Streamside Management Zones Affect Movement of Silvicultural Nitrogen and Phosphorus Fertilizers to Piedmont Streams

Joseph M. Secoges, Wallace M. Aust, John R. Seiler, C. Andrew Dolloff, and William A. Lakel

ABSTRACT

Forestry best management practices (BMP) recommendations for streamside management zones (SMZs) are based on limited data regarding SMZ width, partial harvests, and nutrient movements after forest fertilization. Agricultural fertilization is commonly linked to increased stream nutrients. However, less is known about effectiveness of SMZ options for controlling nutrient movements after silvicultural fertilization. Diammonium phosphate and urea were applied to 12 subwatersheds in 3-year-old loblolly pine (*Pinus taeda L.*) plantations in the Virginia Piedmont. Three replicates of four SMZ treatments were superimposed on 12 subwatersheds in a previous SMZ harvest sediment study (7.6-m SMZ, 15.2-m SMZ thin, 15.2-m SMZ, and 30.5-m SMZ). Surface, near-surface, subsurface, and stream water samples were collected monthly for 1 year and analyzed for nitrate (NO_3^-), ammonium (NH_4^+), and orthophosphate (ortho-P). Transected measurements from streamside to fertilized plantations allowed interpretations of spatial nutrient measurements across SMZs. When compared with wider SMZs, 7.6-m SMZs had 3–10× surface water NO_3^-, 3–6× near-surface water NO_3^-, and 1–2× more stream water NO_3^-. No significant differences were detected for NH_4^+ for any SMZ treatment. The 15.2-m SMZ thin had small but significant increases (2–8×) in surface runoff for ortho-P relative to other SMZ treatments, perhaps because of increased surface water movement along thinning corridors. Across all SMZ treatments, comparisons of stream edges with fertilized stands indicated NO_3^- reductions of 33–98%, NH_4^+ reductions of 68–97%, and ortho-P reductions of 70–98%. A 39% rainfall deficit during the study influenced results, but conventional SMZs $\geq$ 15.2 m protected streams from fertilization nutrient increases.

Keywords: riparian forests, fertilization, forest operations, best management practices, water quality

Approximately 0.4 million ha per year^{-1} of nitrogen (N) and phosphorus (P) fertilizers are applied to southeastern forests to increase site productivity (Fox et al. 2007a, 2007b). Increased nonpoint source pollutants (commonly N and P) caused by agricultural operations (Daniels and Gilliam 1996, Lowrance et al. 1997) are associated with stream eutrophication, lower oxygen levels, and degraded habitat for aquatic organisms (Vitousek et al. 1997, Binkley et al. 1999, Dosskey et al. 2010). Forest fertilization occurs less frequently and at lower rates than agricultural operations, but silvicultural fertilization has potential to increase nonpoint source pollutants (Neary et al. 1989, Binkley and Brown 1993, Binkley et al. 1999, McBroom et al. 2008).

Forestry best management practices (BMP) were developed by states to prevent or minimize nonpoint source pollutants as recommended by the Federal Water Pollution Control Act of 1972 and its amendments (Ice et al. 1997, Ice 2004, Shepard 2006). Streamside management zones (SMZ) are an important forestry BMP in managed forested watersheds for control of nonpoint source pollutants (Lynch et al. 1985, Aust and Blinn 2004, Broadmeadow and Nisbet 2004, Correll 2005, Dosskey et al. 2010, Anderson and Lockaby 2011a, 2011b). Specific water quality protection benefits of SMZs generally include sediment trapping (Beasley et al. 1986, Keim and Schoenholtz 1999, Rivenbark and Jackson 2004, Lakel et al. 2010), nutrient uptake and transformations (Lowrance 1992, Lowrance and Sheridan 2005), nutrient storage (Daniels and Gilliam 1996), reduction of thermal pollution (Swift and Messer 1971, Hewlett and Fortson 1982, Aust et al. 2011), and stabilization of streambanks (Allmendinger et al. 2005). SMZs may protect water quality simply by providing an obvious boundary that minimizes the inadvertent direct application of fertilizer to streams (Liechty et al. 1999, Fox et al. 2007b).

SMZs are typically viewed as a critically important BMP because SMZs are the last protective BMP between silvicultural operations and streams if other BMPs fail (Aust and Blinn 2004, Anderson and Lockaby 2011a). Although the importance of SMZs to water quality is recognized, there is no clear consensus regarding SMZ width and efficacy. Three reviews of SMZ literature found that SMZ width recommendations varied from 4.5 to 90 m (Castelle et al. 1994, Aust and Blinn 2004, Broadmeadow and Nisbet 2004). The lack of consensus regarding SMZs is also reflected by the variety of SMZ recommendations provided by states in the Piedmont region. For example, current Piedmont SMZ recommendations are 10.7 m for

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This article uses metric units; the applicable conversion factors are: meters (m): 1 m = 3.3 ft; millimeters (mm): 1 mm = 0.039 in; milliliters (mL): 1 mL = 0.061 in.^{2} (dry) = 0.27 fluid dram (liquid); milligrams (mg): 1 mg = 0.015 gram; kilograms (kg): 1 kg = 2.2 lb; hectares (ha): 1 ha = 2.47 ac.

perennial streams in Alabama (Alabama Forestry Commission 2007), 12.2–30.5 m in Georgia (Georgia Forestry Commission 2009), 9.1–61.0 m in North Carolina (North Carolina Division of Forest Resources 2006), 12.2 m in South Carolina (South Carolina Forestry Commission 1994), and 15.2 m in Virginia (Virginia Department of Forestry 2002). State BMP recommendations do have consensus regarding partial harvests and fertilization in that manuals indicate that partial harvests of SMZs are acceptable and that fertilization should not be conducted within SMZs.

Sediment is generally considered to be the most important pollutant associated with forest operations; thus, more studies have examined the effect of SMZ widths on sediment (Anderson and Lockaby 2011a, 2011b). However, sediment trapping also has relevance to nutrient removals because N and P trapping has been linked to sediment trapping (Barling and Moore 1994). Anderson and Lockaby (2011b) evaluated the available research regarding SMZ effectiveness for sediment control in the Southeast and concluded that there is still considerable uncertainty. Keim and Schoenholtz (1999) evaluated stream sediment with four types of SMZs located adjacent to harvests in the Loess Hills of Mississippi. Their SMZs included no buffer, partial harvests with no traffic, no harvest SMZs, and a no harvest control. The unrestricted harvests increased stream sediments relative to the nonharvested area streams. Both other SMZ treatments increased the variability of the stream sediments. Overall, they concluded that SMZs are most effective in controlling sedimentation when the forest floor is undisturbed. Lakel et al. (2010) evaluated sediment inputs with different width SMZs adjacent to clearcut harvests on the same watersheds as this study. They concluded that the SMZs $\geq$ 15.2 m provided adequate sediment prevention and that the SMZs trapped approximately 97% of delivered sediment before it reached the stream. Rivenbark and Jackson (2004) evaluated sediment trapping of various width SMZs across 30 sites in the Georgia Piedmont. They concluded that a 30.5-m-wide SMZ would be sufficient to prevent 86% of the sediment. Ward and Jackson (2004) evaluated Piedmont SMZs and conclude that 75% of eroded material is maintained on the site, whereas approximately 25% of eroded material is transferred to the SMZ.

Fewer forest studies have examined SMZ width with specific regard to nutrient movement from forest management activities. Mayer et al. (2007) reviewed 45 research projects that investigated a variety of forested and other vegetated buffers adjacent to agricultural and silvicultural operations and found that buffers $\geq$50 m consistently removed more N than narrower buffers (0–25 m). They hypothesized that the wider width SMZs had greater area for root uptake of N or soil conditions that favored denitrification. Barling and Moore (1994) evaluated sediment and nutrient removal by SMZs and concluded that SMZs are less effective for removing nutrients than sediment. Their explanation was that sediment removals are less effective for smaller particles, but the smaller particles have a higher proportion of attached nutrients. Lakel et al. (2006) examined the effect of SMZ width on stream nutrients after forest harvesting in the Piedmont region. Streams had buffers of 7.6-, 15.2-, and 30.5-m widths. Data were limited by drought conditions, but the few available storm water samples indicated that all SMZ widths were effective in minimizing N and P movements after clearcutting.

