Chapter 19.

Influences of Management of Southern Forests on Water Quantity and Quality

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Abstract—*Water is a key output of southern forests and is critical to other processes, functions, and values of forest ecosystems. This chapter synthesizes published literature about the effects of forest management practices on water quantity and water quality across the Southern United States region. We evaluate the influences of forest management at different temporal and spatial scales, and we recognize the heterogeneity of forest ecosystems; e.g., wetlands and uplands in the South. Hydrologic models that were developed specifically for southeastern forests were reviewed. We conclude that the greatest streamwater yield or ground-water table changes occur immediately following forest land disturbances. The overall water-quantity impact of silvicultural operations on wetlands is much less than in areas having greater relief and shallow soils. Water quality from forested watersheds is the best when compared to that from other land uses. Silvicultural practices in the South caused relatively minor water-quality problems. Roads without best management practices (BMP) are the major source of sedimentation. Studies on the cumulative effects of land use changes on water quality are lacking. Exiting computer modeling tools are useful but limited in describing the forest hydrologic processes and providing practical guidance in designing forest BMPs. Recommendations to future research on forestry BMPs and forest hydrology in general are proposed.*

INTRODUCTION

Water is a key output of southern forests and is critical to other processes, functions, and values of forest ecosystems. Most of the drinking water in the South comes from forested watersheds. Much of our current understanding of the linkages between southern forest management and water quantity and quality is derived from long-term watershed-scale experiments conducted in more than 140 small watersheds in various physiographic regions in the 13 Southern States (Chang 2002). This chapter synthesizes published literature about the effects of forest management on water quantity and water quality across the region. We evaluate the influences of forest management at different temporal and spatial scales. We recognize the heterogeneity of forest ecosystem, e.g., wetlands and uplands, as affected by climate, geology, and topography. We identify sensitive regions and discuss the effects of management activities on the timing of hydrologic responses across the physiographic regions of the South. We scale up information derived from experiments at field and watershed scales to the regional level. Forest management practices examined in this chapter include harvesting, site preparation, bedding, surface drainage, road building, fertilizer and herbicide applications, and fire management. A review of regional hydrologic characteristics across nine physiographic regions is used as a framework to contrast effects of various management practices on key water-quantity and water-quality variables. Hydrologic models that were developed specifically for southeastern forests are reviewed.

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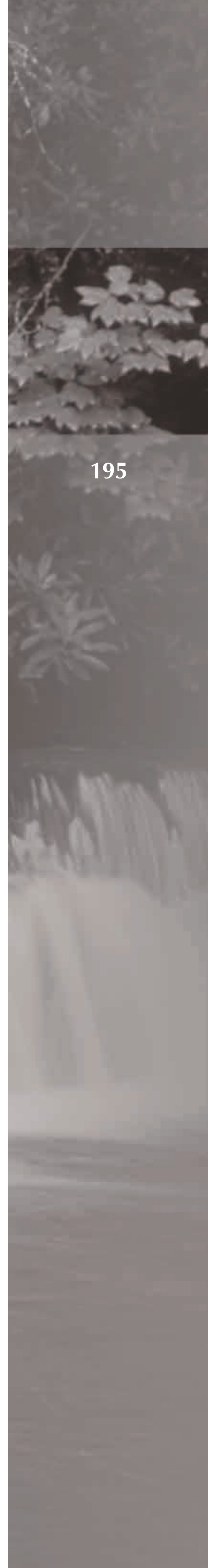

Recommendations for future research on forestry best management practices (BMP) and forest hydrology in general are proposed.

CLIMATE, TOPOGRAPHY, AND HYDROLOGY

Climate, underlying geology, and topography are the major factors that dictate regional hydrologic patterns, soil development, and forest structure and functions. Any attempts to project the impacts of management practices on hydrology must consider these factors as a background. Most southern forests are located in the climate system described as humid forest climate with cool winters and warm-to-hot summers (Muller and Grymes 1998). However, topography and elevation in the Southern United States alter this pattern greatly and result in a variety of hydrologic conditions (Daniels and others 1973).

The South can be described in various ways to facilitate characterization of forest hydrologic conditions in the region and forces that drive them. Bailey's ecoregion classification system is useful in this connection (Bailey 1995) (fig. 19.1). Bailey describes 10 major provinces that intersect with the southern geographic region, and each of these provinces has its unique hydrologic characteristics as affected by climate, topography, soils, and vegetation. We use nine major ecoregion provinces that support forest ecosystems as a spatial framework for describing the general hydrology in the South. We refer to information obtained from regional long-term hydrometeorological databases (Wolock and McCabe 1999) in discussing total annual runoff amount, runoff:precipitation ratio, and seasonal distribution of runoff (table 19.1).

Seasonal dynamics of stream runoff depend on the balances of precipitation input, evapotranspiration (ET) loss, and soil moisture storage capacity. Long-term monthly hydrologic data from four forest sites show that both climate (precipitation and potential ET) and topography (upland vs. flatland) are critical in determining the seasonal distribution of streamflow (figs. 19.2A through 19.2D). Across the Southeastern United States, potential ET is generally higher than precipitation in annual total, and in summer, but lower in other seasons. Therefore, streamflow is highest in winter and lowest in summer. Flatlands have much lower total flow than hilly uplands in summer. Variations of hydrologic characteristics described define all aspects of the water quantity and quality responses to management and design of BMPs (table 19.1).

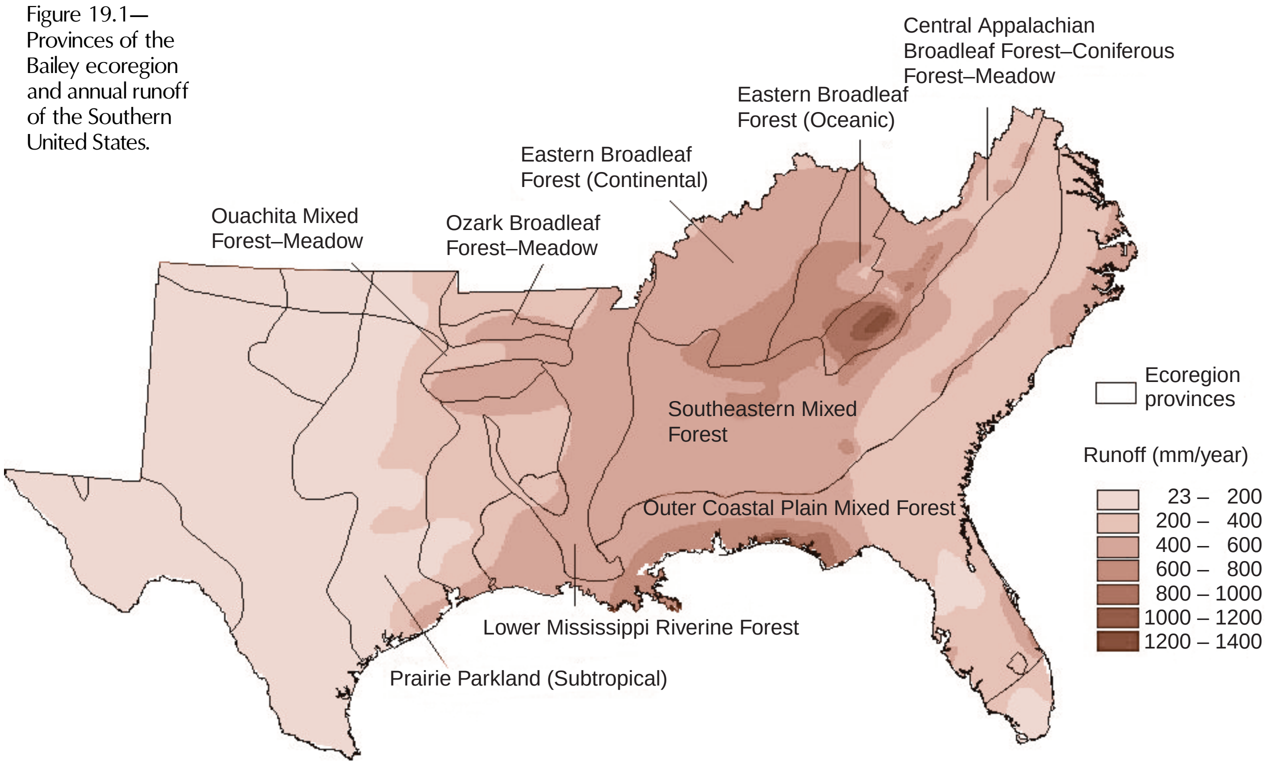

Figure 19.1—Provinces of the Bailey ecoregion and annual runoff of the Southern United States.

Table 19.1—Hydrologic characteristics of eight major physiographic regions in the Southern United States

Physiographic province	Topography	Climate and hydrology	Forest management concerns
Outer Coastal Plain Mixed Forest	Poorly drained flat lands with elevation < 100 m above sea level	High precipitation (PPT) (> 1200 mm/year) and warm temperatures, high evapotransipration (ET) (> 65 percent of PPT), variable runoff amount (300–800 mm/year), low runoff ratio (< 30 percent in most of area) and slow-moving streams	Impacts on forested wetland hydroperiod and water quality; artificial surface drainage essential
Southern Mixed Forest	Covers upper Coastal Plain and Piedmont, elevation 100–300 m	PPT (1200–1400 mm/year) moderate runoff amount (200–800 mm/year) and runoff ratio (20–60 percent)	Sedimentation caused by new forest disturbance and historical land uses; channel erosion common
Central Appalachian Broadleaf Forest—Coniferous Forest—Meadow (Blue Ridge)	Steep hillslopes at high elevations (300–2000 m)	PPT (1100–2000 mm/year), high runoff (400–1200 mm/year) and runoff ratio (> 50 percent); low ET due to low air temperature	Sedimentation caused by new forest road construction and historical land uses; fish habitat
Eastern Broadleaf Forest (Oceanic)	Rugged topography with elevation 300–800 m; long, narrow valleys and ridges	PPT (1000–1600 mm/year), high runoff ratios > 40 percent	Sedimentation caused by hardwood forest harvesting
Eastern Broadleaf Forest (Continental) (Cumberland Plateau)	Dendritic drainage with winding narrow-crested ridges and deep, narrow valleys; elevation 100–400 m	PPT decreases from 1400 to 1000 mm/year northward, runoff amount (500–700 mm/year) and ratios (35–45 percent) decline westward	Sedimentation caused by road and harvesting operations
Lower Mississippi Riverine Forest	Alluvial plains, elevation < 50 m	PPT (1200–1500 mm/year) with a decreasing south-north gradient; low runoff amount (450–550 mm/year) and runoff ratio (< 35 percent)	Harvesting impacts on wetland water quality and site productivity
Ouachita Mixed Forest—Meadow and Ozark Broadleaf Forest—Meadow	Rugged topography with relief over 300 m and narrow valleys and floodplains	Precipitation decreasing northwestward from1500 to 400 mm/year with runoff (400–550 mm/year) and runoff ratio (> 45 percent)highest at high elevations in the Ouachita and Boston Mountains	Effects of forestry activities on water quality and site productivity
Prairie Parkland (Subtropical)	Flat, with elevation < 5 m in the Coastal Plain to 500 m in the western Great Plains	Low but variable precipitation decreases dramatically westward (700–1100 mm/year); high ET, low runoff (50–300 mm/year) and runoff ratio (5–25 percent)	Water shortage, intensive pine forest management effects on sedimentation

