Forest Operations and Water Quality in the South

J. M. Grace III

Abstract. *Southern forests, which rely on intensive management practices, are some of the most productive forests in the U.S. Intensive forest management utilizes forest operations, such as site preparation, fertilization, thinning, and harvesting, to increase site productivity and reduce rotation time. These operations are essential to meet the ever-increasing demands for timber products. Forest managers utilize forest operations as tools in an attempt to manage the nation's forestlands for multiple uses while maintaining or improving resource quality. Forest operations can influence nonpoint-source (NPS) pollution by disturbing natural processes that maintain water quality. In recent years, NPS pollution has been identified as the nation's largest source of water quality problems. Forest management activities have been identified as activities that influence NPS pollution in the South. Results of watershed-scale studies that investigated the effect of forest operations on water quality in the 13 southern states are highly variable. However, taken collectively, the results indicate that forest operations have little impact on the quality of water draining from forests in the South. Based on this review, best management practices (BMPs) show the potential to protect water quality following forest operations; however, accurate assessments of the overall effectiveness of BMPs are not possible because the benefits of BMPs on different scales are relatively unknown.*

Keywords. *BMPs, Fertilization, Forest roads, Forestry, Harvesting, Hydrology, Nonpoint, Reviews.*

In recent years, nonpoint-source (NPS) pollution has been identified as perhaps the greatest threat to the nation's water quality (USEPA, 2003). The Clean Water Act (CWA) of 1977, a result of amendments to the Federal Water Pollution Control Act (FWPCA) of 1972, has two primary goals for the achievement of objectives set forth in the CWA: eliminate discharge of pollutants into the nation's waters, and achieve water quality levels in the nation's waters that are fishable and swimmable. Today, the majority of the nation's waters meet water quality levels set forth in the goals of the CWA. The CWA established the Environmental Protection Agency (EPA) as the chief agency responsible for permitting, enforcing, and administrating the law to states. Section 208 of the CWA identified timber harvesting and silvicultural activities as NPS pollution sources. In Section 208, states were required to establish best management practices (BMPs) for forestry related activities to reduce NPS pollution.

Section 303(d) of the CWA also established the Total Maximum Daily Load (TMDL) program in an attempt to achieve "Cleaner Waters across America." The mission of the program is to ensure healthy watersheds and public health protection. The TMDL program identifies impaired waters, determines pollution reductions required for health, and ensures corrections to reduce NPS pollution. As of 1999, 20,000 of the nation's water bodies, including 300,000 river and shore miles and 5 million lake acres, were identified as polluted (USEPA, 1999). The reduction of runoff through more efficient use of water, fertilizer, and pesticides is an action suggested by the EPA for cleaner waters while TMDLs are developed. Within watersheds on 303(d) lists, many nonpoint sources are extremely difficult to pinpoint, measure, and control due to the intertwined land use categories within a given watershed. Possible nonpoint sources of sediments in managed forested lands include harvesting, roads, log decks, skid trails, and site preparation.

Intensive management practices have been reported to influence water yield and quality. The use of intensive management practices in the South has made the region one of the most productive in the world (Prestemon and Abt, 2002). The use of intensive management practices in combination with the abundant water resources in the region increases the potential for water quality impacts. In this context, the South would likely be the optimal region to evaluate the extent and nature of the water quality impacts of forest operations. The objective of this article is to provide a review of the watershed research on the nature and extent of NPS pollution attributed to forest operations, specifically harvesting, site preparation, fertilization, and road construction and maintenance, in the South. This article also explores the role of BMPs in NPS issues related to forest operations.

NPS Pollution in the South

Major environmental concerns related to water quality exist in the 13 states (fig. 1) that constitute the southern region of the U.S. due to the 1.5 million km of rivers and streams flowing through the region. In 1998, approximately 25% of the rivers and streams flowing through the region were assessed and reported in the state water quality inventories (USEPA, 2000). Based on these inventories, 55% of the assessed rivers and streams fully supported their designated uses. The remaining 45% of assessed rivers and streams in the southern states were impaired by some form of pollution.

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Point sources of pollution (municipal, urban runoff, industrial, and land disposal activities) are the major contributors to impaired state rivers and streams in Georgia and Texas. Nonpoint sources are the leading cause of impairment to rivers, streams, and lakes due to pollution in the other states in the southern region. The NPS pollution activities include agriculture, hydrologic/habit modification, resource extraction, storm sewers/urban runoff, construction, silviculture, and natural activities. Agricultural activities are by far the leading nonpoint-source activity, accounting for 71,000 km of impaired rivers and streams in the southern region during the period from 1988 to 1998 (USEPA, 2000). Agricultural activities accounted for more polluted miles than the combination of all point sources and more than 60% of the total assessed nonpoint-source pollution (110,000 impaired km) impairing rivers and streams in the South. Silviculture accounted for 5,900 km of impaired rivers and streams and ranked 9th of the 10 leading sources of pollution of rivers and streams in the South (West, 2002). The contribution of silviculture (hereafter referred to as forest operations) to pollution of rivers and streams in the South is relatively small (8% of the total impaired rivers and streams). However, forestry operations have the potential to impact water quality and fisheries habitat (Fulton and West, 2002).