Effects of partial harvests within SMZs have also received limited research. Clinton (2011) examined the partial harvests of Appalachian hardwoods with SMZs of 0-, 10-, and 30-m SMZs and compared stream values to those of a nonharvested control. Stream N values doubled for the harvest having the 0-m SMZ, but streams with the 10- and 30-m treatments had minimal responses. The overall conclusion was that the 10-m-wide SMZ was of sufficient width to adequately protect these streams from sediment and nutrient inputs after harvesting.

SMZ specifications necessary to minimize effects of forest fertilizations are also uncertain. Binkley et al. (1999) reviewed over 70 studies related to the potential effect of forest fertilization on water quality and concluded that fertilization generally increases N and P levels in streams. Overall, data that they examined indicated that narrower or thinned SMZs allow slight increases in fertilizer that reaches streams. More specifically, McBroom et al. (2008) evaluated the effects of extensive and intensive site preparation with fertilization on N and P losses to adjacent streams in East Texas with standard width SMZs. They detected increased runoff losses of both N and P fertilizer in the intensively site prepared watersheds, but concluded that the losses did not adversely decrease stream water quality. Overall, they concluded that the BMP, which included SMZs, were effective for minimizing fertilization effects on water quality. Daniels and Gilliam (1996) examined the different SMZ widths that may be needed for sediment versus fertilizer nutrients. They evaluated the effectiveness of grass and riparian forest buffers in the Piedmont for removal of sediment, N, and P from water delivered from fertilized agricultural fields. Both types of buffers removed 80% of sediment loads, but only 50% of N and P. The authors noted that losses were exacerbated by high precipitation events and ephemeral drains that bisected the riparian buffers. Hutchens et al. (2004) evaluated SMZ research with emphasis on logging effects on water quality and aquatic organisms and concluded that remarkably few studies had evaluated specific BMP effectiveness. Edwards and Willard (2010) reviewed forest BMP research that could be used to calculate BMP effectiveness (including SMZs). They found only three studies in the eastern United States facilitated these calculations: one in the Allegheny Plateau (Kochenderfer and Hornbeck 1999), one in the Cumberland Plateau (Arthur et al. (1998), and one in the Coastal Plain (Wynn et al. (2000)). These studies indicated that BMP efficiencies range between 53 and 94% for sediment removal, 60 and 80% for total N removal, and 85 and 86% for P removal. Wynn et al. (2000), the only study that specifically evaluated nitrate (NO_3^-) found a low removal efficiency of 12%. They speculated that removal processes in surface waters that were linked to sediment trapping appeared to be more efficient than those in subsurface waters. SMZs are important BMP.

In addition to efficacy, SMZ width and partial harvests should consider potential costs to landowners. Brian et al. (2004) reviewed riparian buffer research and found that riparian buffers $\geq$30 m were adequate for pollutant removal but lamented the lack of studies that focused on the pollution control provided by narrower buffers. They concluded that additional research is needed for smaller SMZ widths that are more likely to be readily adopted by landowners. Broadmeadow and Nisbet (2004) noted that wider SMZ widths impose substantial costs to landowners and that SMZ width should be balanced with SMZ costs. Lakel et al. (2010) similarly pointed out that forest SMZs of different width can translate into substantial financial considerations to landowners.

Overall, research indicates that SMZs can positively reduce pollution-associated forest management activities. However, relatively few studies have examined the effects of different widths of SMZs on

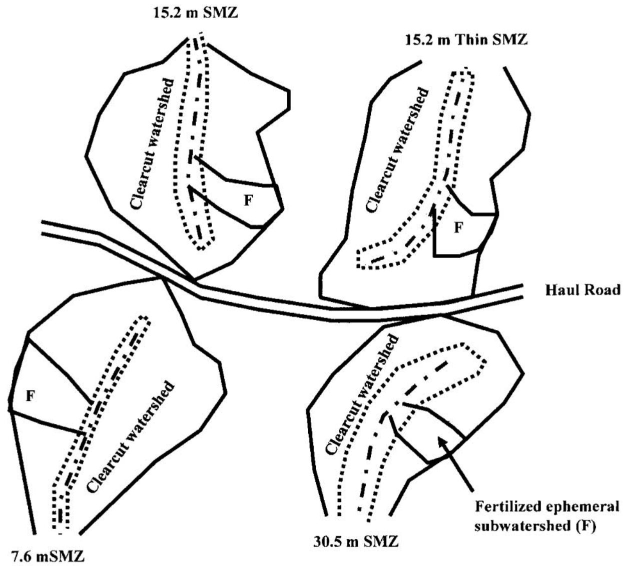

Figure 1. General location of the 12 SMZ treatment study sites located in the Piedmont region, Buckingham County, Virginia.

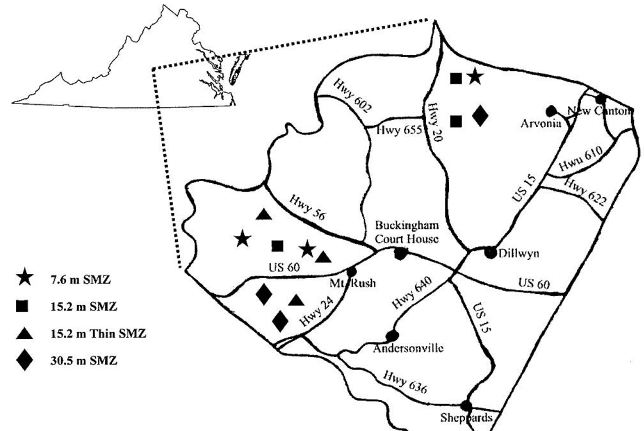

Figure 2. Idealized layout (not to scale) of 4 of 12 watersheds, fertilized watersheds, and SMZ treatments.

water quality after forest stand fertilization (McBroom et al. 2008, Dosskey et al. 2010, Anderson and Lockaby 2011b). The primary goal of this research was to examine the effect of SMZ widths on movement of (NO_3^-), (NH_4^+), and orthophosphate (ortho-P) fertilizer from recently fertilized watersheds, through SMZs, and into streams in the Piedmont region. A secondary goal was to compare the efficiency of nutrient removals from the fertilized stands, SMZ edge, and streamside. A third goal was to use transect water samples to examine the relative efficiency of removals between fertilized areas, SMZ edges, and streamside samples. Specific study objectives were to

1. Compare the NO_3^-, NH_4^+, and ortho-P levels in the surface, near-surface, subsurface, and streams for four levels of SMZ widths (7.6, 15.2 with thinning, 15.2, and 30.5 m) adjacent to 3-year-old, fertilized, loblolly pine plantations.
2. Compare the values of NO_3^-, NH_4^+, and ortho-P from the fertilized plantation across transects within SMZ treatments to the stream.
3. Use the transect data to calculate removal efficiencies for NO_3^-, NH_4^+, and ortho-P where significant transect values were detected for surface, near-surface, and subsurface waters.

Methods

This research was superimposed on a portion of the clearcut watersheds and SMZ treatments that evaluated SMZ sediment removal after harvesting and site preparation as reported by Lakel et al. (2006, 2010). The original study used 16 first-order intermittent and perennial streams with watersheds averaging 30 ha in size. Twelve of these watersheds were selected for use in the fertilization study. Streams in the upper end of the watershed are intermittent and transition to small perennial streams toward the lower end of the watersheds. All sites are in the upper Piedmont region (Fenneman 1938) of Buckingham County, Virginia, on forestlands managed as industrial, loblolly pine (*Pinus taeda*) plantations (Figure 1). The sites had been cleared for agricultural crops in the 1800s and reverted to old-field forests by the early 1900s, and pine plantations replaced the old-field forests in the 1960s (Trimble 1974, Cowell 1998). Terrain is rolling with sideslopes ranging between 15 and 40%. Average annual precipitation for this region is 1,066 mm. Average winter (December to February) temperature is 3.3° C and average growing season (April to September) temperature is approximately 21.1° C. Upland soils are primarily Fairystone channery loams (clayey skeletal parasesquic and mesic typic hapludults) and Spears Mountain silt loams (fine, mixed, semiactive, and mesic typic hapludults). Soil textures are gravelly loams to gravelly sandy loams over 1:1 clay subsoils (Easterbrook-Walker et al. 2003, Lakel et al. 2010). Riparian soils are dominated by the Hatboro loams (fine loamy, mixed active, nonacid, and mesic fluvaquentic endoaquepts; Lakel et al. 2010).