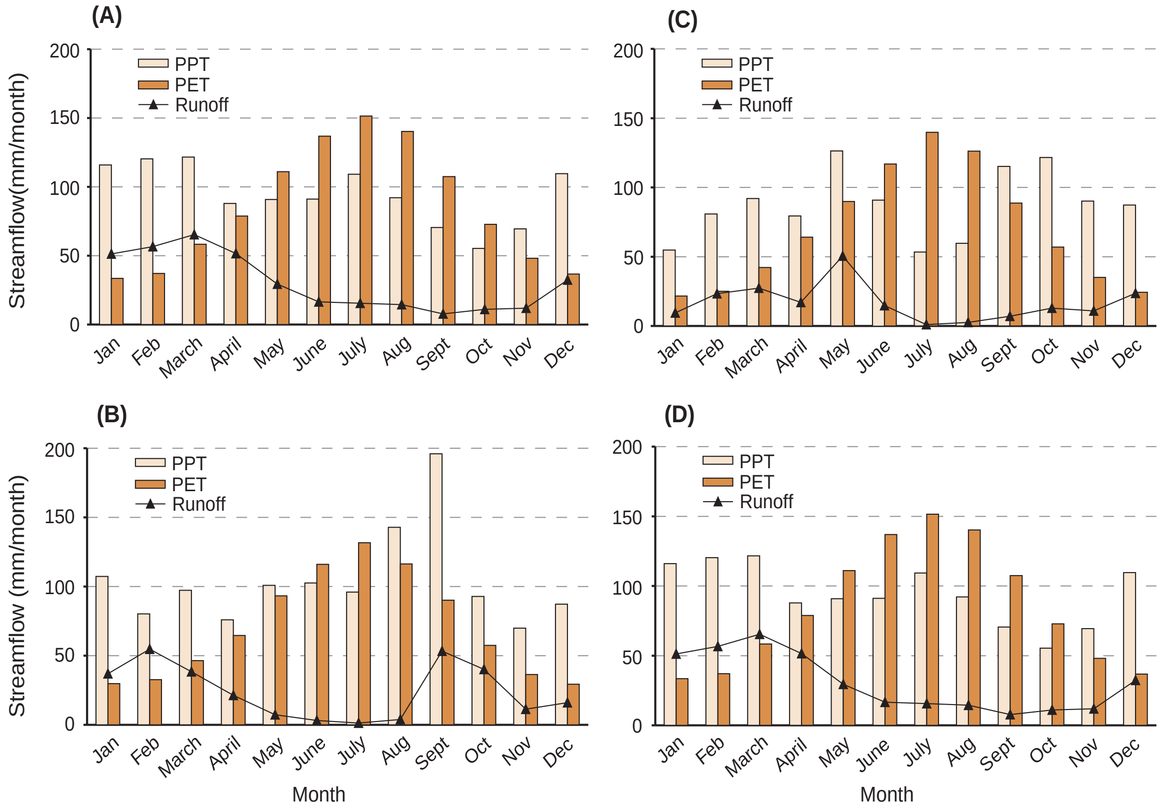

Figure 19.2—Seasonal distribution of streamflow from four representative forested watersheds across the Southern United States: (A) Arkansas mountains, (B) North Carolina Coastal Plain, (C) Texas shrubland, and (D) Georgia Piedmont (PPT= precipitation; PET= potential evapotranspiration).

WATER-QUANTITY IMPACTS

Water quantity or water yield refers not only to total waterflow volume on an annual basis but also to the timing of flow over shorter periods, as in the case of seasonal flow patterns or peak flow rates. For forested wetlands, the most important hydrologic variable is hydroperiod, the number of days when surface water is present per year. Hydroperiod, which is affected both by precipitation and by ET, controls the chemical and biological processes of wetlands. The fluctuation of the water table reflects the change in soil water storage that is roughly the difference between precipitation and ET and runoff. Because the hydrologic characteristics of uplands differ from those of wetlands, and because the forest management practices employed on uplands differ from those employed on wetlands, we discuss the effects of management on water quality for uplands and for wetlands separately.

Water-Quantity Impacts: Uplands

Harvesting—Without exception, experiments at Coweeta Hydrologic Laboratory (Coweeta) showed that forest harvesting in the Appalachian Mountains by clearcuts or partial cuts resulted in increased streamflow (Swank and others 1988). Clearcuts caused water yield to increase by 26 to 41 cm, or 28 to 65 percent of control during the first year after harvest (Douglass and Swank 1972, Swank and others 2001). Harvesting riparian vegetation resulted in a 12-percent increase in daily discharge on average and a nearly complete loss of diurnal variation in discharge (Dunford and Fletcher 1947). Douglass and Swank (1972) quantified the annual response of water yield to harvesting as a function of tree species, aspects and elevations of the watersheds, solar energy received, and basal area removed.

A paired watershed approach similar to the one developed at Coweeta was used to study effects of pine forest management on stormflow and soil loss in the Ouachita Mountains of Oklahoma and Arkansas in the 1980s (Scoles and others 1994). Small-watershed experiments in this region showed that (1) clearcutting increased annual stormflow volume by 100 mm, or 40 percent, and increased the number of stormflow events the first year after harvesting as a result of reduction of ET, however, subsoiling, a practice of breaking down large forest soil pores, reversed this response; (2) stormflow and peak flows under flood-producing conditions (high rainfall, wet soils) were not affected by forest removal; (3) peak flows tended to increase as harvest intensity increased. However, the differences were not large enough to be statistically important. Harvesting had the most influence on peak flows during late summer and early fall, when storms were generally small.

Blackburn and others (1985) reported hydrology and water-quality changes following clearcut timber harvesting in nine small gauged watersheds in eastern Texas. Three watersheds were clearcut, sheared, windrowed, and burned; three were clearcut and drum chopped; and three were left untreated as controls. During the first year following harvesting, stormflow averaged 147 mm for the intensively site-prepared watersheds, 84 mm for the chopped watersheds, and 25 mm for the controls. Stormflow in the second through fourth years following harvesting was less than half that measured in the first year.

With natural regeneration of plants, the increases in water yield declined logarithmically each year after the initial treatment (Swank and Helvey 1970, Swank and others 1988) (fig. 19.3). Depending on the history of forests, water yield recovered 30 to 50 years after harvesting in the Appalachian region (Swift and Swank 1981). At Coweeta, the water-yield increases on the treatment watersheds decreased at a rate of approximately 2 mm per 1-percent increase in regenerative forest cover (Hibbert 1966). Trimble and Weirich (1987) derived similar conclusions from correlations between historic hydrology and reforestation data (1919 to 1967) for large basins (2820 to 19 450 km^2) across the Piedmont regions. They found that reforestation reduced water yield in the Piedmont region, with the most significant reductions in water yield occurring in dry years. Over large heterogeneous basins, 10- to 28-percent increases in forest cover were correlated with reductions in annual water yield of 30 mm (4 percent) to 100 mm (21 percent). Small changes of water yield resulting from modest land use conversion, while not observable on first-order and second-order watersheds, were additive in the downstream direction and resulted in significant water-yield reductions for the encompassing larger basins.

Increases in water yield that result from forest harvesting are not distributed equally over the seasons. It was estimated that approximately 60 percent of the observed increase in water yield resulting from deciduous tree harvesting occurred in the late summer (July) to early winter (November) period. The majority of this increase occurred during the normally low-flow months of August, September, and October (Douglass and Swank 1972).

Conversion from deciduous to coniferous— At Coweeta, conversion of broadleaf deciduous forests to eastern pines reduced streamflow by 20 percent (Swank and Douglas 1974). The magnitude

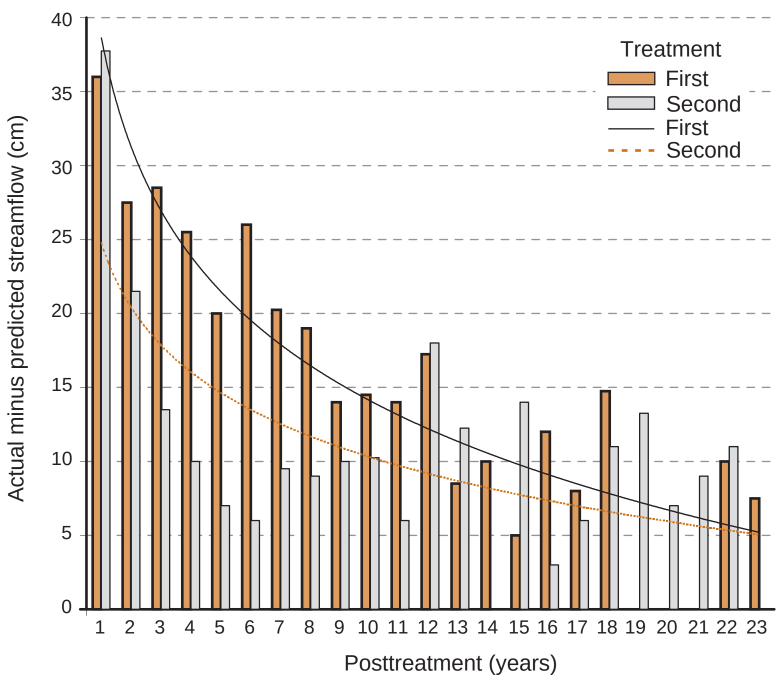

Figure 19.3—Annual streamflow responses to repeated harvesting of mixed hardwood forest (watershed 13 at Coweeta Hydrologic Laboratory) (adapted from Swank and others 1988).

of yield reduction is sensitive to precipitation, with greater reductions occurring during relatively wet years. Two factors are chiefly responsible for the observed reductions in streamflow. First, the interception capacity and subsequent evaporative losses of the white pine (*Pinus strobus* L.) stands can be twice those of the mixed hardwood communities they replaced (Swank and Vose 1994). Second, most of the precipitation occurring at Coweeta falls during low-volume, low-intensity rainfall events. Under these conditions, large amounts of precipitation can be intercepted by the evergreen tree canopy and can evaporate. Thus, conversion from mixed deciduous to coniferous species has a significant impact on water yield.

Although monthly water yield was always reduced after conversion from hardwoods to pines, the reduction was most pronounced during December to April when the differences of ecosystem ET (especially from interception losses) between the two forest types were the largest (Swank and others 1988).

Conversion from forest to grass—Conversion of 80 percent of a forested watershed with low tree density to Kentucky 31 fescue grass (*Festuca* L.) did not cause immediate increase in water yield at Coweeta (Hibbert 1966). A fertilized, highly vigorous, and productive grass system (> 1 m in height) could have used as much water as a forest. However, grass decline resulted in increased total annual water yield and baseflow, especially during the winter seasons. Flow frequency analysis suggested that dense grass or recolonizing forest might use more water than natural mature hardwoods during the summer growing season (Burt and Swank 1992). Stormflow frequency also increased as a result of forest conversion to grass.

Conversion from forest to mountain grazing—Although conversion from forest to grazing increased total water yield, the impacts of grazing on infiltration and ultimately peak flow rates were the most significant effects of the hydrologic changes. Before grazing, 102 mm of effective precipitation produced a maximum peak discharge of approximately 1.15 $m^3/s/km^2$. After grazing, it generated a peak discharge of approximately 28.7 $m^3/s/km^2$ (Johnson 1952). Hydrologic impacts of the mountain grazing were no longer detectable after 4 years of regrowth.

Conversion from forest to farmland—Researchers at Coweeta conducted experiments that combined forest removal practices with land use change. These experiments were designed to investigate the effects of typical land use practices in the Southern Appalachians on water yield and quality. For example, one 9-ha watershed was converted to an operational mountain farm by removing the forest and allowing farmers to utilize local agricultural practices including row cropping and unregulated grazing (Johnson and Kovner 1956). Following the treatment, annual water yield from the watershed increased 22 cm (Bosch and Hewlett 1982).

Water-Quantity Impacts: Wetlands

Wetland hydrology is extremely dynamic, involving complex interactions of surface water and ground water. Wetland hydrology research in the Southern United States is relatively new (Sun and others 2001). Most studies of the hydrologic impacts on southern forested wetlands have been conducted in the last two decades. Wetland hydrologic processes are ground-water driven. This review focuses on water-table responses to forest management practices. We review hydrologic impacts by wetland types because each wetland type has unique hydrologic features and responses.