Road Construction and Maintenance

The forest road system is recognized as one of the primary risk areas in relation to NPS pollution from forest management activities. Forest road systems have been cited as a major source of sediment and eventual sedimentation in forest streams (Authur et al., 1998; Binkley and Brown, 1993; Haupt, 1959; Kochenderfer and Helvey, 1987; Packer, 1967; Patric, 1976; Trimble and Sartz, 1957; Yoho, 1980). In fact, forest roads have been identified as accounting for the majority of all forest erosion (Anderson et al., 1976; Patric, 1976; Swift, 1984a). Quantifying and mitigating the effects of forest roads on forest systems is now emphasized on both public and private forestland holdings. However, the overall effects of forest roads on water quality continue to be defined and require further study.

Understanding and mitigating the effects of forest roads on water quality requires an understanding of the factors that increase the potential for accelerated erosion losses. Factors that increase the potential for soil erosion and water quality impacts include:

- Interruption of natural watershed drainage patterns.
- Bare soil surface exposed to storm energy.
- Concentrated flow in roadside ditches.
- Alteration of natural soil structure during construction.
- Sideslopes increased beyond those naturally occurring in watersheds.
- Recurring disturbance of the road surface (traffic and maintenance).
- Reduced infiltration from compacted surfaces.
- Altered subsurface hydrology.

The alterations in drainage patterns, decreased infiltration, and greater slopes in combination with an exposed soil surface increase the potential for detachment and transport of sediments. Sediments can transport attached nutrients directly to stream systems and present additional NPS problems in forested watersheds.

Information on sediment delivery to streams resulting from construction of forest roads is lacking in the southern U.S. The majority of investigations of forest road effects primarily focus on soil erosion. Erosion rate studies have been conducted extensively throughout the southern states (Appelboom et al., 1998; Barnett et al., 1967; Blackburn et al., 1986; Beasley et al., 1986; Beasley and Granillo, 1982, 1988; Grace, 2000, 2002a, 2003; Kochenderfer and Helvey, 1987; Swift, 1984a, 1984b, 1985; Swift et al., 1993). However, few have related observed erosion rates to the quantity of sediment delivered to water systems or water quality. Sediment delivery to forest streams does not necessarily mirror erosion losses observed upslope. Down−

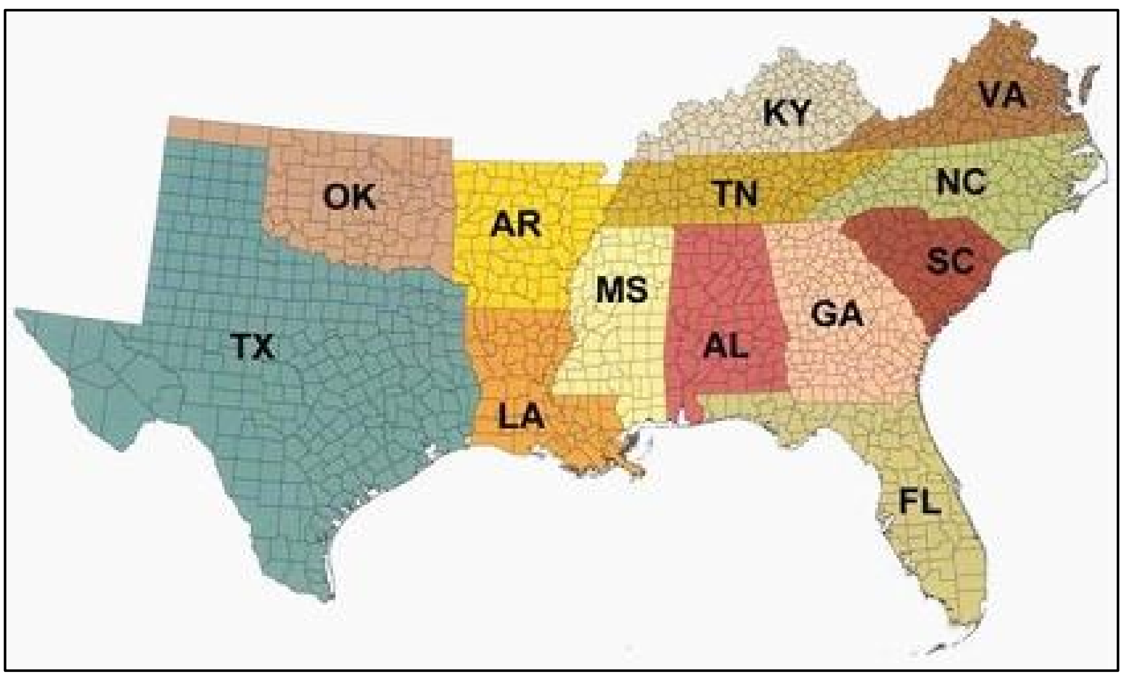

Figure 1. The 13-state region (Southern Region) considered in the review of NPS related to forest management activities.