For the original sediment study, clearcut harvests for an SMZ and harvest-related sediment study occurred between summer 2003 and spring 2004 using typical rubber-tired feller-bunchers and grapple skidders (Lakel et al. 2010). The loblolly pine stands were between 23 and 28 years old at harvest. The use of feller-bunchers for the thinning operations in the SMZ resulted in thinning corridors within the riparian zones. Sites were subsequently site prepared with prescribed burning during fall 2004 (Lakel et al. 2010) and hand planted with loblolly pine seedlings in winter 2004 to spring 2005.

For the subsequent fertilization study (this article), one smaller contributing subwatershed (zero-order, ephemeral stream) was selected within each of 12 larger (first-order perennial streams) clearcut watersheds (Figures 2 and 3). Thus, the original four SMZ treatments were replicated three times. The 12 subwatersheds that received fertilization ranged from 0.2 to 1.4 ha. The subwatershed boundaries were defined as the area outside of the SMZ; thus, fertilized watershed sizes were independent of SMZ size. The selected subwatersheds were selected so that they had well-defined boundaries and concentrated flow pathways toward the SMZs.

The 12 subwatersheds in the fertilization study each had one of four SMZ treatments: (1) 7.6-m-wide SMZ with no SMZ harvest (7.6-m SMZ), (2) 15.2-m wide SMZ with 50% SMZ thin (15.2 thin SMZ), (3) 15.2-m-wide SMZ with no SMZ harvest (15.2-m SMZ), and (4) 30.5-m-wide SMZ with no SMZ harvest (30.5-m SMZ). During the third growing season (July 2007), diammonium phosphate and urea fertilizers were applied to the 12 subwatersheds at common industrial rates of 250 and 140 kg ha^{-1}, respectively, by

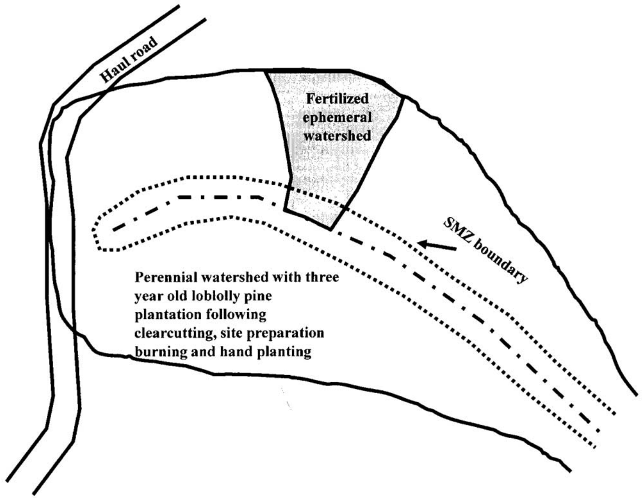

Figure 3. Idealized (not to scale) layout for 1 of 12 watersheds, fertilized subwatersheds, and SMZ boundary.

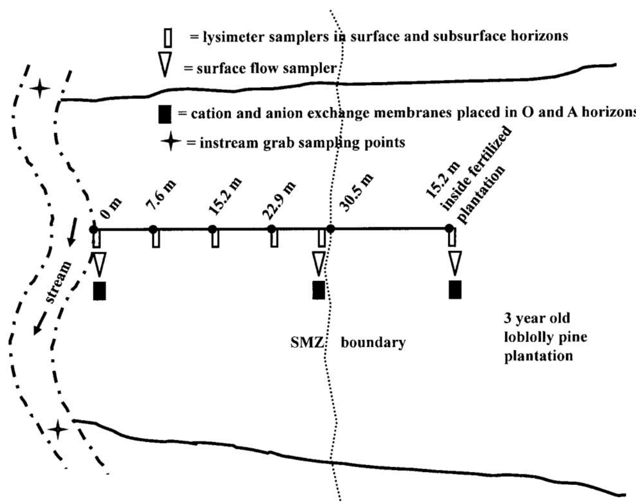

Figure 4. General sampling locations for the surface, near-surface, subsurface, and stream water samples for the 30.5-m-wide SMZ treatment.

all-terrain vehicle and by cyclone hand spreader. Application yielded 28 kg ha^{-1} of elemental P and 140 kg ha^{-1} of elemental N. These rates are representative of forest industry fertilization in the southeastern United States (Fox et al. 2007a, 2007b).

Four types of water samples were collected, including overland flow (surface water), O and A horizons (near surface), the argillic subsoil Bt horizons (subsurface), and stream water to determine the effects of SMZs on fertilization and nutrient movements. Nutrient movements from the fertilized clearcut toward and through the SMZs were examined by installing sampling stations at various distances in the fertilized clearcut and within the SMZs (Figure 4). The numbers of stations varied by SMZ width, but all SMZs had sampling locations located in the fertilized stand, SMZ edge and at streamside.

Two surface water samplers (Daniels and Gilliam 1996) were placed in the SMZ of each study subwatershed. One sampler was installed at the upslope SMZ edge and the second sampler was placed on the immediate streambank (Figure 4). Tarpaulins (0.5 × 0.5 m) were placed approximately 0.3 m above the surface samplers to minimize additions from throughfall and canopy drip. Surface water samples were collected every 2–3 weeks (except when there was no rainfall) and two subsamples from each canister were frozen in 20-mL scintillation vials until laboratory analysis could be completed. Each sampler was used to develop a data set over 12 months. Potentially, each sampler could have collected 36 samples, but limited rainfall resulted in as few as 20 samples.

For the near-surface movement of NH_4^+ and NO_3^-, cation and anion transfer membranes were inserted in the Oa and Ap horizons as described by Cooperband and Logan (1994), Subler et al. (1995), and Pratt and Fox (2009). These transfer membranes provide an index of nutrient movement across their surface, but they are reported in milligrams per square meter per day for the nutrient rather than concentrations. Transfer membranes were installed systematically across each SMZ (Figure 4). At each position (clearcut, SMZ dripline, and streamside) membranes were installed in sets of four (one cation and one anion membrane between the Oa horizon and Ap horizon and one cation and one anion membrane were inserted in the Ap horizon). Because of concerns about ion saturation potential, membranes were exchanged every 2 weeks for the first 6 weeks after fertilization. After it became evident that ion saturation was not an issue, exchange membranes were exchanged every 3–4 weeks for the remainder of the study. The Oa and Ap horizon data were combined to develop a data set potentially having 144 samples or 36 samples per SMZ treatment. Again, low rainfalls reduced individual SMZ samples to approximately 20 per station.

Subsurface water was monitored with one-bar, porous cup lysimeters (Soil Moisture Equipment Corp. 1999). Lysimeters were placed in pairs at the bottom of the Ap horizon and in the Bt horizon (approximately 0.6 m). The Ap lysimeters collected so few samples that their values were simply combined with Bt subsurface values. Each SMZ treatment had lysimeters stations placed within 1 m from the main stream, at the upslope edge of the SMZ, and in the fertilized stand approximately 15.2 m from the SMZ dripline (Figure 4). For the 30.5-m SMZs, lysimeter stations were placed relative to the stream channel at 0, 7.6, 15.2, 22.9, 30.5 within the SMZ, and 45.7 m (15.2 m within the clearcut; Figure 4). Subsurface water samples were collected from lysimeters with an irrometer-style vacuum pump into scintillation vials. Samples were promptly frozen in 20-mL scintillation vials until laboratory analysis could be conducted. Samples were collected approximately monthly for 12 months after fertilization. Because of dry sampling conditions and the infrequency of Ap lysimeter samples, the Ap data were combined with the Bt data. Potentially, the monthly samples could have resulted in a data set consisting of 144 samples (12 months × 3 replications × 4 treatments) or 36 samples per SMZ treatment, but low rainfall conditions resulted in approximately 20 samples.