Bottomlands—Harvesting of bottomland forests usually has little long-term effect on hydroperiod if BMPs are employed (Lockaby and others 1997a) or alternative harvesting methods, e.g., helicopters, are used (Perison 1997, Rapp and others 2001). The most common hydrologic change following harvesting of bottomlands is elevation of the water table (Aust and Lea 1992, Lockaby and others 1997b, Perison 1997, Wang 1996). This "watering-up" is attributed to these: (1) reduction of canopy interception and plant transpiration, (2) reduction of soil saturated hydraulic conductivity and increase of bulk density if harvesting sites are severely disturbed, and (3) increase of surface water storage and blocking of surface and subsurface drainage. However, one exceptional response has been reported: the water table in dark-colored organic soils dropped 20 to 40 cm during the postharvest period (Lockaby and others 1994). The causes of this unique response are not well understood. The water-table effects of forest harvesting on flood plains are most pronounced during the first two growing seasons (Lockaby and others 1997b, Wang 1996).

Depressional wetlands (cypress domes and Carolina bays)—Many depressional wetlands that are seasonally dry and isolated from streams or rivers are present in the Southeastern United States, and especially in Florida and on the

Atlantic Coastal Plain (Tiner and others 2002). Examples are cypress domes (ponds), Carolina bays, and pocosin wetlands of different sizes. They are important habitats for reptiles and amphibians (Russell and others 2002). Many of the small isolated wetlands are imbedded in intensively managed plantations. Isolated wetlands have similar hydrologic characteristics. They receive water and nutrient inputs mainly from precipitation, and lose water through the ET process. Physical and chemical interactions with surrounding uplands and deeper aquifers vary depending on local geology, but usually are minor as a consequence of the generally flat topography and the presence of thick fine-textured layers beneath the shallow surficial aquifer.

A hydrologic impact study at a 42-ha northcentral Florida site consisting of a mosaic of cypress ponds/pine flatwoods (Bliss and Comerford 2002, Crownover and others 1995, Sun and others 2000b) suggests that tree removals from isolated wetlands and surrounding uplands caused the water table to rise and runoff to increase. Harvesting of wetlands and surrounding uplands, and harvesting of wetlands alone, raised the level of the water table by as much as 100 cm. This effect on the water table was greatest during the first dry growing season. It was not significant subsequently; and in fact, the water table was somewhat lower than the control in the subsequent relatively wet years. The hydrologic regime at this site appeared to recover within 5 years. A retrospective 25-year study on a similar landscape in the Coastal Plain of Georgia suggests that the hydrologic impacts of harvesting pine plantations on cypress wetlands are inconclusive (Batzer and others 2000).

Wet flats and pocosins—Wet flats and pocosins occur on broad interstream divides on poorly drained soils. Wet flats occurring on higher elevations are better drained than pocosins, which develop thick organic layers. Most of the wet pine flats and pocosins have been intensively managed for timber production. Forest harvesting practices generally result in short-term water-table rise and an increase in runoff. A long-term watershed-scale (48 to 64 ha) study on a cypress-pine flatwoods landscape at the Bradford Forest in northern Florida showed that "maximum" disturbance caused a 15-cm or 150-percent increase in water yield and the "minimum" disturbance resulted in only an insignificant increase (3 cm) in water yield. Water table rose significantly for both treatments, especially during the drought months (Riekerk 1989b). In the sixth year after treatment, runoff from the maximum disturbance watershed was still significantly 65 percent higher than the predicted value from a regression equation. The ground-water tables in both disturbance sites remained higher than the control. Hydrologic changes (water table and runoff) were most pronounced in dry years. These findings were consistent with those of other experimental and modeling studies (Sun and others 1998a) in the region. Williams and Lipscomb (1981) found a water-table rise of 15 to 35 cm after partial cutting in a coastal pine forest on sandy soils. However, Rodriguez (1981) concluded that clearcutting a wet savanna watershed did not significantly alter the watershed hydrology.

Harvesting under wetland conditions, such as those encountered on wet pine flats, can alter soil hydrologic properties (e.g., hydraulic conductivity, macropores) by soil compaction, rutting, and puddling (Greacen and Sands 1980). The physical property changes affect subsurface flow and water-table depth. Soil compaction, rutting, and puddling become greater with increased soil wetness, clay content, and traffic (Green and others 1983). However, Aust and others (1995) found that skidding altered the hydrology of poorly and very poorly drained soils less than it altered that of moderately well-drained or somewhat poorly drained soils. This suggests that lateral subsurface water movement is an important factor in hydrologic impacts on wet pine flats, especially for fine-textured soils.

A field-scale study on wet pine flats in South Carolina has examined the effects of wet-weather harvesting, dry-weather harvesting, and bedding on hydrology and site productivity (Preston 1996, Xu and others 2002). Two site preparation levels (nonbed and bed) were randomly assigned to both dry-weather and wet-weather harvested plots, and an additional level of preparation (moleplow plus bed) was applied only in the wet-weather harvested plots. Dry-weather harvesting raised the water table 14 cm, and wet-weather harvesting raised it 21 cm. The response differences were largest (> 10 cm) during the growing seasons from May to October. Churning and deep rutting affected the water table significantly in wet-harvesting areas, but not in the dry-harvesting areas. Bedding lowered water tables initially in both areas, but the dry-harvesting site recovered within 2 years after replanting. Bulk density, macroporosity, and hydraulic conductivity were significantly affected by all levels of wet-harvesting disturbance. Dry-weather harvesting also altered those soil physical properties. The degree and extent of impacts were much greater for wet-weather harvesting than dry-weather harvesting (Xu and others 2002). Overall, the study suggests that change in water-table depth resulted from change in vegetation, and not from soil changes caused by harvest traffic. Similar changes in soil physical properties followed harvesting and regeneration in a wet pine-flat site in North Carolina, where soil macroporosity was reduced by half within a 200-cm profile (Blanton and others 1998).

Artificial drainage is used to increase operability and forest productivity on poorly drained soils in the Coastal Plain. Hydrologic effects of ditching vary depending on soil characteristics and the stage of vegetation development. Campbell and Hughes (1991) reported that free drainage in pine plantations on pocosins lowered the water table 30 to 60 cm during wet seasons. Standing water was minimized, but soil saturation was maintained and there was less fluctuation in the water table. Drainage did not change the basic hydrologic cycle or convert wetlands to uplands. A retrospective study in Virginia found that ditching significantly lowered the water table in pine plantations from 0 to 3 years old on wet flats during wet seasons when the water table was close to the soil surface (Andrews 1993). However, the ditching effect was dramatically reduced during the growing season at stand age 23. On Pomona sand in Florida, ditching affected water-table levels up to 45 m from the ditch (2 m deep and 3 m wide) for high and average water-table conditions (80 cm from surface) (Segal and others 1986). Hughes and others (1990) reported that flow volumes and seasonal hydrographs for a 16-year-old plantation, unditched natural timber, fully stocked pine plantations, a mixed plantation and naturally regenerated watershed, and a ditched natural stand did not differ. Simulation by the DRAINMOD hydrologic model suggested that ditch spacing had major effects on the composition of runoff from forest lands but caused limited change in total flow volume (Skaggs and others 1991).

Many field- to watershed-scale experiments and modeling studies have been conducted to determine how artificial drainage affects waterflow quantity and quality on the North Carolina Coastal Plain and to test various controlled drainage methods (Amatya and Skaggs 2001; Amatya and others 1996, 2000, 2002; Chescheir and others 2001; McCarthy and others 1991, 1992). Amatya and others (1997a) describe the 5-year hydrology of a 340-ha drained forested pocosin watershed in eastern North Carolina that had heterogeneous soils and underwent changes in vegetation in different fields during the study period. Total annual outflows from the watershed varied from 29 percent of the rainfall during the driest year, when most of the trees present were mature, to as much as 53 percent during a year of normal rainfall after about a third of the trees were harvested. Average annual ET, estimated as the difference between the gross rainfall and outflow, was 58 percent of the gross rainfall. Flow rates per unit area were consistently higher from a smaller harvested block (82 ha) of the watershed than from the whole watershed, partially as a consequence of routing effects in ditches and canals in the whole watershed. In an ongoing large watershed (2950 ha) study of cumulative impacts of management practices on the North Carolina coast, the runoff:rainfall (R:R) ratio varied from 15 to 32 percent as rainfall varied from 960 to 1410 mm. The forested watershed generally yielded no outflows in summer, when ET demands were high, except during periods when large tropical storms brought the water table to the surface. Annual ET, which was estimated as R:R, averaged 970 mm over 5 years. Heath (1975) gave a similar annual water-budget estimate for pocosins—1300 mm for precipitation, 910 mm for ET, 369 mm for runoff, and 13 mm for seepage. Accumulated data suggest that drainage of forested wet flats and forested pocosin wetlands has rather less impact on runoff than might be surmised. Peak flow rates for free-drained lands are higher than those for nondrained or control-drained areas (Amatya and others 1996).

WATER-QUALITY IMPACTS

In the United States, the best water comes from forested watersheds (Binkley 2001, Binkley and Brown 1993), even when forests are managed primarily for timber production (Binkley and others 1999). Many forest management practices, including timber harvesting, site preparation, prescribed burning, and the application of chemicals (insecticides and fertilizers), have the potential to degrade water quality. The impacts of forest management on water quality in the Southern United States have been summarized in review papers by Riekerk and others (1989a), Shepard (1994), Walbridge and Lockaby (1994), Lockaby and others (1997a), Lockaby and others (1999), and most recently by Fulton and West

(2002). Effects of forest management on the water quality in upland and wetland landscapes are discussed separately.

Water-Quality Impacts: Uplands

Riekerk and others (1989) synthesized findings of studies of silvicultural nonpoint-source pollution in uplands of the South. They noted that sediment production during silvicultural operations was low in the mountains and lower Coastal Plains, but high in the Piedmont and upper Coastal Plains. Nutrient export in the Piedmont and upper Coastal Plains was elevated, and rates of nutrient export were controlled by the degree of soil disturbance and the recovery rate of the vegetation. Nutrient exports in the lower Coastal Plains were not much affected by silvicultural operations.

Harvesting Impacts on Streamwater Chemistry—Change of streamwater chemistry is one important signature of ecosystem response to watershed disturbance. The impacts of forest harvesting on water chemistry and nutrient export were reported in a number of papers based on studies conducted at Coweeta (Douglass and Swank 1975, Johnson and Swank 1973, Swank and Swank 1981, Swank and Vose 1994, Swank and others 2001).

Johnson and Swank (1973) analyzed long-term water-chemistry responses for calcium (Ca), magnesium (Mg), potassium (K), and sodium (Na). They found that clearcutting treatments did not cause substantial losses of Ca, Mg, K, and Na over the duration of experimental record.

Converting hardwoods to white pine at Coweeta not only reduced water yield but also altered streamwater quality (Swank and Vose 1994). During the 20 years following initial treatment, streamwater solutes in the pine watersheds and the mixed hardwood control watersheds were generally similar. However, flow weighted mean nitrate-N (NO_3-N) concentrations increased slightly, 0.1 mg/L, while sulphate ion (SO_4^{2-}) concentrations increased nearly threefold. Johnson and Swank (1973) reported that reductions in the losses of Ca, Mg, K, and Na were 2.3, 1.7, 4.4, and 1.2 kg/ha/year, averaged over both watersheds; and Swank and Vose (1994) reported that the rates of reduction were unchanged 20 years later.

Swank and others (2001) contrasted the long-term water-chemistry records of a grazing and clearcut watershed with those of a control watershed. Increases in nutrient export occurred following harvesting with the largest, though relatively small, losses—1.3 kg/ha NO_3-N, 2.4 kg/ha K, 2.7 kg/ha Na, 3.2 kg/ha Ca, 1.4 kg/ha Mg, 0.4 kg/ha sulphur and 2.1 kg/ha chlorine—occurring during the third year following treatment. Export increases were frequently lower than background rates of atmospheric deposition. As in other studies, the nutrient losses returned to baseline levels within a few years after treatment. However, a second phase of increased NO_3-N losses started 14 years after treatment, and this effect had not been observed in other studies. It was hypothesized that mortality and shifts in species composition, nutrient releases from decomposition, elevated soil nitrogen (N) transformation, and reductions in soil carbon (C):N ratio could have contributed to the elevated NO_3-N export.