slope sediment trapping characteristics of the forest floor and watershed topography influence infiltration of runoff and deposition of sediments suspended in road storm runoff.

Sediment delivery to streams from forest road systems within a 120 km^2 watershed was evaluated in the Ouachita Mountains in Arkansas during a 17-month period (Miller et al., 1985). Mean annual sediment loss from four segments (64 km of road) within the watershed was 55 t ha^{-1} $year^{-1}$ (with 58% suspended and 42% deposited) during the study period. The investigators estimated the quantity of sediment delivered to streams from roads in four grade classes by examining drainage structures and the downslope terrain. Based on projections, sediment delivery from roads to streams was 0.09 t ha^{-1} $year^{-1}$. This delivery was 70% of the basin-wide delivery rate of 0.12 t ha^{-1} $year^{-1}$. Estimated delivered sediment was less than observed sediment loss, and this reduction was attributed to the trapping of coarse sediments by downslope vegetation. The benefit of vegetation and obstructions in reducing sediment travel distances has been well documented in the literature (Barnett et al., 1967; Grace, 2000, 2002b; Swift, 1985, 1986).

In another study in the Ouachita Mountains, Vowell (1985) investigated erosion rates and water quality impacts of a recently established forest road. Average sediment yields ranged from 18 to 170 t ha^{-1} $year^{-1}$ with a mean of 90 t ha^{-1} $year^{-1}$. However, elevated concentrations in the receiving stream were only observed during one of eleven storms. Sediment delivered from the forest road to the stream was below the limits of detection and did not cause a measurable change in water quality.

Van Lear et al. (1997), in an assessment of the Chattooga River watershed in Georgia, documented 1100 sources of sedimentation. Based on a survey method, roads accounted for 80% of the documented sediment sources in this assessment. However, the assessment was unable to separate the contribution of roads to sedimentation and quantify sediment delivery from forest roads. In a later study, forest roads with varying maintenance levels, surfacing, and sediment control features were evaluated in the Chattooga River watershed (Clinton and Vose, 2003). Total suspended solid concentrations for unpaved surfaces were much greater than background levels (110 mg/L) of the undisturbed reference locations. Suspended solids were similar for graveled roads with routine maintenance (1500 mg/L) and high maintenance with sediment control features (2000 mg/L). Paved road surfaces had the least total suspended solids with a mean of 150 mg/L. However, a portion of the difference between surfacing types was attributed to the topography, the forest floor characteristics, and soils.

One of the primary road sediment mitigation principles involves locating roads at adequate distances from streams to allow deposition of sediments before sediments can reach streams (Burroughs and King, 1989; Haupt, 1959; Swift, 1986; Trimble and Sartz, 1957; Van Lear et al., 1997). Roads in close proximity to streams have increased potential for NPS pollution by sediments. However, stream crossings are sometimes unavoidable due to topography, practicality, and feasibility. Stream crossings have increased potential for sediment delivery to forest streams due to the interaction between water and roads. These crossings can come in the form of low-water crossings (fords), log crossings, culverts, and bridges, the latter having the greatest economic costs and least NPS pollution by sediments (Thompson et al., 1996). Construction of a low-water crossing in the Alabama Piedmont resulted in a 2800 mg/L increase in sediment concentration. Vehicle traffic in this study caused peak sediment concentration to increase by 260, 110, and 45 mg/L at 20, 45, and 92 m sampling locations downstream, respectively (Thompson et al., 1996). The investigators concluded that elevated sediment concentrations decreased with distance from low-water crossings and dissipated within 1 h following disturbance.

Harvesting

Sediments are perhaps the greatest risk to water quality following harvesting operations. Sediments can transport attached nutrients directly to stream systems. In addition, suspended sediments have the potential to degrade water quality by altering light penetration into water bodies, which alters photosynthetic fixation of energy by aquatic plants (Kirk, 1994). Increased turbidity also has the potential to reduce visual clarity, which affects the behavior of visual predators in aquatic ecosystems and influences aesthetic quality (Davies-Colley and Smith, 2001). The greatest quantity of sediment export observed was less than 1.5 t ha^{-1} $year^{-1}$ following a clear-cut harvest and site preparation (Beasley and Granillo, 1982). However, a clear-cut harvest without BMPs, which represents a worst-case scenario, exported less than 1.0 t ha^{-1} of sediment during the first 17 months (Authur et al., 1998). Despite the disregard for BMP practices, sediment export observed in the investigation was much less than typically observed from agricultural practices.