Stream water grab samples (US Department of Education 1981, Danielson 2004) were collected from perennial streams 20 m above and 20 m below treatment watersheds to determine whether downstream samples were increased by fertilizer nutrients (Figures 3 and 4). Monthly samples from three replicates of four SMZ treatments (12 subwatersheds) provided 144 upstream and 144 downstream stream samples (36 samples for each treatment). The water samples were frozen until laboratory analysis for NO_3^-, NH_4^+, and ortho-P could be performed.

Water samples collected from surface canisters (surface water), lysimeters (subsurface water), and in-stream (stream water) locations were filtered using Whatman 42 qualitative filter strips and

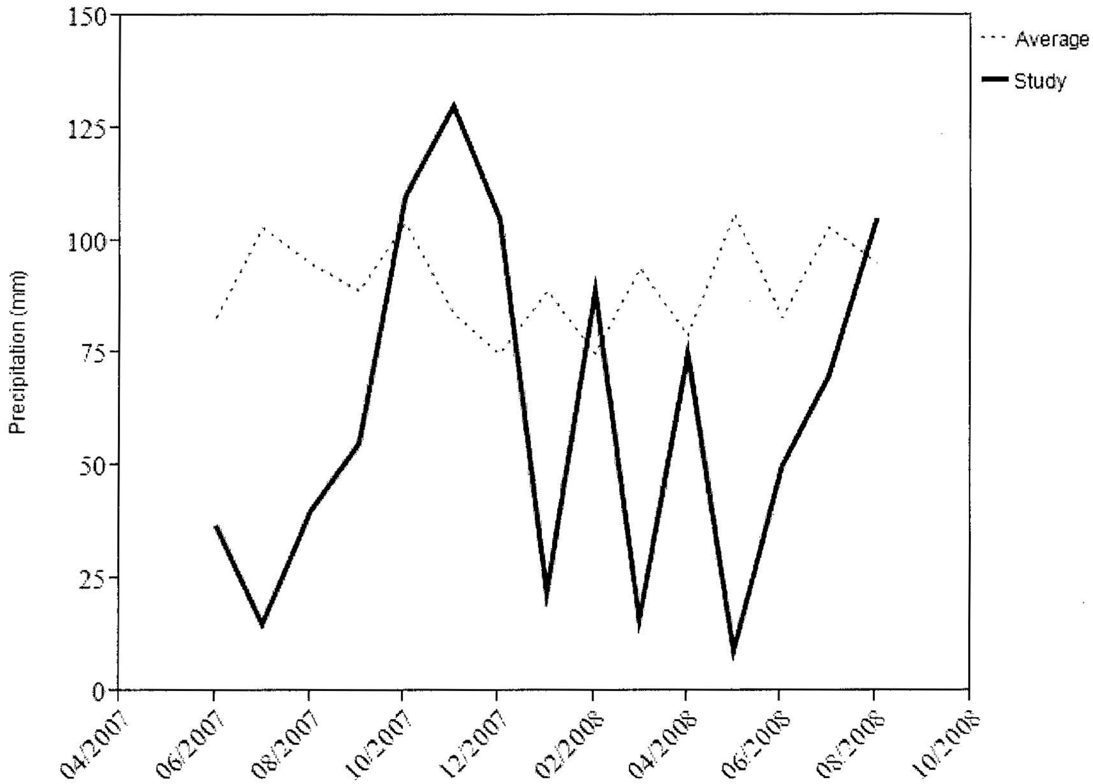

Figure 5. Monthly long-term precipitation averages (dashed line) versus monthly precipitation during the study period (solid line) for the period of the fertilizer SMZ study.

frozen until subsequent processing on an autoanalyzer3 (SEAL, Mequon, WI) for NO_3^-, NH_4^+, and ortho-P. After membranes (near-surface water) were returned to laboratory, membranes were brushed clean and placed in centrifuge tubes. Twenty-five mL of 1-M KCl solution were added to each centrifuge tube, covered with parafilm, and capped. The tubes were shaken for 30 minutes before the supernatant was filtered into 20-mL scintillation vials using Whatman Grade Two Qualitative Grade Circles and Sheets. Finally, the vials were placed in a freezer for later analysis. The membranes were then cleaned with deionized (DI) water to remove any remaining soil or organic matter. The process of one 5-minute interval of DI water, one 5-minute interval of HCl acid solution, and three 5-minute intervals of DI water in plastic containers on the shaker was repeated before the membranes were returned to the 1-M NaCl solution to be charged. Potassium chloride extractions from the Ionics exchange membranes were analyzed using a TRAACS 2000 auto-analyzer (Bran and Luebbe, Buffalo Grove, IL).

The SMZ treatments were analyzed as a completely randomized design with three replications of the four SMZ treatments (7.6-m SMZ, 15.2-m SMZ thin, 15.2-m SMZ, and 30.5-m SMZ) with repeated measures. For examination of movement across the SMZs, differences in sampling stations were considered within each SMZ with repeated measures (Steel and Torrie 1980, Gomez and Gomez 1984, British Columbia Ministry of Forestry 1995). Analysis of variance tests were conducted on NO_3^-, NH_4^+, and ortho-P samples from surface water, litter, subsurface, and stream water for treatments and sampling locations. These methods and statistical designs were used to accept or reject the null hypotheses that (1) differences in SMZ width do not affect applied fertilizer from entering the stream at an alpha level of 0.10 and (2) different sampling stations across the SMZ do not have different levels of nutrients. As significant differences were detected, means were separated using the Tukey honestly significant difference mean separation tests. All statistical analyses were performed using JMP 9 (SAS Institute 2010).

Results and Discussion

After fertilization in mid-July 2007, the central region of Virginia experienced below-average precipitation during much of the following year (Figure 5). Monthly measurements of on-site rain gauges indicated rainfall events during the study period were less frequent than usual (Figure 5), although study sites still experienced periods of intense precipitation. Typical long-term cumulative rainfall for the 15 months monitored is normally 1,585 mm, but study watersheds averaged 972 mm of precipitation ($\approx$39% deficit). Furthermore, rainfall was low in the month immediately after fertilization when greater nutrient movement was anticipated. The low precipitation certainly may have influenced fertilizer movement, as was noted by Governo et al. (2004) in their nutrient flux study.

Despite lower precipitation during the study, SMZ treatments had significant differences for surface, near-surface, subsurface, and changes in stream water (Table 1). For surface water, near-surface water, and stream samples, significantly higher NO_3^- values were found in the narrower 7.6-m SMZ than in the 30.5-m SMZ treatment. Values for the 15.2-m thin and 15.2-m SMZ tended to be transitional between the wider and more narrow SMZs. Vaidya et al. (2008) evaluated the effect of different widths and harvest levels within SMZs after timber harvests of mixed red spruce-balsam fir-red pine stands in Nova Scotia. They evaluated 20-m wide SMZs with partial harvests, 20-m SMZs with no partial harvests, and 30-m SMZs. Two years after harvest, they concluded that the 30-m SMZ

Table 1. SMZ treatment effects on mean levels of NO_3^-, NH_4^+, and ortho-P in surface runoff, near-surface (O and A horizon), subsurface, and Δstream water grab samples (downstream–upstream) for 1 year after fertilizer application in the Virginia Piedmont.