Harvesting Impacts on Sediment—Long-term effects of forest road construction and harvesting on watershed sediment loading were studied at Coweeta (Swank and others 2001). Prior to construction of a forest access road (~3 km) and cutting, sediment yield averaged 0.23 metric ton/ha/year while that from a control watershed averaged 0.1 metric ton/ha/year. Most of the logging was completed with a cable yarding system; tractor skidding was restricted to a 9-ha area where slopes were under 20 percent. Road construction and harvesting resulted in significant increases in water yield and soil loss. Over the period of monitoring, the rate of soil loss increased by a factor of 3.5. The majority of the measured sediment resulted from road erosion. The average sediment yield was about 340 metric tons/ha/year or 50 percent above the pretreatment level at the end of this 15-year experiment (fig. 19.4).

Harvesting trees without disturbing the soil generally did not increase sediment levels in runoff in the South, but mechanical site preparation with shearing and windrowing of debris generated significant sediment pollution (Ursic 1986). However, this was not the case in a study that compared the hydrologic responses to clearcutting with skidders and logging with a cable yard on a hilly upland site in the southern Coastal Plain in north Mississippi (Ursic 1991b). In the latter study, skidder harvesting increased sediment slightly (by 0.12 metric ton/ha/year), while cable harvesting increased sediment sixfold (3.3 metric tons/ha/year) over the first 5 years. Subsurface flow was critical in elevated channel erosion and deposition in forested landscapes on unconsolidated formations in the Coastal Plain

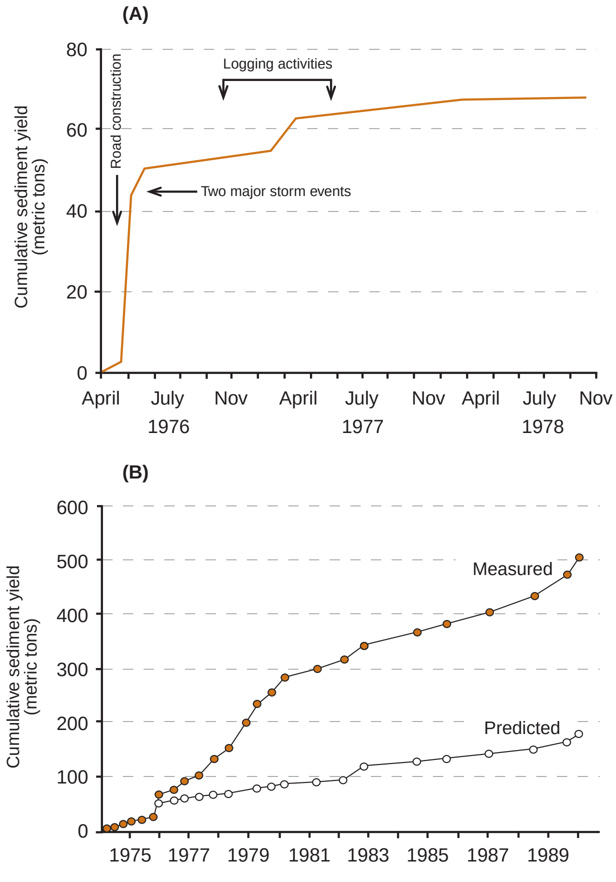

Figure 19.4—Cumulative sediment yield measured at one Appalachian watershed at Coweeta Hydrologic Laboratory (A) in one of the first-order streams below a logging road during the first 32 months after treatment, and (B) in the ponding basin of the second-order stream (Swank and others 2001).

(Ursic 1991b). Another study, which employed 16 small mature loblolly pine (*P. taeda* L.) plantation watersheds on the upper Coastal Plain in Tennessee, also indicated the importance of channel erosion on sediment loading (Ursic 1991a). This study suggested that harvesting of trees growing on previously degraded lands had limited effects (< 0.093-metric ton/ha/year increase) on sediment loading, usually within 4 years.

Mechanical disturbance of the forest floor during harvesting and site preparation is generally supposed to reduce soil productivity and increase soil erosion. One might expect that soil structure and site hydrology would be affected by increased disturbance intensity. However, a recent study at a Piedmont site in Alabama found that surface runoff was lower at a clearcut and bedded site than it was at a site that was clearcut but not bedded (Grace and Carter 2001). Bedding produced additional surface roughness and storage. Although this was a small-scale study, it demonstrated the uncertainty of hydrologic responses to current silvicultural practices.

Scoles and others (1994) described a 15-year study of hydrology and water quality on gauged watersheds in the Ouachita Mountains of Arkansas and Oklahoma. They noted threefold-to-twentyfold increases in soil erosion following selection harvesting and clearcutting. However, the amount was still low, about one-thirtieth of the loss from cropland, and recovery to baseline conditions occurred in the first 5 years of the 35-year rotation. Most erosion occurred during a few large storms each year, and 90 percent of annual stream sediment came from roads. Projected sediment delivery to streams in the Ouachitas as a result of harvesting, site preparation, and erosion from roads was about 0.16 metric ton/ha/year.

Lawson (1985) reported that sediment losses in undisturbed pine forests in the Ouachita Mountains, Ozark Plateaus, and Boston Mountains averaged < 0.02 metric ton/ha/year. Maximum sediment losses of 0.13 metric ton/ha/year were observed during the first year following clearcut timber harvesting. Recovery to preharvest levels of sediment production occurred within 3 years.

Harvesting Impacts on Stream Temperatures— Forest harvesting along streams usually results in increased stream temperatures (Swift 1973, 1982; Wooldridge and Stern 1979). Swift and Messer (1971) monitored stream temperatures in treatment and control watersheds. On the watersheds that were harvested completely, summer stream temperatures increased from an average of 19 °C to more than 23 °C. The most intensive treatments raised temperatures by more than 7 °C. Water temperatures in streams with uncut streamside or riparian vegetation did not increase. Also, temperatures in the impacted streams returned to pretreatment levels when regeneration of riparian vegetation provided shading. The temperature increases significantly altered streamwater quality, in that water temperatures were raised above levels suitable for the native trout populations (Swift and Messer 1971).

Swift (1973) investigated the effectiveness of preserving a 12-m buffer of streamside vegetation in mitigating potential streamwater temperature impacts of forest harvesting on a small, mountain watershed. The stream flowed through alternating cut and uncut riparian zones. Water temperature rapidly increased by up to 6 °C as the stream flowed through a 900-m cut area. It then decreased by approximately 1 °C as the stream flowed through an 800-m uncut riparian area, and increased again as it passed through a 200-m cut area. The stream's temperature eventually stabilized to normal (12.8 °C) when the stream passed through two forested sections of a total of 2100 m. The alternating network of cut and uncut riparian areas limited maximum water temperatures to < 20 °C, a temperature above which trout habitat is impaired.

Swift (1982) reported long-term impacts of cable logging on streamwater temperatures. In the first 2 years following cable logging, average summer streamwater (38 percent shaded) temperatures increased 3.3 °C. Regeneration of streamside trees by stump and root sprouting increased streamside leaf biomass to 78 percent of the pretreatment condition within 3 years following treatment. Subsequent temperature increases averaged 1.2 °C. Minimum temperatures were elevated only in the first year of treatment, whereas daily temperature range (maximum – minimum) was elevated during the 5 years of the study. Swift (1982) predicted that the increase in streamwater temperature would decrease at a rate of 0.3 °C per year, and that streamwater temperature would ultimately return to pretreatment levels.

Harvesting Impacts on Biota—Williams and others (2002) investigated the effects of timber harvesting on physical stream features and regional fish and macroinvertebrate assemblages during a 3-year period in six hydrologically variable streams (basin area 1517 to 3428 ha) in the Ouachita Mountains, AR. Most of the variations in fish assemblages were explained by drainage basin differences, and both basin and year-of-sampling influenced macroinvertebrate assemblages. Williams and others (2002) argue that the lack of logging effects on biota may be due to the scale of the study, timing of the sampling, and high levels of natural variability in the streams.

Forest conversions—The conversion of cove hardwood to mountain grazing at Coweeta resulted in decreased water quality. Peak concentrations of sediment during stormflow events were up to three times higher than expected as the clay fraction of the suspended load significantly increased beyond pretreatment conditions (Johnson 1952).

Converting hardwoods to blue fescue grass at Coweeta (see water-quantity sections also) required spot applications of herbicides to suppress competition. Erosion of the stream channel margin occurred after the first herbicide application. When herbicides were applied a second time, a 10-foot buffer of grass along the channel was left unsprayed. Atrazine was detected in the streamwater following storm events for 4 months after the herbicide treatments. The largest concentration of atrazine, ranging from 24 to 34 parts per billion, occurred immediately after application. Following the second treatment, a sustained, background concentration (4 parts per billion) of atrazine existed in the streamwater for 3 months. No detectable atrazine was found 3 months after the last application (Douglass and others 1969).

Forest road construction—Much of the road research at Coweeta was conducted in conjunction with forest harvesting experiments and was designed to identify methods that would reduce sedimentation from access roads for timber harvesting operations. A comprehensive summary of these experiments and their results is given by Swift (1988) and Swift and Burns (1999).

Hursh (1935, 1939, 1942) reported that soil loss from forest roads could be reduced significantly by mulching or vegetating the adjacent cut-and-fill slopes. Several best construction methods employing bioengineering techniques to stabilize slopes were identified. Use of these methods significantly reduced sedimentation following road construction.

As a part of research on exploitive logging at Coweeta, loggers were allowed to construct roads in a typical fashion. This included the construction of skid trails directly upslope, from roads, and adjacent to and within streams (Swift 1985). These practices resulted in the loss of 408 m^3 of soil/km of road constructed (Lieberman and Hoover 1948b). Sediment delivery to streams was high, and turbidity peaked at 5700 parts per million (Lieberman and Hoover 1948a), significantly reducing downstream aquatic fauna (Tebo 1955). Road erosion became so severe that the roads had to be closed and stabilized. It was concluded that the sedimentation observed in the stream resulted almost exclusively from road erosion and not from other forest harvesting activities (Swift 1988).

Swift (1988) found that soil loss potentials were highest immediately after road construction at Coweeta. The pulses of soil loss after road construction were triggered by intense rainfall events. Soil losses from bare roadbeds were eight times those from graveled roadways. The erosion rates declined in the ensuing 6 months; however, losses from the bare soil were still six times greater than those from the gravel treatments. The largest losses were from roads that received frequent traffic. When logging trucks were present, losses of soil from roads surfaced with thin layers of rock were similar to those from bare-soil roads. Surfacing with large (20 cm) crushed stones afforded the most protection against erosion; however, this stone was deemed too coarse for many vehicular operations. Medium (15 cm) crushed rock provided roadbed protection similar to that obtained with the 20-cm stone but at a significantly reduced cost. Rates of erosion from roadbeds of fine (5 cm) crushed rock were similar to those for bare-soil roads. Seeding of the roadbeds and cut-and-fill slopes of lightly traveled roads reduced erosion rates to 50 percent of those for bare-soil conditions (Swift 1984a).

Swift (1984b) monitored rates of erosion from roadbeds, cut slopes, and fill slopes along a series of road treatments (Swift 1984a). Although soil losses from all surfaces were high during heavy rains, rates of erosion from bare cut-and-fill slopes that made up half of the road prism accounted for 70 to 80 percent of the total soil losses from the sites. Graveling of the roadbed reduced soil losses to < 20 percent of those for the bare-soil condition. A combined treatment that consisted of vegetating the cut-and-fill slopes and graveling the roadbeds reduced total erosion rates to < 10 percent of those for the pretreatment condition. Despite these improvements, the net loss of soil from the entire roadway was 20 times greater than that estimated for undisturbed forest (Swift 1984b).