In the studies in this review, the influence of harvesting on suspended sediments was highly variable. Sediment concentrations were elevated in treatment watersheds for some watersheds and decreased in others (table 1). Perhaps of greater importance than elevated sediment concentration is the quantity of sediment exported following harvesting due to increases in water yield. The primary hydrological influence of harvesting and thinning is increased water yield due to decreased evapotranspiration. These water yield increases alone do not present a significant environmental concern; however, when combined with sediment and nutrient concentrations following harvesting, they may result in increased pollutant export. Increases in water yield for harvest treatments ranged from 69 to 210 mm/year for the studies in this review of watershed effects related to forest operations (table 1). Water yield increases as a result of forest canopy removal have been well documented over the past 40 years (table 1) (Beasley and Granillo, 1982; Blackburn et al., 1986; Douglass and Swank, 1972; Douglass et al., 1982; Grace and Carter, 2001; Grace et al., 2003; Hewlett et al., 1984; Hibbert, 1966; McBroom et al., 2002; Riekerk, 1983; Swindel et al., 1983a, 1983b; Van Lear et al., 1985; Williams et al., 1999). The magnitudes of these water yield increases vary considerably from watershed to watershed depending on factors such as soils, topography, climate, and forest type. Hibbert (1966), based on a worldwide review of watershed studies (39 studies) of the effect of canopy removal on water yield, presented an upper limit increase of 4.5 mm/year for each percent reduction in forest canopy. The majority of treatments in the review produced less than 2.3 mm/year, and results of treatments were largely unpredictable. Similarly, Neary et al. (1982) found water yield increases of 2.5 mm per

Table 1. The effects of harvesting on water yield and quality based on studies in southern states.

Region	Location	Area (ha)	Primary Forest Cover	Prescribed Treatment	Change in Constituent[a]	Increase in Water Yield	Reference
Appalachian Highlands	Coweeta, North Carolina	140	Mixed deciduous hardwoods	Total of 66% removal consisting of a 77 ha clear-cut and a 40 ha thinning	NR[b]	83 mm/year (5%) over the first seven years	Hibbert, 1966
Coastal Plain Lower	Starke, Florida	64	Slash pine (*Pinus elliottii* Engelm.), longleaf pine (*Pinus palustris* Mill.), and pond cypress (*Taxodium distichum* (L) Rich.)	Clear-cut harvest, chop, bed, and plant	NO_3-N −0.01 mg/L NH_4-N −0.17* mg/L PO_4-P 0.00 mg/L Sed 2.3 mg/L	69 mm (1st year); −32 mm (second year)	Riekerk, 1983
Coastal Plain Lower	Starke, Florida	48	Slash pine, long-leaf pine, and pond cypress	Clear-cut harvest, stump removal, burn, windrow, disk, bed, and plant	NO_3-N 0.03 mg/L NH_4-N 0.10 mg/L PO_4-P −0.01 mg/L Sed 11.7* mg/L	150 mm (1st year); −33 mm (second year)	Riekerk, 1983
Cumberland Plateau	Southeast Kentucky	NR	Deciduous hardwood	Clear-cut harvest with BMPs	NO_3-N 3.5* mg/L PO_4-P −0.05 mg/L Sed 500* kg/ha	180 mm with BMPs (first 17 months)	Authur et al., 1998
				Clear-cut harvest without BMPs	NO_3-N 3.3* mg/L PO_4-P 0.02 mg/L Sed 1200* kg/ha	210 mm without BMPs (first 17 months)	
Piedmont	Putnam County, Georgia	NR	Loblolly pine (*Pinus taeda*), shortleaf pine (*Pinus echinata* Mill.), and deciduous hardwood mixed	Clear-cut harvest	NO_3-N −0.09 mg/L	190 mm/year (first two years)	Hewlett et al., 1984
Coastal Plain Upper	Alto, Texas	3.0	Shortleaf pine (*Pinus echinata* Mill.) and deciduous hardwood mixed	Clear-cut harvest	NO_3-N −0.02 mg/L	NR	Blackburn et al., 1986
Piedmont	Clemson Forest, South Carolina	0.4-2.2	Loblolly pine (*Pinus taeda* L.)	Prescribe burning followed by clear-cut harvest	(Year 1 results) NO_3-N 0.01 mg/L NH_4-N 0.00 mg/L PO_4-P 0.00 mg/L Sed 50* mg/L (0.13 t ha^{-1} $year^{-1}$)	>150%	Van Lear et al., 1985
Coastal Plain	Monticello, Arkansas	2.3-4.0	Loblolly pine, shortleaf pine, and deciduous hardwood mixed.	Clear-cut harvested, site prepared, and planted	Sed 170 mg/L[c] (1.3 t/ ha)*	120 mm (first year)*	Beasley and Granillo, 1982
				Selectively harvested to achieve uneven-aged stand	Sed −13 mg/L[c] (0.0 t/ha)*	40 mm (first year)	
Coastal Plain Upper	Lexington, Tennessee	0.2-0.6	Loblolly pine	Clear-cut harvested	Sed 100 mg/L*	NR	McClurkin et al., 1985
Piedmont	Patrick County, Virginia	1.6-3.6	Mixed pine hardwood forest	Clear-cut harvested.	NH_4-N 0.71 mg/L* NO_3-N 0.70 mg/L* PO_4-P 0.03 mg/L Sed 328 mg/L*	NR	Fox et al., 1983