Treatment	Surface runoff sample	Near-surface samples	Subsurface samples	ΔStream water samples
	(mg L^{-1} NO_3^-)	(mg m^{-2} per day NO_3^-)	(mg L^{-1} NO_3^-)	(mg L^{-1} NO_3^-)
	P-value = 0.0449	*P*-value = 0.001	*P*-value = 0.0673	*P*-value = 0.008
7.6-m SMZ	0.20 a *(0.05)*	4.43 a *(0.82)*	0.12ab *(0.06)*	0.01 a *(0.005)*
15.2-m SMZ thin	0.06 ab *(0.06)*	1.44 b *(0.62)*	0.38 a *(0.11)*	0.006 b *(0.002)*
15.2-m SMZ	0.02 ab *(0.02)*	0.73 b *(0.81)*	0.29 a *(0.07)*	0.005 b *(0.003)*
30.5-m SMZ	0.03 b *(0.04)*	0.68 b *(0.68)*	0.09 b *(0.08)*	0.004 b *(0.002)*
	(mg L^{-1} NH_4^+)	(mg m^{-2} per day NH_4^+)	(mg L^{-1} NH_4^+)	(mg L^{-1} NH_4^+)
	P-value = 0.1432	*P*-value = 0.1143	*P*-value = 0.1845	*P*-value = 0.1858
7.6-m SMZ	4.19 a *(1.94)*	0.50 a *(0.09)*	0.17 a *(0.04)*	0.06 a *(0.05)*
15.2-m SMZ thin	4.53 a *(2.19)*	0.68 a *(0.07)*	0.14 a *(0.07)*	0.07 a *(0.06)*
15.2-m SMZ	2.89 a *(1.63)*	0.59 a *(0.09)*	0.20 a *(0.04)*	0.05 a *(0.05)*
30.5-m SMZ	7.53 a *(1.32)*	0.75 a *(0.06)*	0.06 b *(0.05)*	0.07 a *(0.04)*
	(mg L^{-1} ortho-P)	(mg m^{-2} per day ortho-P)	(mg L^{-1} ortho-P)	(mg L^{-1} ortho-P)
	P-value = 0.0565	NA	*P*-value = 0.2235	*P*-value < 0.0211
7.6-m SMZ	0.11 b *(0.18)*	NA	0.005 a *(0.001)*	0.01 a *(0.002)*
15.2-m SMZ thin	0.88 a *(0.25)*	NA	0.001 a *(0.002)*	0.002 b *(0.0003)*
15.2-m SMZ	0.11 b *(0.18)*	NA	0.005 a *(0.002)*	0.02 a *(0.002)*
30.5-m SMZ	0.39 b *(0.15)*	NA	0.002 a *(0.002)*	0.01 a *(0.002)*

Within a column, *P*-values are displaced at the top and numbers followed by different letters are statistically different at alpha level = 0.10. Standard errors, presented in parentheses and italics, are based on 20–36 sample values.

was more effective at N control than the two types of 20-m SMZs. They also found that the intact 20-m SMZ was more effective at N removals than the 20-m SMZ with partial harvests, but they also noted that increases in N were slight. Stand types, soil, terrain, and SMZ treatments are clearly different than the research presented in this article, but the trends are similar. In both Nova Scotia and Virginia, wider SMZs tend to be more effective with regard to NO_3^- removals and partial harvests in the SMZs favored nutrient movement to the stream.

The 30.5-m SMZ was adequate for NO_3^-, NH_4^+, and ortho-P removals from surface, near-surface, subsurface, and stream waters (Table 1) and the 15.2-m SMZ with no partial harvests had only one significant treatment difference, NO_3^- in subsurface samples. The 15.2-m thin was generally effective, with one major exception. The thinning treatment resulted in a 2–8× increase in ortho-P in surface samples. This was probably associated with the thinning sample that ran perpendicular to the stream and increased surface flows. These results are supported by other studies that have found SMZs ≥ 15.2 m to be adequate for removal of pollutants. Governo et al. (2004) evaluated water quality after clearcut harvests with three types of SMZs: clearcut to stream, 15-m SMZ with partial harvest, and 15-m SMZ with no partial harvest. Their nutrient analyses were based on limited samples, because of drought conditions, but the authors speculated that the 15-m SMZ, with or without harvest, appeared to be sufficient for minimizing NO_3^- and P movement to the stream. Pratt and Fox (2009) evaluated the effectiveness of 15.2-m-wide SMZs for minimizing movement of N and P to streams after applications of biosolids in the Virginia Piedmont. The biosolids were applied to an 18-year-old loblolly pine plantation at rates of 1,120 kg ha^{-1} for N and 629 kg ha^{-1} for P. After 1 year of monitoring, their results indicated that the 15.2-m buffer was adequate for nonpoint source pollution protection, although they did find one period of elevated stream P after overland flow. Pinho et al. (2008) evaluated the efficiency of 10-m-wide mixed hardwood riparian forests in the Georgia Piedmont. They found that intact buffers removed 51% of dissolved P, but removal of the litter layer resulted in lower P removals (35%). Their litter layer disturbances were not unlike the disturbances created by thinning corridors.

Migration of fertilizer along transects within the SMZs was evaluated. All SMZ widths exhibited significant differences in NO_3^- with regard to transect positions (Table 2). For surface waters, all SMZ widths exhibited higher NO_3^- levels at the SMZ edge (0.03–1.2 mg L^{-1}) and lower NO_3^- values (0.06–0.2 mg L^{-1}) at streamside. The mechanisms of removal may have been physical (e.g., nutrients trapped as sediment attachments), biological (e.g., plant uptake), or chemical (reduction–oxidation), but regardless of mechanism, NO_3^- levels in surface waters were lower near the stream for all SMZ widths. These data clearly show the benefits that SMZs of any width can have on water quality.

As expected, the near-surface and subsurface water samples indicated that the fertilized watershed had the higher near-surface (4.21–28.5 mg m^{-2} of NO_3; Table 2) and subsurface (0.89–9.75 mg L^{-1} of NO_3^-; Table 2) concentrations than were found within the SMZ sample locations. As with the surface waters, the NO_3^- diminished along transects within treatments as sample stations neared the stream. For example five sample positions within the 30.5-m SMZ indicated a fourfold decrease in near-surface NO_3^- and a 2.5-fold decrease in subsurface NO_3^- between the SMZ edge and the streamside.

As expected, the cation NH_4^+ movements were less striking than NO_3^- anions (Table 3). If the data collected from fertilization stations in the clearcuts are ignored, almost no significant differences were found for NH_4^+ (Table 3) for surface, near-surface, subsurface, or stream waters. As indicated by the SMZ data (Table 1), little NH_4^+ appears to be moving from the fertilized area.

Ortho-P levels were found to be significantly higher in the surface water samples for the 30.5-m SMZ and 15.2-m SMZ edge (Table 4). There is no obvious explanation for this particular response, but some type of preferential movement or contamination may have occurred, because this pattern was not found even with the narrower SMZs. The ortho-P data indicated few differences across transects within the SMZ treatments, indicating that the SMZs were sufficient for protection against ortho-P pollution.

These data indicate that 7.6-m SMZs were less effective for minimizing mobile NO_3^-. The 30.5-m SMZ was the most effective and

Table 2. Effects of transect distance within SMZ treatments on mean NO_3^- values for surface runoff, near-surface (O and A horizons), and subsurface water samples for 1 year after fertilizer application in the Virginia Piedmont.

Transect within treatment	*P*-value	Clearcut	30.5 m	22.9 m	15.2 m	7.6 m	Streamside
Surface water transects (mg L^{-1} NO_3^-)							
7.6-m SMZ	0.0971					1.2 a *(0.07)*	0.2 b *(0.12)*
15.2-m SMZ thin	0.1036				1.2a *(0.06)*		0.06 a *(0.04)*
15.2-m SMZ	0.0548				0.03 a *(0.005)*		0.02 b *(0.004)*
30.5-m SMZ	0.0238		0.05 a *(0.008)*				0.02 b *(0.007)*
Near-surface water transects (mg m^{-2} per day NO_3^-)							
7.6-m SMZ	0.0001	10.41 a *(1.56)*				0.76 b *(1.52)*	0.71 b *(1.53)*
15.2-m SMZ Thin	0.0001	14.8a *(1.98)*			2.4 b *(1.49)*		0.33 b *(1.96)*
15.2-m SMZ	0.0001	28.5 a *(3.25)*			8.25 b *(3.28)*		0.52 c *(1.32)*
30.5-m SMZ	0.0001	4.21 a *(0.52)*	1.18 b *(0.52)*	0.77 b *(0.51)*	0.87 b *(0.50)*	0.27 b *(0.52)*	0.29 b *(0.51)*
Subsurface water transects (mg L^{-1} NO_3^-)							
7.6-m SMZ	0.0006	0.89 a (0.16)				0.11b (0.15)	0.12b (0.12)
15.2-m SMZ Thin	0.0415	9.75 a (2.77)			2.96 ab (1.96)		0.38 b (2.06)
15.2-m SMZ	0.0015	2.55 a (0.48)			0.50 b (0.42)		0.29 b (0.44)
30.5-m SMZ	0.0001	1.83 a *(0.157)*	0.25 b *(0.052)*	0.14 b *(0.049)*	0.13 b *(0.050)*	0.14 b *(0.049)*	0.09 b *(0.038)*

Within a row numbers followed by different letters are statistically different at alpha level = 0.10. Each number presented is the average of 20–36 measurements. Standard errors are presented in italics and are based on 20–36 sample values.

