watersheds in a basin have high concentrations. Downstream effects can be more dramatic than on-site effects in some cases. For example, modest increases in suspended or stored sediments in upstream, high velocity reaches may lead to substantial deposition in downstream, lower-velocity reaches.

We also note that the risks associated with the combination of extreme events (such as storms with very long return periods) and heightened susceptibility from forest practices may also lead to "cumulative" effects. For example, a road system designed to accommodate 25-yr storms may show only minor cumulative effects in a normal year, but may show major, basin-wide failures during extreme events. Much more work is needed to clarify the likelihood of certain types of cumulative effects, including the characteristics of sensitive basins, approaches to routing water and materials through stream networks, and evaluating the combined risk of management activities and extreme events.

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