[a] Change in water quality parameter over that of the experimental control (i.e., treatment value − control value). Values in parentheses are differences in exports or fluxes for the given nutrients in the associated studies; "*" indicates a statistically significant difference in comparison to control treatment in the specified study.
[b] NR = not reported.
[c] Discharge-weighted sediment concentrations.

percent of forest canopy removed in humid regions. Equations incorporating additional factors describing the effect of canopy removal on water yields have been developed for a more detailed description of hydrologic influences (Swank et al., 1988).

Harvesting can also elevate nutrient concentrations of water flowing from treated watersheds in comparison to undisturbed controls; however, responses are highly variable (table 1). The primary nutrient concentrations of concern related to forest practices are phosphate and nitrate. Phosphate is of concern because elevated concentrations can result in eutrophication of estuaries and freshwater lakes. A phosphate concentration standard of 0.1 μg/L was established to protect estuaries, and a threshold of 0.5 mg/L is considered acceptable to protect freshwater lakes (MacDonald et al., 1991). Elevated nitrate concentrations greater than the drinking water standard (>10 mg/L) are of concern due to drinking water risks for infants.

Of the ten watershed studies considered here, only two found significant increases in nitrate concentrations following harvesting. Elevated nitrate concentrations were reported from three paired watersheds located in the Robinson Forest within the Cumberland Plateau in southeastern Kentucky (Authur et al., 1998). Nitrate concentrations during a

Table 2. The effects of site preparation on water yield and quality based on studies in southern states.

Region	Location	Area (ha)	Primary Forest Cover	Prescribed Treatment	Change in Constituent[a]	Increase in Water Yield	Reference
Coastal Plain Upper	Mississippi	0.7-1.0	Shortleaf pine (*Pinus echinata* Mill.) and deciduous hardwood mixed	Three separate site preparation treatments:			Beasley, 1979
				1. Brush chopping	Sed 344 mg/L (10 t/ha)* Sed 277 mg/L (2.0 t/ha)	480 mm (first year); 320 mm (second year)	
				2. Shearing and windrowing	Sed 710 mg/L (11 t/ha)* Sed 401 mg/L (1.9 t/ha)	420 mm/ (first year); 250 mm (second year)	
				3. Bedding on the contour	Sed 462 mg/L (12 t/ha)* Sed 1950 mg/L (4.9 t/ha)	480 mm (first year); 210 mm (second year	
Coastal Plain Upper	Alto, Texas	3.0	Shortleaf pine and deciduous hardwood mixed	Two separate site preparation treatments:			Blackburn et al., 1986
				1. Shearing, windrowing, and burning	Sed 2030 mg/L (2.9 t/ha) Sed 109 mg/L (0.07 t/ha)	120 mm (first year)*; 40 mm (second year)*	
				2. Roller chopping and burning	Sed −60 mg/L (−0.01 t/ha) Sed −34 mg/L (0.00 t/ha)	57 mm (first year)*; 24 mm (second year)*	
Ouachita Highlands	Central Arkansas	0.5-13	Shortleaf pine and deciduous hardwood mixed	Site preparation burn and plant	NO_3-N −0.38 mg/L NH_3-N 0.20 mg/L*	NR	Lawson and Hileman, 1982
Upper Piedmont	Chattahoochee National Forest, Georgia	0.85-1.09	Shortleaf pine and deciduous hardwood mixed	Site preparation herbicide application	NO_3-N 870 mg/L (<0.01 t/ha)* NH_4-N −110 mg/L (<0.01 t/ha) PO^3_4 –P −100 mg/L (<0.01 t/ha) Sed 25 mg/L (0.04 t/ha)	660 m^3 or approx. 100 mm (first two years)	Neary et al., 1986
Appalachian Highlands	Nantahala National Forest		Mixed pine hardwood forest	Site preparation burn	NO_3-N 0.07 mg/L	NR	Knoepp and Swank, 1993
Ouachita Highlands	Hot Springs, Arkansas	150-325	Loblolly pine (*Pinus taeda* L.) and pine-hardwood forest	Urea (437 kg/ha) and diammonium-phosphate (DAP) (140 kg/ha) fertilizer application	NO_3-N 2.0 mg/L (first two months)	NR	Liechty et al., 1999
Appalachian Highlands	Nantahala National Forest		Oak-pine forest	Stand replacement burn	NO_3-N no measurable effect	NR	Clinton et al., 2003
				Fell and burn	NO_3-N 0.07 mg/L (first seven months)		

