JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION

Vol. 46, No. 2 AMERICAN WATER RESOURCES ASSOCIATION April 2010

WATER QUALITY FUNCTIONS OF A 15-YEAR-OLD RIPARIAN FOREST BUFFER SYSTEM[1]

J. Denis Newbold, Susan Herbert, Bernard W. Sweeney, Paul Kiry, and Stephen J. Alberts[2]

ABSTRACT: We monitored long-term water quality responses to the implementation of a three-zone Riparian Forest Buffer System (RFBS) in southeastern Pennsylvania. The RFBS, established in 1992 in a 15-ha agricultural (row crop) watershed, consists of: Zone 1, a streamside strip (~10 m wide) of permanent woody vegetation for stream habitat protection; Zone 2, an 18- to 20-m-wide strip reforested in hardwoods upslope from Zone 2; and Zone 3, a 6- to 10-m-wide grass filter strip in which a level lip spreader was constructed. The monitoring design used paired watersheds supplemented by mass balance estimates of nutrient and sediment removal within the treated watershed. Tree growth was initially delayed by drought and deer damage, but increased after more aggressive deer protection (1.5 m polypropylene shelters or wire mesh protectors) was instituted. Basal tree area increased ~20-fold between 1998 and 2006, and canopy cover reached 59% in 2006. For streamwater nitrate, the paired watershed comparison was complicated by variations in both the reference stream concentrations and in upslope groundwater nitrate concentrations, but did show that streamwater nitrate concentrations in the RFBS watershed declined relative to the reference stream from 2002 through the end of the study in early 2007. A subsurface nitrate budget yielded an average nitrate removal by the RFBS of 90 kg/ha/year, or 26% of upslope subsurface inputs, for the years 1997 through 2006. There was no evidence from the paired watershed comparison that the RFBS affected streamwater phosphorus concentration. However, groundwater phosphorus did decline within the buffer. Overland flow sampling of 23 storms between 1997 and 2006 showed that total suspended solids concentration in water exiting the RFBS to the stream was on average 43% lower than in water entering the RFBS from the tilled field. Particulate phosphorus concentration was lower by 22%, but this removal was balanced by a 26% increase in soluble reactive phosphorus so that there was no net effect on total phosphorus.

(KEY TERMS: nonpoint source pollution; best management practices; nutrients; riparian forest buffer.)

Newbold, J. Denis, Susan Herbert, Bernard W. Sweeney, Paul Kiry, and Stephen J. Alberts, 2010. Water Quality Functions of a 15-Year-Old Riparian Forest Buffer System. *Journal of the American Water Resources Association* (JAWRA) 46(2):299-310. DOI: 10.1111/j.1752-1688.2010.00421.x

[1]Paper No. JAWRA-09-0039-P of the *Journal of the American Water Resources Association* (JAWRA). Received February 20, 2009; accepted September 21, 2009. **Discussions are open until six months from print publication**.

[2]Respectively, Research Scientist, Research Technician, and Director, Stroud Water Research Center, 970 Spencer Road, Avondale, Pennsylvania 19311; Staff Scientist, Patrick Center for Environmental Research at the Academy of Natural Sciences, 1900 Benjamin Franklin Parkway, Philadelphia, Pennsylvania 19103; Former Graduate Student, Department of Health, West Chester University, West Chester, Pennsylvania 19383, Currently Environmental Scientist, MWH Americas, Inc., Malvern, Pennsylvania 19355 (E-Mail/Newbold: newbold@stroudcenter.org).

INTRODUCTION

Riparian forest buffers have become well established as a management practice that can reduce the surface and subsurface transport of agrichemicals to streams when used as a component of an integrated farm management system (Dwire and Lowrance, 2006). Nonetheless, it remains difficult to quantify the nutrient and sediment load reductions that can be expected from riparian afforestation. This difficulty reflects, in part, a discord between the very high nutrient and sediment removal rates that many studies have demonstrated (e.g., Lowrance *et al.*, 1997; Mayer *et al.*, 2007) and the cautions that these potentials are not always achieved (e.g., Puckett, 2004; Vidon and Hill, 2004; Knight *et al.*, 2008). While such cautions do not lessen the advisability of riparian reforestation to enhance stream habitat and stream ecosystem services (Sweeney *et al.*, 2004; Jones *et al.*, 2006; Sweeney and Blaine, 2007), they do point out the need for better estimates of buffer function. Among the large number of studies that have been conducted, examinations of the temporal response to riparian afforestation of agricultural land, particularly at the whole watershed level, are rare. Such studies are needed not only to quantify the time required to achieve buffer function but also to control for the potential bias of comparing existing mature forest buffers with existing nonbuffered agricultural riparian zones, as it is often lands less suitable for tillage that are left in forest.