Table 3. Effects of transect distance within SMZ treatments on mean NH_4^+ values by SMZ treatment for surface runoff, near-surface (O and A horizons), and subsurface water samples for 1 year after fertilizer application in the Virginia Piedmont.

Transect within treatment	*P*-value	Clearcut	30.5 m	22.9 m	15.2 m	7.6 m	Streamside
Surface water transects (mg L^{-1} NH_4^+)							
7.6-m SMZ	0.005					4.18 a *(1.10)*	0.30 b *(0.20)*
15.2-m SMZ Thin	0.0310				4.53 a *(1.40)*		1.41 b *(0.80)*
15.2-m SMZ	0.0069				5.06 a *(0.98)*		1.58 b *(0.76)*
30.5-m SMZ	0.0001		17.8 a *(2.79)*				1.37 b *(2.16)*
Near surface water transects (mg m^{-2} per day NH_4^+)							
7.6-m SMZ	0.0001	12.93 a *(1.67)*				0.52 b *(1.61)*	0.49 b *(1.64)*
15.2-m SMZ Thin	0.0001	14.21 a *(1.31)*			0.70 b *(0.99)*		0.63 b *(1.29)*
15.2-m SMZ	0.0001	18.70 a *(2.53)*			0.70 b *(2.55)*		0.48 b *(2.58)*
30.5-m SMZ	0.0001	5.61 a *(0.68)*	0.97 b *(0.68)*	0.75 b *(0.66)*	0.72 b *(0.68)*	0.59 b *(0.67)*	0.73 b *(0.67)*
Subsurface water transects (mg L^{-1} NH_4^+)							
7.6-m SMZ	0.6010	0.11 a (0.05)				0.14 a (0.04)	0.17 a (0.04)
15.2-m SMZ Thin	0.6639	0.19 a (0.075)			0.21 a (0.05)		0.13 a (0.06)
15.2-m SMZ	0.3904	0.13 a (0.04)			0.13 a (0.03)		0.20 a (0.04)
30.5-m SMZ	0.0331	0.29 a *(0.14)*	0.23 ab *(0.04)*	0.18 ab *(0.04)*	0.18 ab *(0.04)*	0.14 ab *(0.04)*	0.06 b *(0.03)*

Within a row numbers followed by different letters are statistically different at alpha level = 0.10. Standard errors are presented in italics and are based on 20–36 sample values.

Table 4. Effects of transect distance within SMZ treatments on mean ortho-P values by SMZ treatment for surface runoff and subsurface water sample for 1 year after fertilizer application in the Virginia Piedmont.

Transect within treatment	*P*-value	Clearcut	30.5 m	22.9 m	15.2 m	7.6 m	Streamside
Surface water transects (mg L^{-1} ortho-P)							
7.6-m SMZ	0.1181					0.11 (0.07)	0.02 (0.04)
15.2-m SMZ Thin	0.1045				0.88 (0.52)		0.60 (0.50)
15.2-m SMZ	0.0456				0.20 a (0.05)		0.06 b (0.04)
30.5-m SMZ	0.0001		1.01 a (0.20)				0.02 b (0.015)
Subsurface water transects (mg L^{-1} ortho-P)							
7.6-m SMZ	0.7142	0.004 (0.002)				0.003 (0.002)	0.005 (0.001)
15.2-m SMZ thin	0.0501	0.005 a (0.001)			0.001 b (0.001)		0.001 b (0.001)
15.2-m SMZ	0.1920	0.01 (0.004)			0.003 (0.003)		0.0006 (0.003)
30.5-m SMZ	0.5783	0.001 (0.004)	0.001 (0.001)	0.002 (0.001)	0.001 (0.001)	0.002 (0.001)	0.002 (0.001)

Within a row numbers followed by different letters are statistically different at alpha level = 0.10. Standard errors are presented in italics and are based on 20–36 sample values.

the two types of 15.2-m SMZs were intermediate in terms of effectiveness. However, it should be noted that all SMZ treatments resulted in very low changes to stream nutrients (0.004–0.01 mg L^{-1} of NO_3^-, 0.05–0.07 mg L^{-1} of NH_4^+, and 0.002–0.02 mg L^{-1} of ortho-P; Table 1). To evaluate stream degradation, the Virginia Department of Environmental Quality (2010) has developed potential critical values of 2.3 mg L^{-1} for NO_3^- and 2.6 mg L^{-1} for total N and 0.4 mg L^{-1} for total P. All treatment stream values after fertilization were well below the Virginia Department of Environmental Quality standards.

Transect data were used to evaluate the effect of the SMZ treatments on the relative levels of NO_3^- (Table 5), NH_4^+ (Table 6), and ortho-P (Table 7) within the fertilized stand, at the edge of the SMZ, and at streamside. Significant effects are reported in Tables 2–4. Relative to the fertilized stand, NO_3^- levels at streamside were 33–95% lower in surface waters, 93.2–98.2% lower in near-surface

Table 5. Nitrate changes (%) of in surface, near-surface, and subsurface water samples from fertilized stand to SMZ edge and fertilized stand to stream edge by SMZ treatment.

Sample location	SMZ treatment	Change between fertilized stand and SMZ edge (%)	Change between fertilized stand and streamside (%)
Surface water	7.6-m SMZ	NA	−83.3
	15.2-m thin	NA	−95.0
	15.2-m	NA	−33.3
	30.5-m	NA	−60.0
Near-surface water	7.6-m SMZ	−92.6	−93.2
	15.2-m thin	−83.8	−97.8
	15.2-m	−71.1	−98.2
	30.5-m	−72.0	−93.1
Subsurface water	7.6-m SMZ	−88.0	−86.5
	15.2-m Thin	−69.6	−96.1
	15.2-m	−80.3	−88.6
	30.5-m	−86.3	−95.1

Negative number represent decreases, positive numbers represent increases. Percent changes were only calculated for treatments found to be significantly different at alpha levels = 0.10.

Table 6. Ammonium changes (%) of in surface, near-surface, and subsurface water samples from fertilized stand to SMZ edge and fertilized stand to stream edge by SMZ treatment.

Sample location	SMZ treatment	Change between fertilized stand and SMZ edge (%)	Change between fertilized stand and streamside (%)
Surface water	7.6-m SMZ	NA	−92.8
	15.2-m thin	NA	−68.9
	15.2-m	NA	−68.7
	30.5-m	NA	−92.3
Near-surface water	7.6-m SMZ	−96.0	−96.2
	15.2-m thin	−95.1	−95.6
	15.2-m	−96.3	−97.4
	30.5-m	−82.7	−87.0
Subsurface water	7.6-m SMZ	NS	NS
	15.2-m thin	NS	NS
	15.2-m	NS	NS
	30.5-m	NS	NS

Negative number represent decreases, positive numbers represent increases. Percent changes were only calculated for treatments found to be significantly different at alpha levels = 0.10.

Table 7. Ortho-P changes (%) of in surface and subsurface water samples from fertilized stand to SMZ edge and fertilized stand to stream edge by SMZ treatment.