[a] Change in water quality parameter over that of the experimental control (i.e., treatment value − control value). Values in parentheses are differences in exports or fluxes for the given nutrients in the associated studies; "*" indicates a statistically significant difference in comparison to control treatment in the specified study.

17-month period following treatment increased from 1.0 mg/L on the unharvested control to an average of 4.5 mg/L from clear-cut watersheds. These increases in nitrate concentrations in the Kentucky watersheds returned to control levels within a short period (2 years). Fox et al. (1983) also reported elevated nitrate concentrations (0.70 mg/L greater than the control) following clear-cut harvesting in the Virginia Piedmont. However, nitrate concentrations were well below the drinking water standard of 10 mg/L (USEPA, 1986) in each of the watersheds following harvesting. Based on the results of studies in this review (table 1) taken collectively, harvesting does not appear to adversely impact water quality of waters in southern states.

Site Preparation

Site preparation is commonly used by forest managers to reduce rotation time (Gent et al., 1983). Site preparation typically prepares the soil to facilitate planting and control vegetative competition. However, site preparation has the potential to increase sediment and nutrient concentrations by exposing soil for detachment and transport (Beasley, 1979, 1982; Blackburn et al., 1986; Edwards and Larson, 1969; Harr and Fredriksen, 1988; Schoch and Binkley, 1986; Ursic, 1979; Van Lear and Danielovich, 1988; Yoho, 1980). The extent of soil erosion and potential NPS pollution is largely dependent on site preparation treatments (Beasley, 1979; Blackburn et al., 1986; Grace and Carter, 2001; Switzer et al., 1978).

Mechanical methods (i.e., shearing, plowing, ripping, raking, chopping, bedding, and windrowing) scarify the surface and expose mineral soil to the energy of raindrop impact. In addition, mechanical methods remove much of the litter layer and debris, which can increase the erosion energy in surface runoff. Site preparation burning also increases potential for sediment and nutrient losses by removing forest floor cover and exposing soil. Mechanical, burning, and chemical site preparation methods result in the removal of forest vegetation, which typically results in increased water yields, soil moisture, and solar radiation on the soil surface. These changes can initiate accelerated decomposition, mineralization, and weathering processes, thereby increasing mobile nitrate (Clinton et al., 2003; Knoepp et al., 2004; Knoepp and Swank, 1993; Vose and Swank, 1993) and phosphate carrier anions (Johnson et al., 1986). An increased pool of nutrients in combination with increased water yield resulting from vegetation removal can translate to increased nutrient and sediment export from treated watersheds. Yet, research shows that the effects of site preparation on sediment and nutrient loss are highly variable (table 2).

Water yield increases similar to those reported above for harvesting were observed following site preparation prescriptions. A water yield increase of 480 mm was reported by Beasley (1979) during the first year for brush chopping, shearing and windrowing, and bedding on contour in the Mississippi coastal plain. Water yields during the second year continued to be elevated for each of the treatments, ranging from 210 to 320 mm. Beasley (1979) also reported significant increases in sediment concentrations in the water draining from site preparation treatment areas. The increased water yield, in combination with elevated sediment concentration, resulted in significant increases in sediment export from treatments in comparison to the control in Beasley's experiment. Similarly, water yield increases have been reported during the first two years following site preparation treatments in Texas (Blackburn et al., 1986) and in the Piedmont of Georgia (Neary et al., 1986) (table 2). However, sediment concentrations and exports were not significantly increased in the two above-mentioned studies. Sediment losses were within the range typically observed from undisturbed forest lands in the region (<0.30 t ha^{-1} $year^{-1}$).