This study used a paired watershed approach and mass balance analysis to quantify nutrient and sediment removal by a three-zone riparian forest buffer system (RFBS) (Welsch, 1991) established in 1992 on an agricultural headwater stream in the Pennsylvania Piedmont.

MATERIALS AND METHODS

Study Site

The study was conducted on three small watersheds located in the Piedmont province of southeastern Pennsylvania (Figure 1) in the Brandywine River drainage. Field slopes range from 5 to 10%. Soils are mainly typic hapludults, but those in the riparian areas are aquic fragiudults with seasonally high water tables reaching to within 1.5-0.5 m of the surface. A weathered rock or saprolite extends to a typical depth of 5-7 m with bedrock consisting mainly of fractured schist.

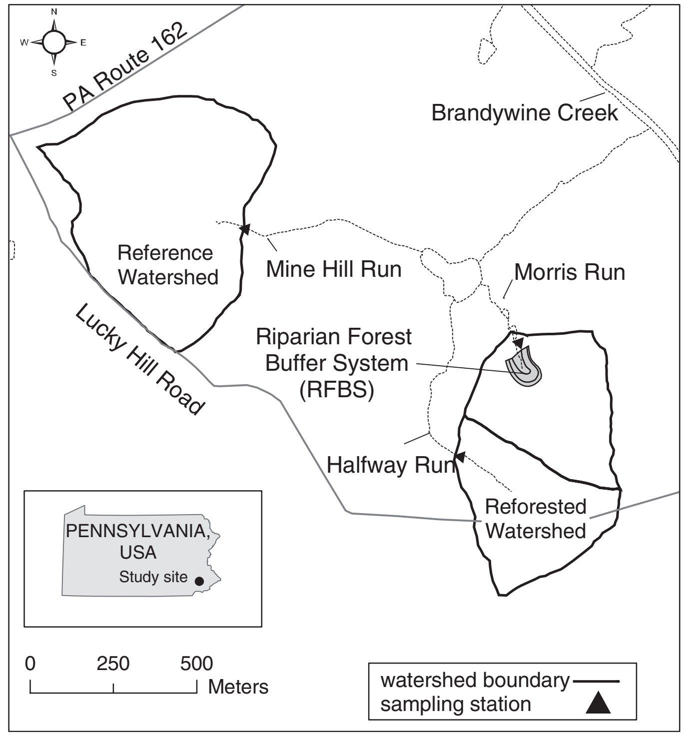

FIGURE 1. Location Map for Project Watersheds on the Stroud Preserve. The Morris Run sampling station is located at 39°56′46″N, 75°39′12″W.

The RFBS was established in one watershed, while a second watershed remained unaltered (hereafter "control"). In the third watershed, all of the tilled land was reforested. The reforested watershed provided perspective regarding the maximum rate and extent of water quality improvement that could be locally achieved. The 14.9-ha RFBS watershed is drained by Morris Run, a perennial first-order stream. All but a few hectares of the RFBS watershed were maintained in crop strips (primarily corn, soybeans, and hay) which were laid out on the contour and rotated periodically. Fertilizer nitrogen applications to the RFBS watershed averaged 54 kg/ha of watershed area per year between 1991 and 2006. Annual applications varied with the cropping, at the discretion of the farmer, and generally declined throughout the study from 75 kg/ha/year in 1991 to 42 kg/ha/year in 2006 (Figure 2). In April 1992, an RFBS surrounding the headwaters of Morris Run was established in accordance with the specification published by the USDA Forest Service (Welsch, 1991). The RFBS (Figure 3) consists of: Zone 1, a ~10-m-wide streamside strip of permanent woody vegetation for stream habitat protection; Zone 2, an 18- to 20-m-wide strip, upslope of Zone 1, reforested in hardwoods; and Zone 3, a 6- to 10-m-wide grass filter strip with a level lip spreader between Zone 2 and