Sample location	SMZ treatment	Change between fertilized stand and SMZ edge (%)	Change between fertilized stand and streamside (%)
Surface water	7.6-m SMZ	NA	NS
	15.2-m thin	NA	NS
	15.2-m	NA	−70.0
	30.5-m	NA	−98.0
Subsurface water	7.6-m SMZ	NS	NS
	15.2-m thin	−80.0	−80.0
	15.2-m	NS	NS
	30.5-m	NS	NS

Negative number represent decreases, positive numbers represent increases. Percent changes were only calculated for treatments found to be significantly different at alpha levels = 0.10.

waters, and 86.5–95.1% lower in subsurface waters (Table 5). These values are higher than the total N removal levels of 60–80 found in the Allegheny Plateau (Kochenderfer and Hornbeck 1999) and Cumberland Plateau (Arthur et al. 1998) and are six times higher than those reported by Wynn et al. (2000) for the Coastal Plain. The higher NO_3^- removal efficiencies found in this study are suspected to be partially caused by the dry conditions, but Pratt and Fox (2009) also found ≥90% NO_3^- removal efficiencies from subsurface water in SMZs located adjacent to biosolid applications in the Virginia Piedmont. NH_4^+ removals between the fertilized stands and stream edge ranged from 68.7 to 92.8 in surface waters and 87 to 97.4 for near-surface waters (Table 6). Ortho-P values were only found to be significantly different because of locations for surface waters. For surface waters ortho-P removal efficiencies ranged from 70 to 98% (Table 7). The lower ortho-P reduction in the thinned stands may be caused by the thinning corridors that facilitated movement of surface waters to the stream, similar to the ephemeral drain bypasses detected by Daniels and Gilliam (1996).

Conclusions

The literature supports the use of 15.2-m SMZs with thinning for sediment control or nutrient control after harvesting. The data reported here support the use of 30.5-m SMZs and generally support the use of 15.2-m SMZs for minimizing fertilization losses. It also indicated that thinning corridors should be avoided within SMZs if corridors will concentrate surface waters. Overall, the 7.6-m SMZ generally allowed increased NO_3^- movement to streamside relative to the wider SMZs. However, all SMZs showed positive influences on nutrients along transects from fertilized stand to SMZ edge and streamside. Additionally, stream water concentrations were within those deemed nondegraded by Virginia stream standards. However, it should be noted that this study occurred within a relatively dry year and nutrient movement may have been greater if normal rainfall had occurred. The observed nutrient movements combined with the low rainfalls support the use of the 15.2-m or wider nonthinned SMZs for minimizing fertilizer movement in loblolly pine plantations of the Piedmont. As with many field studies, this research should be considered within the context of its weaknesses, which include short duration (1 year), watershed and stand variability, and dry weather conditions. Overall, no single SMZ width can be expected to be appropriate for all management regimes. SMZ width should be matched to the watershed management goals, land-management activities, landowner objectives, and water quality needs.

Literature Cited

ALABAMA FORESTRY COMMISSION. 2007. *Alabama's best management practices for forestry.* Alabama Forestry Commission, Montgomery, AL. 31 p.

ALLMENDINGER, N.E., J.E. PIZZUTO, N.P. POTTER, AND W.C. HESSION. 2005. The influence of riparian vegetation on stream width, eastern Pennsylvania, ISA. *Geol. Soc. Am. Bull.* 117(1/2):229–243.

ANDERSON, C.J., AND B.G. LOCKABY. 2011a. Research gaps related to forest management and stream sediment in the United States. *Environ. Manage.* 47:303–313.

ANDERSON, C.J., AND B.G. LOCKABY. 2011b. The effectiveness of forestry best management practices for sediment control in the southeastern United States: A literature review. *South. J. Appl. For.* 35(4):170–177.

ARTHUR, M.A., G.B. COLTHARP, AND D.L. BROWN. 1998. Effects of best management practices on forest streamwater quality in eastern Kentucky. *J. Am. Water Resour. Assoc.* 34:481–495.

AUST, W.M., AND C.R. BLINN. 2004. Forestry best management practices for timber harvesting and site-preparation in the eastern United States: An overview of water quality and water quality research during the past 20 years (1982–2002). *Water Air Soil Pollut. Focus* 4:5–36.

AUST, W.M., M.B. CARROLL, M.C. BOLDING, AND C.A. DOLLOFF. 2011. Operational forest stream crossings effects on water quality in the Virginia Piedmont. *South. J. Appl. For.* 35(3):123–130.

BARLING, R.D., AND I.D. MOORE. 1994. Role of buffer strips in management of waterway pollution: A review. *Environ. Manage.* 18(4):543–558.

BEASLEY, R.S., A.B. GRANILLO, AND V. ZILLMER. 1986. Sediment losses from forest management: Mechanical vs chemical site preparation after clearcutting. *J. Environ. Qual.* 15(4):423–426.

BINKLEY, D., AND T.C. BROWN. 1993. Forest practices as nonpoint sources of pollution in North America. *Water Resour. Bull.* 29:729–739.

BINKLEY, D., H. BURNHAM, AND H.L. ALLEN. 1999. Water quality impacts of forest fertilization with nitrogen and phosphorous. *For. Ecol. Manage.* 121:191–213.

BRIAN, M., C. HICKEY, AND B. DORAN. 2004. A review of the efficiency of buffer strips for the maintenance and enhancement of riparian ecosystems. *Water Qual. Res. J. Can.* 39(3):311–317.

BRITISH COLUMBIA MINISTRY OF FORESTRY. 1995. *Analyzing ANOVA designs.* Biometrics Handbk. 5, British Columbia Ministry of Forest, Victoria, Canada. 61 p.

BROADMEADOW, S., AND T.R. NISBET. 2004. The effects of riparian forest management on the freshwater environment: A literature review of best management practices. *Hydrol. Earth Sys. Sci.* 8(3):286–305.

CASTELLE, A., A. JOHNSON, AND C. CONOLLY. 1994. Wetland and stream buffer size requirements—A review. *J. Environ. Qual.* 23:878–882.

CLINTON, B. 2011. Stream water responses to timber harvests: riparian buffer width effectiveness. *For. Ecol. Manage.* 261:979–988.

COOPERBAND, L.R., AND T.J. LOGAN. 1994. Measuring in situ changes in labile soil phosphorous with anion-exchange membranes. *Soil Sci. Soc. Am. J.* 58:105–114.

CORRELL, D.L. 2005. Principles of planning and establishment of buffer zones. *Ecol. Eng.* 24:433–439.

COWELL, C.M. 1998. Historical change in vegetation and disturbance on the Georgia Piedmont. *Am. Midl. Nat.* 140:79–89.

DANIELS, R.B., AND J.W. GILLIAM. 1996. Sediment and chemical load reduction by grass and riparian buffers. *Soil Sci. Soc. Am. J.* 60:246–251.

DANIELSON, T. 2004. *Protocols for collecting water grab samples in rivers, streams, and freshwater wetlands.* Document DEPLW0637, State of Maine Department of Environmental Protection, Augusta, ME. 5 p.

DOSSKEY, M.G., P. VIDON, N.P. GURWICK, C.J. ALLAN, T.P. DUVAL, AND R. LOWRANCE. 2010. The role of riparian vegetation in protecting and improving chemical water quality in streams. *J. Am. Water Resour. Assoc.* 46(2):261–277.

EDWARDS, P.J., AND K.W.J. WILLARD. 2010. Efficiencies of forestry best management practices for reducing sediment and nutrient losses in the eastern United States. *J. For.* 108(5):245–249.

EASTERBROOK-WALKER, A.C., W.M. AUST, C.A. DOLLOFF, AND P.D. KEYSER. 2003. *Natural erosion rates for riparian buffers in the Piedmont of Virginia.* Virginia Water Res. Symp. Oct. 7–10, 2003. Virginia Water Resources Research Center, Virginia Tech, Blacksburg, VA. 206 p.

FENNEMAN, N.M. 1938. *Physiography of Eastern United States.* McGraw-Hill Book Co., Inc., New York and London. 714 p.

FOX, T.R., H.L. ALLEN, T.J. ALBAUGH, R. RUBILAR, AND C.A. CARLSON. 2007a. Tree nutrition and forest fertilization of pine plantations in the southeastern United States. *South. J. Appl. For.* 31(1):5–11.

FOX, T.R., H.L. ALLEN, T.J. ALBAUGH, R. RUBILAR, AND C.A. CARLSON. 2007b. Forest fertilization and water quality in the United States. *Better Crops* 91:7–9.