Nutrient concentrations of water draining from the site prepared watersheds in this review are also highly variable. In central Arkansas, Lawson and Hileman (1982) reported no significant impact on nitrate concentrations and a significant increase in ammonia-nitrogen (NH_3-N) concentrations (0.20 mg/L) following burning and planting. Conversely, Neary et al. (1986) reported significant increases in nitrate concentrations (870 mg/L) following herbicide application in the upper Piedmont of Georgia. The significant increase in nitrate concentrations in the above study translated to less than 0.01 t ha^{-1} of nitrate export.

Based on the studies in this review, water yield increases are likely following mechanical, chemical, and burning site preparation treatment, although water yield increases do not necessarily translate to degraded water quality. Sediment and nutrient concentrations increased for some studies while remaining constant or decreasing for other studies. Sediment concentrations did reach levels greater than 500 mg/L in the majority of the studies reviewed, but this increase was typically a short-lived response following treatment. Nitrate concentrations remained below the drinking water standard for the studies in this review.

FERTILIZATION

Similar to the results of harvesting and site preparation studies, the results of studies of the effects of fertilization on forest water quality vary considerably. The majority of these studies have been conducted outside of the southern U.S. In studies of six watersheds in Oregon and Washington, Fredriksen et al. (1975) found peak nitrate concentration increases averaging 0.37 mg/L following fertilization with 225 kg N/ha as urea. However, the conclusion from these investigations was that stream water concentrations were not raised to degrading levels by fertilization.

Investigations conducted at the Fernow Experimental Forest, which borders the region considered by this review, in West Virginia are in contrast to the Pacific Northwest studies. Peak NO_3-N concentrations were 16 mg/L, exceeding the drinking water standard (10 mg/L), following fertilization with 225 kg N/ha as urea. Similarly, in the Fernow Forest, peak NO_3-N concentrations exceeded the drinking water standard for three weeks following fertilization with 340 kg N/ha as ammonium nitrate and 100 kg P/ha as triple super phosphate (Edwards et al., 1991; Helvey et al., 1989). Concentrations (NO_3-N, Ca^{++}, and Mg^{++}) from treated watersheds in both these investigations were detected as greater than the control watersheds up to three years following fertilizer application.

In one of the few studies in the southern U.S., Liechty et al. (1999) investigated fertilization with 440 kg N/ha as urea and 140 kg P/ha as diammonium-phosphate on a 150 ha watershed in the Ouachita Mountains of Arkansas. Total organic N (TON) concentrations showed a dramatic increase (to its maximum level) 5 h following urea fertilization (from 0.3 to 45 mg/L). Similarly, NH_3-N concentrations peaked within 24 h after urea application (at 4.9 mg/L). NO_3-N concentration response was not as immediate as TON and NH_3-N. NO_3-N concentrations began to increase following urea application and peaked at 3.6 mg/L nearly 50 days after application. NO_3-N concentration elevations were also observed downstream (in the 2270 ha watershed) of the fertilized watershed during the 1-month period immediately following urea fertilization.

ROLE OF BMPS

Reviews of BMP guidelines for states in the southern U.S. have reported differences related to forestry activities (Blinn and Kilgore, 2001; Grace, 2002c; Stringer and Thompson, 2000, 2001). Perhaps the greatest difference in BMP programs pertains to legislation for BMPs. Grace (2002c) reviewed BMP guidelines for the 13-state region and reported differences in regulatory legislation and evaluation standards. For example, with the exception of Kentucky, Georgia, and North Carolina, states in the region have voluntary (non-regulatory) forestry BMP guidelines. Kentucky is the only state in the region with comprehensive laws regarding forestry BMPs enacted in 2000. North Carolina and Georgia BMPs are quasi-regulatory, that is, having components that are both regulatory and non-regulatory in nature.

Reported compliance with both regulatory and non-regulatory programs across the region is high (>80% in Arkansas, Florida, and South Carolina) and expected to continue to increase as BMP awareness increases (Adams, 1998; Eagle and Hameister, 2002; Vowell and Lima, 2002). The trend in BMP compliance and implementation is the key to further reducing the impacts of forest operations on water quality. Vowell (2001) concluded that proper application of BMPs can provide adequate protection to water quality based on stream bioassessment effectiveness studies. State effectiveness monitoring programs seem to support this conclusion and indicate that BMPs are effective in protecting water quality when properly applied. For example, in perhaps the most complete BMP effectiveness study in the region, Vowell and Lima (2002) reported 97% compliance for all forest operations in 2001 for Florida. Compliance to BMPs in Florida has increased from around 85% in the early 1980s to 97% in 2001.