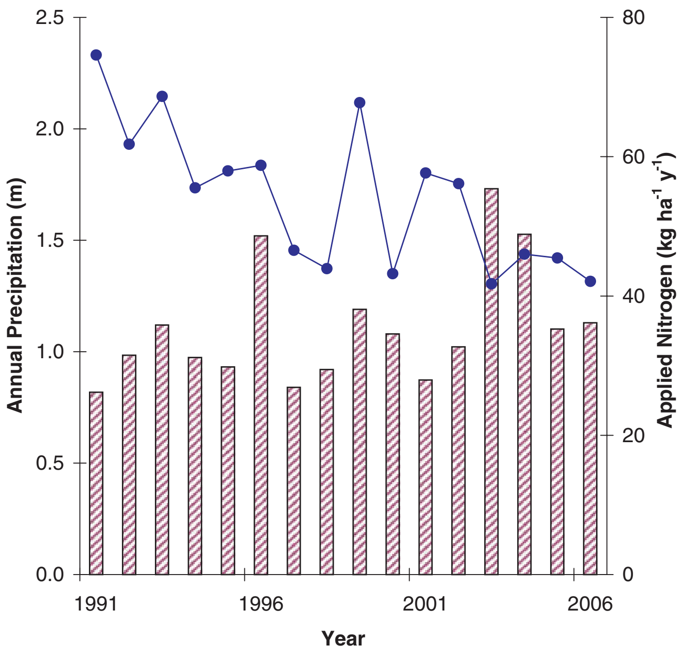

FIGURE 2. Annual Precipitation at the Stroud Preserve and Annual Fertilizer Nitrogen Applications to the Morris Run (RFBS) Watershed Per Hectare of Watershed Area.

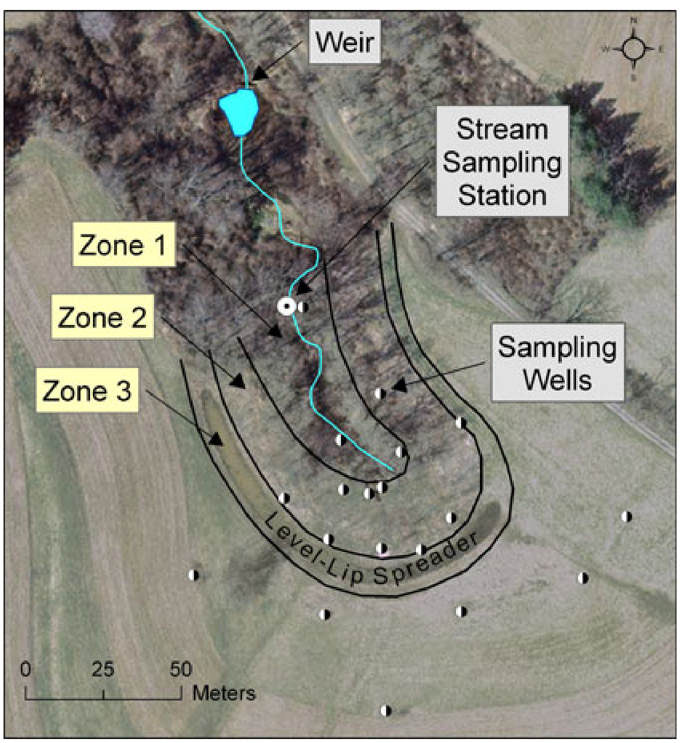

FIGURE 3. Morris Run and the Riparian Forest Buffer System With Level-Lip Spreader in April 2005.