Fertilization—Binkley and others (1999) and National Council for Air and Stream Improvement (1999d) reviewed world literature about response of streamwater chemistry to forest fertilization in upland and wetland forest ecosystems. They found that forest fertilization usually results in elevated N and phosphorus (P) concentrations, especially if pellets are deposited directly into streams and ditches. However, maximum concentrations of N were rarely above drinking water-quality standards, and elevated concentrations were short-lived (weeks to months). Elevated concentrations were typically one-tenth of those observed in agriculture. Highest N concentrations were observed with repeated applications, and when ammonium nitrate rather than urea was used. No evidence of effects on aquatic organisms was found, but few studies included such an assessment.

In the South, it is common to fertilize with N and P to increase tree growth. In a study on the North Carolina Coastal Plain, fertilization resulted in elevated concentrations of ammonium as much as 3.8 mg/L, total N as much as 9.3 mg/L, total phosphate (PO_4-P) as much as 0.18 mg/L, orthophosphate as much as 0.1 mg/L, and urea as much as 1.2 mg/L measured at the field (27 ha) edges (Campbell 1989). After 3 weeks of treatment, concentrations of all ions returned to pretreatment levels. Concentration of NO_3-N ranged from 0 to 1.2 mg/L during the 60-day monitoring period. Segal and others (1986) and Riekerk (1989b) reported similar findings for studies in the Coastal Plain. Information about effects of fertilization on water quality in other physiographic regions can be found in National Council for Air and Stream Improvement (1999d). Fertilization of forest lands has rarely caused NO_3-N concentration in streams to exceed the U.S. Environmental Protection Agency (EPA) drinking water standard of 10 mg/L, especially when care has been taken in applying the fertilizer.

Prescribed Fires—In the Southern United States, about 1 million ha of forest land is subjected to prescribed burning annually to reduce fuel loads, enhance stand health, and release preferred forest species from competition (Clinton and others 2000, Richter and others 1982, Vose and Swank 1993). The negative impacts of this practice on forest communities include reduction of total ecosystem N as a result of volatilization and leaching (Knoepp and Swank 1993) and a potential increase of sediment loading (Knoepp and Swank 1993, Vose and others 1999). The magnitude of effects of this practice varies greatly and depends on fuels, soil properties, topography, climate, weather, and fire frequency and intensity (Richter and others 1982).

Ursic (1969) described effects of prescribed burning on hydrology and water quality in two abandoned fields in Mississippi. Stormflow during the first year increased 48 percent in one catchment, and stormflow increased in the second and third years also. Treatment of the second catchment, which had a fragipan, did not change the volume of stormflow but significantly increased peak discharges and overland flow. During the first year, sediment production increased from 0.11 to 1.9 metric tons/ha in the first catchment and 7.5 metric tons/ha in the other. Sediment production dropped to < 0.56 metric ton/ha the third year.

Douglas and Van Lear (1983) reported responses of nutrient and sediment export to prescribed burning at a Piedmont site in the Clemson Experimental Forest, SC. Four loblolly pine watersheds were burned twice at 18-month intervals. The first burn took place in March and the second in September. The prescribed burns did not change water quality of the streams.

Clinton and others (2000) summarized the results of four experiments that examined stream nitrate (NO_3-N) responses to forest fires in the Nantahala National Forest in western North Carolina. A fell-and-burn fire (Jacob's Branch) and two stand-replacement fires (Wine Spring Creek and Hickory Branch) were implemented to improve degraded xeric oak (*Quercus* spp.)-pine forests. The fourth (Joyce Kilmer) fire was an arson-related wildfire, burning the understory in an old-growth mesic and xeric forest. The Jacob's Branch and Joyce Kilmer fires occurred in the fall, and the fires on Wine Spring Creek and Hickory Branch were spring burns. Stream nitrate was elevated by 0.03 mg/L for 8 months following the burn on Jacob's Branch and by 0.06 mg/L for 6 weeks following the Joyce Kilmer fire. There was no stream NO_3-N response at the two spring burn sites. Clinton and others surmised that N released during the spring burns was immobilized by vegetation uptake, but that N released during the fall burns was not.

Neary and Currier (1982) monitored stream chemistry [NO_3-N, ammonium nitrogen (NH_4-N), PO_4-P, Na, K, Ca, and Mg] and total suspended solids for five streams burned by wildfire in the Blue Ridge Mountains of South Carolina. Increases in streamwater nitrate, NO_3-N, were attributed to fertilizer applications. Elevated concentrations of NO_3-N and PO_4-P in streamwater occurred mostly during stormflow events, and mean concentrations were not significantly higher than those observed on undisturbed watersheds. Concentrations of anions Na, K, Ca, and Mg ranged from 12 to 82 percent above background levels during the monitoring period.

Forest fires can burn significant amounts of the understory canopy, litter, and duff layers of forests, leaving forest soils unprotected against raindrop impact. The combustion of forest litter and plants in high-intensity forest fires can create and concentrate petroleum-based compounds that induce water repellency in forest soils. This reduces infiltration and increases runoff and soil erosion, especially in the forests of the Western United States (Tiedemann and others 1979, Wolgemuth 2001, Wright and Bailey 1982). However, Wolgemuth (2001) found that forests treated with prescribed fires had erosion rates lower than unburned forests had during subsequent fire events on chaparral watersheds in southern California.

The literature suggests that fire generally has less effect on sediment loading in the Southern United States than it has in the Western United States (Goebell and others 1967, Marion and Ursic 1992, Shahlee and others 1991, Swift and others 1993, Van Lear and Danielovich 1988, Van Lear and Waldrop 1986). Increased soil erosion following fires is frequently associated with forest floor disturbances caused by mechanical site preparation during fire controlling activities. Similarly, operationally disturbed sites and especially skid trails have been found to be more susceptible to erosion following fires (Ursic 1970, Van Lear and others 1985). However, it must be noted that most fire research in the Southern Appalachians has involved fires of low-to-moderate intensity (Swift and others 1993, Van Lear and Waldrop 1989).

Pesticides—Pesticides have been increasingly used in the South to control insects and unwanted vegetation. These organic substances have the potential to pollute water by aerial drift, decomposition, leaching and adsorption, and transport by subsurface flows. Substantial effort has been made to discover the fate of applied forestry pesticides (Michael and Neary 1990; Neary 1983; Neary and Michael 1996; Neary and others 1985, 1993; U.S. Department of Agriculture, Forest Service 1994). The literature suggests that the risk of long-term contamination from pesticide application is low when care is taken. Residues are not persistent and do not bioaccumulate. When herbicides are not applied directly to streams or when buffer strips are used, peak residue concentrations are generally low (< 500 parts per billion), and residue levels in surface runoff are < 36 parts per billion for ground application and < 130 parts per billion for aerial applications (Riekerk and others 1989). Most herbicides used in modern silviculture are of low toxicity to aquatic and terrestrial organisms, and thus pose little hazard to wildlife.

Insect outbreaks—Insect defoliation was responsible for the increased stream nitrate concentration in several watersheds at Coweeta (Swank and others 1981). For example, a sudden rise of nitrate concentrations from 0.5 to 0.75 mg/L in one watershed was caused by a widespread outbreak of the locust stem borer (*Ecdytolopha insiticiana* Zeller) in black locust (*Robinia pseudoacacia* L.) (Swank and Crossley 1988).

Cumulative effects—Multiple forest operations that may or may not be separated in space and time can have cumulative effects on water quality. Bolstad and Swank (1997) analyzed the cumulative effects of land use practices separated in both space and time along the Coweeta Creek in the Appalachians. They found that water quality degraded from the creek's headwaters to the watershed mouth. The water quality at the confluence of two forested subwatersheds was very good. However, averaged stormflow turbidity, conductance, NO_3-N, NH_4-N, temperature, total coliform, fecal coliform, and streptococci counts increased by factors ranging from three (turbidity) to eight (total coliform) as the stream flowed through the residential and agricultural lands to its mouth. The increases in these water-quality parameters were strongly and positively correlated with numerous measures of human impacts including percent nonforest land, paved road density, paved road length, building frequency, and building density.

Water-Quality Impacts: Wetlands

Studies of the effects of forest management on wetland forest water quality and geochemical balances in the Southern United States are summarized in Shepard (1994), Walbridge and Lockaby (1994), Lockaby and others (1997a), and Lockaby and others (1999). Major findings from these papers and other recent publications are discussed below by type of management practices.

Drainage—Drainage usually does not alter water quality greatly. Williams and Askew (1988) reported a small increase in suspended sediments from newly built ditches at a pocosin site in South Carolina and concluded that establishment of pine plantations did not have to harm water quality. Lebo and Herrmann (1998) found that harvesting of loblolly pine plantations increased outflow and slightly increased nutrient concentrations. For a 3-year period after harvesting, increases in annual outflow, N export, and P export were 111 to 164 mm, 2.1 to 2.2 kg N/ha/year, and 0.12 to 0.36 kg P/ha/year, respectively, compared with baseline levels. The baseline values for total N and P loading ranged from 2.7 to 3.4 kg N/ha/year and 0.09 to 0.29 kg P/ha/year, respectively. Outflow and nutrient concentrations returned to baseline levels within 2 to 3 years. These relatively small temporary increases in annual nutrient exports associated with harvesting and site preparation can be considered in the context of 30- to 50-year rotations for loblolly pine in coastal North Carolina. On that basis, they represent an increase of 4 to 7 percent above baseline levels.

In a large drained watershed (2950 ha) with mixed land uses in eastern North Carolina, ranges of measured total N concentrations at the field edges were 0.5 to 15 mg/L for the organic soils and 0.3 to 5.0 mg/L for the mineral soils. The annual total N loading varied from about 4.8 kg/ha to as much as 26.6 kg/ha, with an average of 14 kg/ha. Most of the total N was in organic form. It appears that the variation in nutrient loading attributable only to soil can be as great as that caused by forest harvesting.

Harvesting—In wetands, forest harvesting followed by site preparation activities has the potential to disturb the surface soils, alter surface and subsurface flow paths, increase water yield, and accelerate nutrient cycling rates, and thus affect onsite and offsite water quality. Riekerk (1985) found that clearcutting a pine-cypress flatwoods resulted in significant increases in pH, suspended sediment, and total N, K, and Ca during the first year after harvest. Fisher (1981) described the effects of clearcutting and site preparation on the hydrology and water quality of a pine flatwoods site in western Florida. Runoff volume was increased during the first year, but by the second year most water-quality parameters had returned to near background levels. The impact of silvicultural operations was less on the level, sandy site than in areas having more relief and shallow soils (Fisher 1981).

There are over 12 million ha of bottomland hardwoods forest in the Southern United States. A series of harvesting experiments have been conducted to examine their ecological responses to timber harvesting (Aust and others 1991, 1997; Lockby and others 1994, 1997b; Messina and others 1997; Wang 1996). These experiments indicate that onsite effects of harvesting flood plain forests are minor because the site disturbance is not great, because water movement is slow and because harvesting causes an increase in surface roughness for sediment and nutrient retention (Aust and others 1991, Rapp and others 2001). The treatment effects in these studies were often overwritten by seasonal flooding events (Perison 1997).

Prescribed fires—Controlled burning in the Atlantic and Gulf Coastal Plain reduces the risk of wildfire, controls certain tree pathogens, improves wildlife habitat, and restores desired ecosystems. For these reasons, the use of controlled burning is very common. However, few data are available on the effects of this management tool on the hydrology and water quality of wetlands. One exception is a 3-year paired watershed study conducted at the Santee Experimental Forest in eastern South Carolina (Richter and others 1982, 1983). The treatment and control watersheds were about 160 and 200 ha in area, respectively, and had first-order perennial streams. Dominant soils were Aquults (high water table), and the watersheds were covered by natural stands of loblolly pine. Burns prescribed for twenty 7.1-ha compartments were administered during winters and summers. It was concluded that periodic prescribed fires in these southeastern Coastal Plain pine forests are not likely to have great impacts on onsite or offsite water quality (Richter and others 1982).