GEORGIA FORESTRY COMMISSION. 2009. *Georgia's best management practices for forestry.* Georgia Forestry Commission, Macon, GA. 72 p.

GOMEZ, K.A., AND A.A. GOMEZ. 1984. *Statistical procedures for agricultural research.* Wiley and Sons, New York. 680 p.

GOVERNO, R., B.G. LOCKABY, B. RUMMER, AND C. COLSON. 2004. Silvicultural management within streamside management zones of intermittent streams: Effects of decomposition, productivity, nutrient cycling, and channel vegetation. *South. J. Appl. For.* 28(4):211–224.

HEWLETT, J.D., AND J.C. FORTSON. 1982. Stream temperature under and inadequate buffer strip in the southeastern Piedmont. *Water Resour. Bull.* 18(6):983–988.

HUTCHENS, J.J. JR., D.P. BATZER, AND E. REESE. 2004. Bioassessment of silvicultural impacts in streams and wetlands of the eastern United States. *Water Air Soil Pollut. Focus* 4:37–53.

ICE, G.G. 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. *J. Environ. Eng.* 130(6):684–689.

ICE, G.G., G.W. STUART, J.B. WAIDE, L.C. IRLAND, AND P.V. ELLEFSON. 1997. Twenty-five years of the Clean Water Act: How clean are forest practices. *J. For.* 95:9–13.

KEIM, R.F., AND S.H. SCHOENHOLTZ. 1999. Functions and effectiveness of silvicultural streamside management zones in loessial bluff forests. *For. Ecol. Manage.* 118:197–209.

KOCHENDERFER, J.N., AND J.W. HORNBECK. 1999. Contrasting timber harvesting operations illustrate the value of BMPs. P. 128–136 in *Proc. of 12th Central hardwood forest conf.*, Stringer, J.W., and D.L. Loftis (eds.). USDA For. Serv., Gen. Tech. Rep. SRS-24.

LAKEL, W.A., W.M. AUST, AND C.A. DOLLOFF. 2006. Seeing the trees along the streamside. *J. Soil Water Conserv.* 61:22–29.

LAKEL, W.A., W.M. AUST, M.C. BOLDING, C.A. DOLLOFF, P. KEYSER, AND R. FELDT. 2010. Sediment trapping by streamside management zones of various widths following harvest and site preparation. *For. Sci.* 56(6):541–551.

LIECHTY, H.O., J. NETTLES, D.A. MARION, AND D.J. YURTON. 1999. Stream chemistry after an operational fertilizer application in the Ouachita Mountains. P 265–270 in *Proc. of 10th Biennial southern silvicultural research conf.*, Shreveport, LA, Feb. 16–18, 1999, Haywood, J.D., (ed.). USDA For. Serv., Gen. Tech. Rep. SRS 30. 618 p.

LOWRANCE, R. 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. *J. Environ. Qual.* 21:401–405.

LOWRANCE, R., L.S. ALTIER, J.D. NEWBOLD, R.S. SCHNABEL, P.M. GROFFMAN, J.M. DENVER, D.L. CORRELL, ET AL. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. *Environ. Manage.* 21(5):687–712.

LOWRANCE, R., AND J.M. SHERIDAN. 2005. Surface runoff water quality in a managed three zone riparian buffer. *J. Environ. Qual.* 34:1851–1859.

LYNCH, J.A., E.S. CORBETT, AND K. MUSSALLEM. 1985. Best management practices for controlling nonpoint-source pollution on forested watersheds. *J. Soil Water Conserv.* 40(1):164–167.

MAYER, P.M., S.K. REYNOLDS, M.D. MCCUTCHEN, AND T.J. CANFIELD. 2007. Meta-analysis of nitrogen removal in riparian buffers. *J. Environ. Qual.* 36:1172–1180.

MCBROOM, M.W., R.S. BEASLEY, AND M. CHANG. 2008. Water quality effects of clearcut harvesting and forest fertilization with best management practices. *J. Environ. Qual.* 37:114–124.

NEARY, D.G., W.T. SWANK, AND H. RIEKERK. 1989. An overview of nonpoint source pollution in the southern United States. P. 1–7 in *Proc. of Symp. on The forests wetlands of the southern United States*, Orlando, FL, July 12–14, 1988, Hook, D.D., and R. Lea (eds.). USDA For. Serv., Gen. Tech. Rep. SE-50, South. For. Exp. Stn., Asheville, NC. 168 p.

NORTH CAROLINA DIVISION OF FOREST RESOURCES. 2006. *North Carolina forestry best management practices manual to protect water quality.* North Carolina Division of Forest Resources, Raleigh, NC. 243 p.

Pinho, A.P., L.A. Morris, C.R. Jackson, W.J. White, P.B. Bush, and A.T. Matos. 2008. Contaminant retention potential of forested filter strips established as SMZs in the Piedmont of Georgia. *J. Am. Water Resour. Assoc*. 44(6):1564–1577.

Pratt, W.A., and T.R. Fox. 2009. Streamside management zones effectiveness for protecting water quality after forestland application of biosolids. *J. Environ. Qual.* 38:2106–2120.

Rivenbark, B.L., and C.R. Jackson. 2004. Concentrated flow breakthroughs moving through silvicultural streamside management zones: Southeastern piedmont, USA. *J. Am. Water Resour. Assoc.* 40(4):1043–1052.

SAS Institute. 2010. *JMP 9 Design of experiments guide.* SAS Institute, Inc., Cary, NC. 325 p.

Shepard, J.P. 2006. Water quality protection in bioenergy production: The US system of forestry best management practices. *Biomass Bioenergy* 30:376–384.

Soil Moisture Equipment Corp. 1999. *Operating instructions: 1900 Soil water samplers.* Available online at www.soilmoisture.com; last accessed Jan. 1, 1999. 16 p.

South Carolina Forestry Commission. 1994. *South Carolina's best management practices for forestry*. South Carolina Forestry Commission, Columbia, SC. 64 p.

Steel, R.D., and J.H. Torrie. 1980. *Principles and procedures of statistics—A biometrical approach.* McGraw Hill Book Co., New York. 633 p.

Subler, S., J.M. Blair, and C.A. Edwards. 1995. Using anion-exchange membranes to measure soil nitrate availability and net nitrification. *Soil Biol. Biochem.* 27:911–917.

Swift, L.W., and J.B. Messer. 1971. Forest cuttings raise temperatures of small streams in the southern Appalachians. *J. Soil Water Conserv.* 26(3):111–116.

Trimble, S.W. 1974. *Man-induced soil erosion on the southern piedmont, 1700–1970.* Soil Conservation Society of America, Ankeny, IA. 180 p.

US Department of Education. 1981. *Collecting stream samples for water quality. Module 16, Vocational education training in environmental health sciences*. US Department of Education, Washington, DC. 42 p.

Vaidya, O.C., T.P. Smith, H. Fernand, and N.R. McInnis Leek. 2008. Forestry best management practices: Evaluation of alternative streamside management zones on water quality in Pockwock Lake and Five Mile Lake Watersheds in central Nova Scotia, *Can. Environ. Monitor Assess.* 137:1–14.

Virginia Department of Environmental Quality. 2010. *Nutrient criteria development plan for the commonwealth of Virginia.* Virginia Department of Environmental Quality, Water Division, Richmond, VA. 189 p.

Virginia Department of Forestry. 2002. *Virginia's forestry best management practices for water quality*, 4th ed. Virginia Department of Forestry, Charlottesville, VA. 216 p.

Vitousek, P.M., J. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and G.D. Tilman. 1997. Human alterations of the global nitrogen cycle: Causes and consequences. *Issues Ecol*. 1:1–15.

Ward, J.M., and C.R. Jackson. 2004. Sediment trapping within forestry streamside management zones: Georgia Piedmont, USA. *J. Am. Water Resour. Assoc*. 40(6):1421–1431.

Wynn, T.M., S. Mostaghimi, J.W. Frazee, P.W. McClellan, R.M. Shaffer, and W.M. Aust. 2000. Effects of forest harvesting best management practices on surface water quality in the Virginia coastal plain. *Trans. Am. Soc. Agric. Eng*. 43(4):927–936.