Forest operations account for only a small fraction of NPS pollution problems in the southern U.S. However, BMPs for forestry activities may be essential to avoid potential and mitigate existing NPS pollution problems. The effect of BMP practices on protecting water quality from clearcutting in the Cumberland Plateau of southeastern Kentucky was studied by Authur et al. (1998) (table 1). The investigation revealed no significant differences in sediment export from a watershed protected with BMPs and a watershed without BMPs during a 17-month post-treatment period. Both watersheds had significantly elevated sediment exports during this 17-month period in comparison to the untreated control. However, results reported in relation to NO_3-N exports indicated that buffer strips on the BMP-protected watershed may have played a role in reducing NO_3-N impacts. In contrast to results of the study by Authur et al. (1998), BMPs resulted in a ten-fold reduction in suspended sediments following harvesting and no significant changes in nitrate concentrations in the South Carolina Piedmont (Williams et al., 1999). Conclusions from the Williams et al. (1999) study suggested that with the exception of sediment concentrations, watersheds with and without BMPs resulted in high water quality.

IMPLICATIONS

Forest operations, as with any land disturbing activity, have the potential to impact water quality by NPS pollution. This potential is derived from disturbing forest cover, addition of chemicals to the system, and disturbance of the forest floor, which can alter the processes that protect water quality. Numerous small-scale studies (small-plot technology, individual road drainage areas, and point assessments) have been conducted throughout the South on the three practices with the greatest potential for impacts (roads, harvesting, and site preparation). Small-scale investigations have been undertaken to evaluate the effects of BMP treatments on roads (Grace, 2002a, 2003; Hewlett and Douglass, 1968; Hursh, 1939, 1942; Kochenderfer and Helvey, 1987; Swift, 1984a, 1984b, 1985), and harvesting and site preparation (Field and Carter, 2000; Grace and Carter, 2001). However, the effects found from these small-scale studies are not necessarily observable on the watershed scale. Small-scale studies are essential to evaluate the relative effectiveness of practices by removing confounding variables in experiments. However, it appears inappropriate to use small-scale comparisons of treatments for an assessment of water quality effects of forest operations. The forest floor storage, high infiltration rates, detention time, roughness, and flow obstructions existing on the watershed scale are not accounted for in small-scale studies and provide additional protection to water quality.

The results of watershed water quality studies on the impact of selected forest operations on water quality are as varied as the physiographic regions contained in the 13 southern states considered in this review. Clearly, the relative contribution of forest activities to NPS pollution in the region is small in comparison to agriculture and generally all other NPS activities in the South. This is evident by forest operations ranking 9th out of the 10 leading NPS activities in the South (West, 2002). However, the small contribution of forest activities within the southern region does not necessarily indicate that NPS issues should be ignored. In fact, further reductions in forest operations contribution to NPS pollution should be the goal to maintain water quality and ensure sustainable forest management practices.

SUMMARY AND CONCLUSIONS

Based on this review of watershed research related to the impacts of forest operations on water quality, forest operations can impact the quality of water draining from southern forests. Forest road activities have frequently been cited as one of the major contributors to soil erosion and sedimentation on forestlands. Few studies have related observed erosion rates upslope to sediment delivery to stream systems. For example, a sediment yield of 90 t ha^{-1} $year^{-1}$ was reported in the Ouachita Mountains below a newly established road, but only one of eleven storms showed evidence of elevated stream sediment concentrations. Clearly, a gap exists in the understanding of the overall effects of forest roads on water quality. Further information is needed on sediment delivery to streams in the various regions of the South before reliable conclusions can be made on the impacts of road activities on the water that drains southern forests.

Increased water yields as a result of harvesting are the primary influence of harvesting on waters that drain southern forests. Only two of the ten studies reviewed found significant impacts to forest water quality. Elevated nitrate concentrations were reported in the Cumberland Plateau of southeastern Kentucky (Authur et al., 1998) and in the Virginia Piedmont (Fox et al., 1983) following harvesting operations. Based on the results of studies in this review (table 1) taken collectively, harvesting does not appear to adversely impact water quality in the 13 southern states considered in this review.

Water yield increases can be expected following mechanical, chemical, and burning site preparation treatment, although water yield increases do not necessarily translate to degraded water quality. Sediment and nutrient concentrations increased for some studies while remaining constant or decreasing for other studies. Similar conclusions to those drawn for harvesting can be drawn for site preparation and fertilization in regards to water quality impacts in the South.

Based on this review, BMPs appear to have the capacity to mitigate impacts of forest operations on water quality. Two of the three studies in this review reported significant reductions in sediment concentrations from watersheds with BMPs compared to unprotected watersheds. In addition, investigations have reported the effectiveness of BMPs at the plot scale (small-scale) in reducing the impacts of forest activities. However, there is a gap in the understanding of how plot-scale benefits translate to the watershed scale. Information on the actual effectiveness of BMPs for forest operations on the watershed or landscape scale is virtually unknown. Research needs to focus on the effectiveness of BMPs at different scales (i.e., watershed and landscape scale) for effective evaluations of benefits to water quality in southern forests.

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