WATER QUALITY REGULATIONS, AND DESIGN AND EFFECTIVENESS OF FORESTRY BMPS

Federal and State Water-Quality Programs

There are many Federal and State programs designed to protect water quality. The first Federal water-quality regulation applicable to forestry was included in the 1972 amendments to the Federal Water Pollution Control Act (commonly known as the Clean Water Act). This statute introduced two new concepts in Federal water-quality protection. First, its Section 208 required States to prepare area-wide (watershed or regional) waste treatment management plans; and second, it separated water pollution into two categories: pollution for point and nonpoint sources (Ice and others 1997).

Initially, the Clean Water Act was interpreted as requiring States to prepare water-quality management plans only for waters the States identified as impaired. However, the successful lawsuit NRCD vs. Train (1975) resulted in EPA requiring area-wide programs statewide, not just for areas with water pollution problems. In the regulations developed following NRCD vs. Train, EPA specified that States may develop nonpoint-source control programs that prescribe "Best Management Practices" (Rey 1977). The EPA defined BMPs as

> . . . a practice or combination of practices, that are determined by a state, or designated area-wide planning agency, after problem assessment, examination of alternative practices, and appropriate public participation, to be the most effective, practicable (including technological, economic, and institutional considerations) means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals . . . (Ice and others 1997).

Oregon was the only State that had a silvicultural nonpoint-source control program prior to 1972 (National Council for Air and Stream Improvement 1999c). Florida and South Carolina were among the first States to develop BMPs; South Carolina published its first water-quality guidelines in 1976, and Florida began developing its BMPs in 1976 (Greis 1979). The 1987 amendments to the Clean Water Act created Section 319, which required States to develop and implement programs to control nonpoint-source pollution. This amendment stimulated States to accelerate their efforts to develop silvicultural BMPs, and many State programs began shortly after 1987.

Total Maximum Daily Loads

Section 303(d) of the Clean Water Act requires that States list waters that do not meet water-quality standards or that do not meet their designated uses because of a particular pollutant such as sediment, pathogens, nutrients, or temperature. States must then develop a total maximum daily load (TMDL) for each pollutant causing impairment, specifically for each water body. A TMDL is the maximum amount of a pollutant that a water body can receive and still meet water-quality standards and an allocation of that amount among the pollutant's sources. It can be expressed as follows:

$$\text{TMDL} = \Sigma \text{ waste load allocation} + \Sigma \text{ load allocation} + \text{margin of safety}$$

Waste load allocations are point sources of the pollutant, and load allocations are nonpoint sources, including natural background levels of pollutants such as sediment, nutrients, and temperature. Although TMDLs were included in the 1972 Clean Water Act, it was not until the late 1980s that lawsuits forced the EPA and States to begin implementing the program. Hundreds and sometimes thousands of water bodies were listed in States. The total number listed in 2001 was over 40,000 (National Research Council 2001). States are being required by court orders or by EPA guidance to develop these TMDLs within 8 to 13 years, but the amount of effort required dwarfs the budgets of State water-quality agencies. Meeting these deadlines has become ". . . the most pressing and significant regulatory water quality challenge for the states since passage of the Clean Water Act . . ." (National Research Council 2001).

Routine forest management has not yet been affected by TMDL regulations. The EPA is currently operating under regulations promulgated in 1992, but revision of those regulations is pending. Current EPA policy recommends that States with approved forestry BMP programs grant exemptions to enforceable water-quality standards to forestry operators who implement BMPs (Anon. 2000).

Federal Wetlands Regulations

Section 404 of the Clean Water Act regulates the discharge of dredged and fill material into "waters of the United States," which include wetlands adjacent to navigable waters, interstate wetlands, and isolated intrastate wetlands that could affect interstate or foreign commerce (Guzy and Anderson 2001). The U.S. Army Corps of Engineers (Corps) administers the Section 404 program, and the EPA is responsible for policymaking and oversight of the Corps' management of the program. Those who conduct activities that will result in the deposition of more than a *de minimis* (threshold) amount of dredge and fill material in wetlands must apply for a permit from the Corps. Normal farming, silvicultural and ranching activities such as plowing, seeding, cultivating, minor drainage, and harvesting for the production of food, fiber, and forest products are exempted from this regulation when they are part of established operations (33 CFR 323.4). Other activities exempt from the permit program are minor drainage (that does not convert the site to upland), maintenance of existing ditches, and building of forest roads (subject to 15 BMPs).

In 1995, the EPA and Corps released to the field a memorandum about "Application of Best Management Practices to Mechanical Silvicultural Site Preparation Activities for the Establishment of Pine Plantations in the Southeast." This memorandum specified that a permit would be required for mechanical site preparation to establish pine on wetlands supporting riverine bottomland hardwoods, white-cedar [*Chamaecyparis thyoides* (L.) B.S.P.] swamps, Carolina bays, low pocosins, and wet marl forests. It also specified that other wetlands would continue to be exempt from permitting as long as six new BMPs were employed (Anon. 1996). A more comprehensive description of wetlands forestry regulations is found in Gaddis and Cubbage (1998). A comprehensive treatment of the entire Federal wetlands regulatory program, and a discussion of litigation history, is provided by Want (1998).

Design of BMPs

Generally, BMPs designed to protect water quality fall into four categories: (1) those related to riparian areas, (2) those related to disturbed areas such as roads and landings, (3) those related to wetlands, and (4) those related to harvesting practices. In most Southeastern States, BMPs are voluntary. In Virginia, however, loggers or landowners must notify the State Forester at the start of an activity, and the State Forester can mandate corrective actions when a threat to water quality is identified.

High BMP compliance rates are in the interest of the forest industry because the EPA currently accepts BMPs as appropriate and sufficient mitigation to meet the requirements of the TMDL program. If rates of BMP compliance are not high, water-quality regulations may be imposed. Another reason for employing BMPs is the threat and reality of citizen lawsuits. If runoff from a forestry operation causes water-quality problems for a downstream landowner, the downstream landowner can sue for damages. In such cases, the operator's compliance with BMPs is always a major issue.

Most nonpoint-source pollution caused by silvicultural activities starts with exposure of bare soil and soil disturbance. When raindrops strike the ground, they detach and disperse soil particles. Raindrops also compact surface soil, and this compaction promotes overland flow that mobilizes sediment and transports it to the stream system. If fertilizers or pesticides have been applied to the ground surface recently, then overland flow also transports these chemicals to streams. Therefore, most BMPs are designed to minimize the amount and duration of bare soil and the hydraulic connectivity of runoff from bare-soil areas to streams.

The development of silvicultural BMPs over the last 20 years has been based on forestry research and basic principles of stream ecology. Logger and forester experience, common sense, and political negotiation have also factored into the development of BMPs. Because soils, topography, climate, and political environments vary from State to State, the States have issued different BMP prescriptions.

Riparian Areas

The term riparian area refers to a channel-adjacent terrestrial area in which the presence of the stream and high water tables are primary influences on vegetation and soil development. In turn, the vegetation affects channel conditions by altering the microclimate and providing organic inputs to the stream system. Most BMP guidelines call for maintaining natural conditions in a portion of the riparian areas adjacent to channel systems to protect streamwater quality from potential effects of upland management practices. These riparian protection areas are described as riparian zones, riparian management zones, buffers, filter strips, and streamside management zones (SMZs) in the BMP guidelines defined by different States (Stringer and Thompson 2000). Riparian buffers protect water quality by (a) stabilization of stream banks, (b) filtration of overland flow and adsorption of chemicals transported in overland flow, (c) denitrification of shallow ground water, and (d) maintenance of shade and organic debris recruitment for channels.

Stringer and Thompson have published a review of State guidelines on riparian zones. Their findings are summarized: Most States in the Southeast now specify (a) minimum allowable distance between water bodies and the nearest severe disturbance, e.g., roads or landings; and (b) the allowable harvest within the riparian area (Stringer and Thompson 2000). In most States, the allowable distance between severe disturbance and perennial water bodies (perennial streams and lakes) increases as upland slope increases. This is because the potential for surface runoff impacts is greater with higher upland grades (Trimble and Sartz 1957). Most States allow 25 to 50 percent removal of the overstory within the perennial riparian area (Stringer and Thompson 2000). Generally, BMPs for intermittent streams are less restrictive as they are considered to have less potential nonpoint-source pollution impact than perennial streams. In the Southeast, about one-half of the States set a specific minimum distance to the nearest disturbance, and about one-half relate the distance to upland grade; however, these allowable distances are generally narrower than those for perennial water bodies. Allowable harvest generally ranges from 75 to 100 percent canopy removal. The effects of headwater area forestry operations on water quality are poorly understood. Most State BMPs do not explicitly recommend SMZs for ephemeral streams. Most States in the northern part of the Southeast also have guidelines for forestry practices in areas near coldwater streams that support trout. Generally, BMPs for areas adjacent to coldwater streams are more restrictive than those for areas adjacent to perennial streams in terms of buffer width and overstory removal within buffers.

Roads

Because logging roads create permanent areas of bare and compacted soil, they are the principal contributor of sediment from forestry activities (Swank and others 2001). Road and landing position in the landscape, the soil type and geology present, and method of retirement ultimately determine the magnitude of sediment flux to the stream (Ketcheson and others 1999, Swift 1988).

Impacts of road runoff and sediment can be reduced greatly by reducing or eliminating the hydraulic connectivity of roads to streams. This is done by routing water off roads at regular intervals onto hillslope locations where flow can be dispersed and reinfiltrated. Water bars, broad-based dips, and cross-drains are typical methods by which road runoff is routed from roads onto hillslopes (Cook and Hewlett 1979, Swift 1988). Depending on slopes and native soils, surfacing roads with gravel or rock can also reduce surface erosion. A review of specific State guidelines for BMPs related to roads and skid trails can be found in Stringer and Thompson (2001).

The use of filter strips along roads also mitigates the propagation of road sediment through drainage networks and into streams. While filter strips normally include natural vegetation, their performance may be augmented by using trees and woody slash material to form brush barriers. Use of such materials reduced distance of sediment movement by approximately 40 to 50 percent (Swift 1986). Natural forest litter was also instrumental in inhibiting transport at burned sites. Grass-covered sites on which runoff was diverted through forest litter and brush barriers provided the most resistance to flow. Swift (1986) observed the distances traveled by sediment from forest roads. He recommends that the width of forest road buffers be designed on the basis of land slope and the use or nonuse of brush barriers.

Grace (2000) studied the effectiveness of three common treatments for controlling erosion from cut-and-fill slopes of roads in the Talladega National Forest, AL. The three treatments were (1) native species vegetation mix, (2) nonnative species vegetation mix, and (3) nonnative species vegetation mix anchored with an erosion-control mat. Surface runoff and sediment yield were significantly reduced on both the cut-and-fill slopes. The three control measures reduced sediment production by 60 to 90 percent. The erosion-control mat treatment was the most effective of the three.

Clinton and Vose (2002) evaluated the effectiveness of road paving in reducing the delivery and transport of sediment from mountain forest roads. Delivery and transport of sediment were measured for four surface types: (1) 2-year-old pavement, (2) improved gravel, (3) improved gravel with sediment control, and (4) unimproved gravel. The paved road system generated the least sediment while the unimproved road generated the most. The distance of sediment transport away from the roadbed was greater for the paved road surface and decreased progressively for improved gravel, improved gravel with sediment control, and unimproved surfaces.

Appelboom and others (2002) evaluated the effectiveness of four road management practices (continuous roadside berms, noncontinuous roadside berms, a graveled road surface, and a nongraveled road surface) in reducing sediment loading to ditches within a drained forested watershed on the lower Coastal Plain of North Carolina. They found the presence of access roads under the four practices had little impact on sediment loading at the watershed outlet when comparing the sedimentation of drainage from similar watersheds without access roads.

Wetlands

The Clean Water Act allows minor drainage for forestry without a permit. However, such drainage cannot connect wetlands to uplands. Thinning or harvesting of overstory trees is allowed in most wetlands. However, mechanical site preparation to establish pine stands in certain wetland types is prohibited by the Corps and EPA (Anon. 1996). Most State BMP manuals do not provide much guidance about ditches or drainage. However, many individual forest management companies have developed their own wetland BMPs, and these often stipulate basal area or canopy requirements for wetlands. Silvicultural practices in and around wetlands, therefore, vary greatly from landowner to landowner.

Harvesting and Site Preparation Practices

During harvesting, log decks and skid trails become temporary areas of bare soils. As with roads, the water-quality goal for managing these areas is to limit their hydraulic connectivity to streams. On any site, the number of log decks should be minimized and their distance to streams should be maximized. BMPs call for minimizing the number of temporary stream crossings and for using water bars to disperse runoff from skid trails. Soil rutting should be avoided and minimized. Equipment that exerts low ground pressure is recommended for wet sites. Skid trails on wet sites should be matted with a layer of limbs and branches over which equipment will operate.

BMPs require that mechanical site preparation (plowing, bedding, ripping) be done along contours to impede overland flow and minimize erosion. If debris is piled, it should be piled on contour to act as an organic silt fence. Most BMP recommendations preclude site preparation fires on steep slopes and call for cool fires that do not eliminate the duff layer, which is crucial in the prevention of surface erosion.

Effectiveness of BMPs

The National Association of State Foresters (NASF) tracks State BMP program performance. In its fourth survey, NASF (2001) reported that all 50 States have developed forestry BMPs. This is an improvement since NASF's 1990 survey, when only 38 States had BMPs. The national rate of BMP implementation is 86 percent. Half of the 22 States that monitor BMP implementation had overall BMP implementation rates of more than 90 percent (average 94 percent). A few States reported implementation rates below 80 percent. In addition to monitoring implementation, many States have also conducted assessments of BMP effectiveness. These investigations have found that BMPs are highly effective in protecting water quality during forestry operations (Adams and others 1995, Keim and Schoenholtz 1999, Kochenderfer and others 1997, Vowell 2001, Williams and others 1999). However, different States use different methods to survey the effectiveness of BMPs.

Sediment and Flow

In a study conducted on a watershed triplet in eastern Kentucky, Arthur and others (1998) found that clearcutting without BMPs increased suspended sediment loads by a factor of 30 in the 17 months during and following treatment and that clearcutting with BMPs increased sediment loads by a factor of 14 during this period. The effect of clearcutting on sediment fluxes disappeared after 5 years. The increase in load was attributable partly to increases in water yield (138 percent without BMPs and 123 percent with BMPs, respectively, in the first 17 months after harvest), but was attributable mostly to increases in suspended sediment concentrations. Most of the streamflow effect also disappeared within 5 years of treatment, although some flow effect was detectable when the study was completed 9 years after harvest.

Wynn and others (2000) analyzed a watershed triplet in eastern Virginia. They found that median storm total suspended sediment (TSS) concentrations increased by a factor of eight after clearcutting without BMPs. After site preparation, median storm TSS concentrations in the no-BMP watershed were 13 times as great as they had been prior to harvest. In the watershed in which BMPs were employed, there was no increase in TSS when TSS was converted for climatic variations observed in the control.

Using a cross-landscape comparison of first-order watersheds with complete randomized block design, Keim and Schoenholtz (1999) compared water quality for four treatments: (1) unrestricted harvest, (2) SMZs with cable thinning, (3) no-harvest SMZs, and (4) reference. Harvesting was group selection of hardwood species on Mississippi loessal bluffs with steep slopes with highly erodable soils. Grab sample and machine-composited TSS concentrations were higher in the watersheds with unrestricted harvest and with cable-thinned SMZs than in the references. TSS concentrations in the no-harvest SMZ watersheds were not different from these in references. Keim and Schoenholtz concluded that BMPs should focus on eliminating machine traffic within 10 m of streams.

Nutrients

Arthur and others (1998) found that mean nitrate concentrations rose from < 1 mg/L to almost 5 mg/L in the first 3 years after harvesting. The nitrate pulse was similar in no-BMP and in BMP watersheds. Concentrations of PO_4^{3-}, K^+, Ca^{2+}, Mg^{2+}, Na^+, SO_4^{2-}, and alkalinity did not respond detectably to harvesting. Wynn and others (2000) observed a similar pulse of NO_3^- from their no-BMP watershed but not from their BMP watershed. They found that total P loadings increased in the no-BMP watershed, but that this was explained by P bound to sediment, and was not soluble P.

The use of controlled drainage to improve water quality has been studied in a poorly drained loblolly pine plantation on the North Carolina Coastal Plain. Amatya and others (1998) reported that controlled drainage with a raised weir at the field outlets reduced annual export of TSS, NO_3+NO_2-N, and total Kjeldahl N by as much as 57 percent, 16 percent, and 45 percent, respectively, by reducing drainage volume and peak flow rates. The annual total P and NH_4-N loadings were also reduced by 7 to 72 percent. Amatya and others (2002) examined the effects of controlled drainage with a raised weir and an orifice on water quality. The authors reported that this system reduced flow volume and peak rates, and sediment and P loading, but had limited effects on other water-quality parameters.

Water Temperature

Changes in surface water temperatures as a result of forest harvesting activities conducted in riparian areas can have dramatic effects on aquatic biota (Wallace 1988, Webster and others 1988). Shading provided by trees in forested riparian areas cools aquatic habitats and moderates temperature fluctuations by insulation (Swift and Messer 1971). Intensive harvesting of riparian areas has been shown to increase maximum daily stream temperatures from 5 to 10 °C (Lynch and others 1985).

Because harvesting in riparian areas increases water temperatures and affects aquatic biota, BMPs have been designed to minimize changes in water temperature. BMPs have been developed that allow some overstory removal (generally ~ 25 to 50 percent) within riparian areas of perennial streams. Although many studies have shown stream temperature effects following intensive harvesting near streams (National Council for Air and Stream Improvement 2000), few have been designed to test the efficiency of these BMPs in moderating water temperature impacts of harvesting in ripirain areas. Within unharvested riparian areas, 15- to 30-m riparian buffer widths provide 85 to 100 percent effectiveness in mitigating increased solar radiation (National Council for Air and Stream Improvement 2000). Studies in northern Florida found no stream temperature increases in harvested riparian areas ranging from 10.6 to 60.9 m in width when 50 percent of the canopy was removed in the zone and a stringer of trees was left along the stream

(Vowell 2001). At the Fernow Experimental Forest in West Virginia, there were small (~1 °C) increases in stream temperature following a "light" removal of timber in a riparian zone 20 m in width (Kochenderfer and Edwards 1990). At Coweeta in North Carolina, the removal of 22 percent of the basal area had no effect on stream temperature (Swift and Messer 1971). Thus, the few studies of the effects of BMP design on stream temperature in the Southeast do suggest that 25 to 50 percent basal-area reductions within the riparian area lead to little or no increase in stream temperature.

Aquatic Biota

Forestry BMPs, especially those related to riparian areas, have been designed to lessen the impact of harvesting activities on water quality, and protection of water quality is generally considered to be a surrogate for the protection of aquatic communities. Therefore, the condition of aquatic communities has rarely been assessed directly during BMP effectiveness studies.

Intensive forest harvesting or land clearing in riparian areas increases insolation, raises stream temperatures, increases flows, increases both stream sediment and nutrient loads, and generally leads to greater primary productivity and shifts in faunal communities (Barton and others 2000, Richards and Hollingsworth 2000, Wallace 1988).

In northern Florida, there were no changes in habitat or stream condition index, which is based on macroinvertebrate populations, when 50 percent of the riparian overstory was removed and trees adjacent to the stream were not removed (Vowell 2001). In South Carolina, studies indicate that riparian BMPs, when implemented, have little to no effect on stream habitat and macroinvertebrate communities (Adams and others 1995). However, when riparian BMPs are not implemented or are implemented incorrectly, stream habitat and macroinvertebrate populations are affected negatively.

MODELING TOOLS FOR EVALUATING THE EFFECTS OF FOREST MANAGEMENT

Computer models cannot replicate the complex processes that take place in forests, but they are powerful and effective tools in forest management. If they are used properly, models can help us understand the processes and synthesize data at different scales, and may be a cost-effective tool for answering "what-if" questions. Because the Coastal Plains and the uplands have different hydrologic processes, we classify the existing computer models as wetland or upland models according to their applicability. Only those models that were developed for or have been applied to southern forest ecosystems are reviewed.

DRAINLOB and DRAINWAT

The DRAINLOB model (McCarthy and others 1992) is a forest version of the DRAINMOD-based models that were well tested for poorly drained (ditched) conditions in North Carolina and around the world. It is a physically based, lumped, and field scale hydrologic model. Using approximate analytical methods, the model predicts the full daily forest hydrologic cycle including rainfall interception, infiltration, subsurface drainage, surface runoff, ET, and soil water storage based on an hourly water balance conducted for a soil profile at the midpoint between two parallel ditches. ET is simulated using the Penman-Monteith method. Subsurface drainage rate is computed using a "table lookup" procedure that employs tabulated results of numerical solutions to the nonlinear Boussinesq equation. McCarthy and Skaggs (1992) applied DRAINLOB to evaluate the long-term water budget and hydrologic impacts of water-management practices for thinned and unthinned regimes of a pine plantation. This model has been modified to DRAINWAT by the addition of flow routing algorithms, and subsequently applied to forested watersheds with multiple fields and ditches (Amtaya and others 1997b). Major outputs from DRAINLOB and DRAINWAT include ground-water level and total outflow at the field edge and the watershed outlet on a daily basis.

FLATWOODS

The FLATWOODS model is classified as a physically based, distributed, and watershed-scale surface ground-water model (Sun and others 1998a). It was developed and tested specifically to examine the hydrologic impacts of forest harvesting in the heterogeneous cypress-pine flatwoods landscape (Sun and others 1998b). The model simulates the full daily hydrologic cycle of each uniform segment or cell of a watershed and links each cell with shallow ground-water flow. The vertical unsaturated water flow is computed using a simplified Darcy's equation while the 2-D lateral ground-water flow is simulated by the standard ground-water flow equation with Dupuit assumptions. ET is the sum of canopy interception, soil and surface water evaporation, and tree transpiration. Potential ET (PET) is computed using the temperature-based Hamon's PET model. The interception is modeled as a function of leaf area index (LAI), precipitation, and PET. Soil and surface evaporation are modeled as a function of LAI and ground-water level. Transpiration consumes the rest of the PET, but is limited by soil moisture status. Total outflow that is affected by averaged ground-water table and saturated areas is calculated using an empirical power function derived from experimental data. Major outputs from this model are total daily flow and distributed ground-water levels. Because this model explicitly simulates the ground-water fluxes, it has potential applications to isolated wetland systems, e.g., Carolina bays, that are common in the Coastal Plain in the South.

WETLANDS

WETLANDS (Mansell and others 2000) was developed to simulate the dynamic linkages of ground water and surface water in isolated depressional wetlands, such as cypress swamps. It is an altered form of the VS2DT model, a two-dimensional water and solute transport model for variably saturated media (Healy 1990). Radial symmetry and cone-shape geometry was assumed for the seasonally flooded wetland that is surrounded by uplands. ET is estimated by the Priestley-Taylor equation from a minimum set of daily weather data. Water-table levels in the wetland, lateral, and vertical water movement are simulated by solving two coupled equations simultaneously: the Richards equation and the water-balance equation for the wetland-upland system. Major model outputs are water-table distribution across the wetland-upland continuum, ET, and soil moisture dynamics.

PROSPER

PROSPER is a lumped parameter model that was developed to estimate water stress for an upland forest stand by describing the atmosphere-soil-plant water-flow processes (Goldstein and others 1974). As the major component of the model, ET is modeled by a combined energy balance-aerodynamic method. Soil water is depleted by ET, and its movement between layers is modeled by Darcy's law and mass balance by an approximate numerical solution. Major climate data requirements for this model include daily values of precipitation, air temperature, relative humidity, solar radiation, and wind speed. Other parameters include mean values of albedo; leaf area of vegetation; typical resistance values for water movement through soils, plants, and atmosphere; soil hydraulic conductivity; and root distribution. Major outputs from the model are daily ET and soil water potential at different soil layers. PROSPER has been used to examine the hydrologic effects of forest conversions (Swift and Swank 1975) and climate change (Vose and Maass 1999).

ANSWERS (Forest Hydrology Version)

The ANSWERS model represents the first generation of physically based, spatially distributed, watershed-scale models that were designed to simulate the effects of agricultural BMPs on runoff and sediment loss from agricultural watersheds (Beasley and Huggins 1981) on a storm event basis. Thomas and Beasley (1986a) modified the original ANSWERS model with the goal of giving forest managers a tool for evaluating management practices (logging and prescribed burning). Major modifications include the addition of interflow components of seepage and macropore or pipe flow at the surface soil layer, alternation of the canopy interception submodel, and estimation of initial soil moisture distribution. Major input data requirements include soil physical characteristics, topography (digital elevation model), and rainfall intensity. Major outputs are storm flow volumes and storm hydrograph.

This event-based model has been validated successfully on five upland watersheds in the upper Coastal Plain in Mississippi (Thomas and Beasley 1986b). However, unsatisfactory results were reported when the model was tested on two steep mountain watersheds at Coweeta in North Carolina where soils and topography were believed to be unique and baseflow rates are relatively higher than those of the Piedmont watersheds.

PnET-II

The PnET-II model is a lumped-parameter, monthly-time-step, generalized stand-level model that describes the C and water dynamics of mature forests (Aber and others 1995). It simulates both C and water cycles in a forest ecosystem using simplified algorithms that describe key biological and hydrologic processes. This model has been validated and modified for southern upland forest ecosystems (Aber and others 1995, Liang and others 2002), southern pines (McNulty and others 1996, Sun and others 2000a), and hardwoods (Hanson and others 2003). It has been employed to assess the potential effects of climate change on forest hydrology at a regional scale (McNulty and others 1996). Input parameters for vegetation, soil and site locations, and climate may be derived from the literature or measured at a local study site. Stand-level vegetation parameters include those regulating physiological and physical processes such as photosynthesis, light attenuation, foliar N concentration, plant and soil respiration, and rainfall interception. Only one soil parameter, soil water-holding capacity, is required. Climate input variables include minimum and maximum monthly air temperature, total monthly photosynthetic active radiation, and total monthly precipitation. The PnET-II model closely integrates hydrology with the biological processes. ET is defined as the sum of plant transpiration and canopy interception. Transpiration is simulated as a function of C absorbed during photosynthesis and water-use efficiency. The model simulates the C cycle by tracking absorbed C during photosynthesis; allocation of C to foliage, wood, and roots; and respiration from leaves, stems, and roots. The hydrologic cycle is simulated by the water-balance equation. Water that is not subjected to ET eventually ends up as water yield. Major model outputs include annual forest net primary productivity, monthly and annual ET, and water yield.

SCALING-UP WATERSHED HYDROLOGY FOR REGIONAL ASSESSMENT

There have been several attempts to generalize experimental results from small watersheds to guide regional forest management. Douglass and Swank (1972) and Douglass (1983) derived a general empirical equation to estimate water-yield response to forest management in the Appalachian Mountains. However, the model does not include precipitation as an independent variable, and thus has limited applicability for other similar mountain regions. Huff and others (1999) presented empirical methods and a computer program for evaluating water-yield impacts of proposed forest or vegetation thinning over a large area. They tested the system in the Central Sierra Mountains in California and found that the size of the management area has an important bearing on water-yield response. It is not known how well the modeling system works for the Southern United States. Sun and others (2002) tested and modified a conceptual catchment-scale ET model (Zhang and others 2001) using forested watershed hydrologic data from across the Southern United States. A Geographic Information System (GIS) was used to integrate regional databases for forest cover, climate, topography, and predicted potential ET at a 4-km resolution. The regional analysis shows that hydrologic response, as represented by water-yield increase, varies greatly across the complex physiographic gradients in the Southern United States (fig. 19.5).

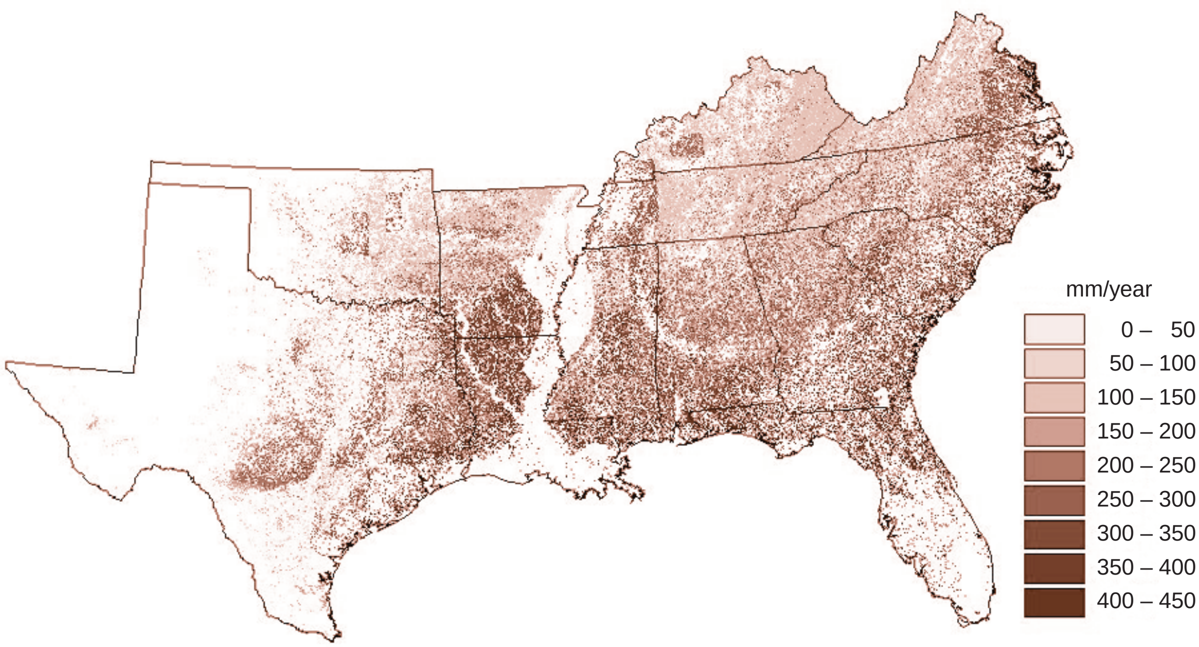

Figure 19.5—Predicted long-term annual water yield response to forest removal at a 4-km resolution. Values are displayed at a 30-m land use/landcover resolution.

SUMMARY AND RECOMMENDATIONS

Research findings regarding water-yield responses to deforestation and aforestation in the Southern United States are consistent with those of studies conducted elsewhere (Bosh and Hewlett 1982, Stednick 1996, Whitehead and Robinson 1993). The greatest increases in water yield occur immediately following harvesting. As more tree cover is removed, the higher the response of water yield and ground-water table increases. The conversion of a forest to a cover type that requires less water, such as agriculture, grazing, and fodder production, significantly increases water yield. Conversely, conversion of a forest to another forest type that intercepts more water, such as conversion from mixed oak-hardwood to eastern white pine, significantly reduces water yield. Regrowth or reforestation increases the interception capacity and consumptive use of water, thus reducing streamwater yield. Large-scale forest manipulation to increase water availability is not practical due to water quality and other ecological concerns. The overall water-quantity effect of silvicultural operations is much less in wetlands than in areas having greater relief and shallow soils. Compared to hilly uplands, southern wetlands on the Coastal Plains or large flood plains have low ratios of runoff to precipitation (< 40 percent). Hydrologic responses of wetlands to tree removal are expected to be low because ET is often near potential and because management activities generally have only minor effects on ET. Wet-weather harvesting in forest wetlands often results in soil compaction, rutting, and churning, but hydrologic responses to forest management are much smaller than hydrologic responses to vegetation changes. Hydrologic recovery appears to be faster in wetlands than in uplands. Climate gradient also influences the effects of timber management on hydrology, because climate affects the recovery of vegetation and the way in which environmental conditions change as a result of forest disturbances. Effects of harvesting in colder and drier water regimes, such as those in the Appalachian Mountains, may last longer simply because it takes longer to establish a forest under such conditions. Intensity of forest management practices can also affect hydrologic responses. Stormflow volumes and peak flow rates are expected to increase when forests are cut heavily or converted to other land uses, especially during nongrowing seasons. Few studies have addressed the relations between forest cover change and stormflow relations. Such studies are important in forest hydrology in the 21st century, when urbanization activities have been intensifying.

Forest watersheds produce better water quality than other land uses. Silvicultural practices in the South cause relatively minor water-quality problems. Forest removal, prescribed burning, and chemical applications cause immediate responses in water-chemistry concentration and loading, but these effects diminish rather quickly. Reductions in nutrient export occur following conversion of grasses to forests. Reductions in forest standing crop as a result of insect outbreaks also increase nutrient export. The major water-quality concern related to forestry activities is sedimentation. The impact of forest harvesting on sediment yield is directly related to harvesting methods and road building. Foresters in modern times have been improving their methods to protect water quality. When BMPs are used, forest harvesting does not necessarily cause stream sedimentation. Numerous road construction practices that minimize erosion and sedimentation have been identified. The cumulative effects of upstream land use conversion and changes in land use composition over time on streamwater quality can be mitigated by proper forest management.

Forestry BMPs have been seen as the best tools for controlling nonpoint pollution and protecting water quality. However, the riparian BMPs for the Southern States are not based solely on the best available knowledge; rather they are the product of battles among forestry groups, environmental groups, and policymakers within each State. Our knowledge of the effects of existing BMPs is imperfect, and we could probably design better BMPs if our knowledge of these effects were increased.

Recent studies show that sediment, nitrate release, stream temperature, and biotic response continue to be issues when BMPs are implemented. Pesticide movement from silvicultural operations remains largely unstudied. Process-based studies are needed to develop specific information on how, when, and where silvicultural BMPs fail to provide adequate protection of water quality. Models and regional analyses are needed to evaluate how BMPs perform at regional scales (National Council for Air and Stream Improvement 1999b). A key question in BMP development and TMDL implementation is: How much nonpoint-source pollution is too much? For instance, what are the ecological thresholds for sediment and nitrate concentrations, and how do these thresholds differ for river and lake systems? Should we protect ephemeral streams during forestry operations, and how? In general, there is a need for research on the linkages between hydrology, water quality, and biological responses (National Council for Air and Stream Improvement 1999a).

There have been few studies of the cumulative effects of different land uses on hydrology and water quality at a large basin scale. Such studies (Trimble and Weirich 1987, Williams and others 2002), which are reviewed in this chapter, present new views on the role of forests that may contradict presumptions or results from traditional small watershed-scale studies. With the development of new technology, such as remote sensing, GIS, and explicit regional hydrologic and water-quality models, the roles of forests and effectiveness of forestry BMPs in reducing nonpoint pollution on complex large landscapes can be evaluated.

Computer models are useful tools for generalizing and scaling-up the findings from individual studies and applying the knowledge to management practices. Most of the existing watershed-scale forest hydrology models lack nutrient, soil erosion, and sediment transport components. There is a need for research that will produce physically based, distributed watershed-scale models that couple hydrologic processes, forest nutrient cycling, and soil erosion on forest lands. The complexity of forested watershed processes has limited progress in development of such models (Band and others 2001, Zhang and others 2002). We should recognize that distinct hydrologic processes exist across the different physiographic provinces in the South. Different types of models in the South are needed. For example, ground water-dominated systems in the Coastal Plain require a model that can describe ground water water-table dynamics, while those upland systems dominated by overland flow or subsurface unsaturated flow require models that can describe hillslope processes. Several models such as WEPP (Nearing 1989), ANSWERS-2000 (Bouraoui and Dillaha 1996), and SWAT (Arnold and others 1998) have been widely used for modeling nonpoint pollution from agricultural lands, but significant modification and revisions are needed before they can be applied to forest systems.

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