the cultivated field. The reforestation of Zone 2, initiated in 1992, consisted of a mix of sugar maple, red oak, tulip poplar, white ash, black walnut, and trembling aspen planted as one year seedlings at approximately 3-m spacing. Seedlings were protected initially by 1.2-m-tall polypropylene tree shelters (Tubex®; Aberaman Park, Aberaman, South Wales, United Kingdom) which were eventually replaced by either 1.5-m-tall shelters or wire mesh. Prior to 1992, Zone 1 contained some hardwood trees up to several decades in age while Zones 2 and 3 had been maintained in hay, with some tilled area. In accordance with the USDA Forest Service specification, the grassland zone (Zone 3) was contoured in May 1994 to form a level lip spreader, designed by the USDA Natural Resources Conservation Service (NRCS). The purpose of the spreader is to intercept surface runoff, which is delivered to the buffer via grassed waterways, and to release the runoff to the forested buffer as dispersed sheet flow in order to minimize concentrated flow and erosion within the forested portion of the buffer (Welsch, 1991). The spreader was constructed by establishing a 3-m-wide grassed area (the "level lip") running 130 m along the original field contour, with minimal re-grading. A swale was excavated along the upslope side of the level lip spreader (Figure 3).

The control watershed is 34.4 ha in area, and is drained by Mine Hill Run, a perennial first-order stream. Most of the watershed was planted in hay, corn, and soybeans, also under NRCS conservation tillage. A sparsely forested, brushy zone extended 50-200 m from the stream. Land use in this watershed was maintained without alteration during the study.

The reforested watershed (14.5 ha) is drained by Half Way Run, which was surrounded by a mature forest extending at least 30 m from the stream. In the spring of 1991, all of the area within the Half Way Run watershed that had been in crop production (26% of the watershed area) was planted with mixed hardwood seedlings. Twenty-four percent of the watershed, occupying its highest elevations, remained unforested, primarily in pasture.

Monitoring Installations

Nineteen groundwater sampling wells (5-8 m deep, screened in the lower 0.5-3 m) were installed in the Morris Run (RFBS) watershed along transects extending radially upslope from the stream. The depth of the wells was established by auger refusal at the interface of saprolite underlain by fractured crystalline bedrock. Seven wells were located at or near the interface of Zones 1 and 2, six at the Zone 2 to Zone 3 interface, and six in the cultivated field. The wells in the field were placed 10-35 m upslope from Zone 3 (Figure 3).

At the RFBS site, 10 overland flow collectors were positioned at the upslope boundary of the reforested buffer zone (Zone 2), and 10 more were positioned downslope from the reforested zone, near the stream. The collector design was a modification (Alberts, 2000) of the Low Impact Flow Event sampler described by Sheridan *et al.* (1996). Each collector consisted of two plastic barriers (6 cm high × 50 cm long) oriented in the direction of the slope and implanted in the soil at a spacing of 15-25 cm so as to funnel surface flow onto a collector plate then, via tubing, to a below-grade 18 l bucket. In large storms, the bucket could fill to capacity, in which case a bypass port diverted excess water that entered the collector. Overland flow entered Zone 3 via two grassed waterways, each with two collectors. Overland flow entered Zone 2 only after filling the swale in the grass buffer that bordered the level lip spreader. Once the swale filled, water flowed over the level lip spreader into Zone 2.

Streamflow from each of the three watersheds was gauged by 90° V-notch weirs installed in 1993 (Morris Run) and 1997 (Mine Hill Run and Half Way Run). Streamflow was calculated (Grant, 1989) from the water level in the stilling pond of each respective weir, which was monitored by either float-wheel or pressure transducer and logged every 15 min. The flow record was separated into storm flow and base flow using the method proposed by Hewlett and Hibbert (1967) as further described by Lesack (1993).

Water Quality Monitoring

The water quality monitoring program was based on a paired watershed design. The riparian forest buffer was established in the first year of monitoring (1992). The next several years, while the seedlings became established and basal area and canopy cover remained negligible, served as a calibration period. Stream water was sampled manually from each stream, upstream of the weir, at one to three week intervals from 1992 to 1997 and at two week intervals from 1997 through March 2007. Samples were analyzed for ammonia, nitrate, soluble reactive phosphorus (SRP), and total phosphorus (TP). As a supplement to the paired watershed design, additional sampling was conducted (only in the RFBS watershed) to estimate nutrient and sediment retention within the riparian buffer by mass balance. This was accomplished through quarterly sampling of the groundwater monitoring wells between 1992 and 2007 (except 1996), and through sampling of storm-generated overland flow from the overland flow collectors, from 1997 through 2001 and again from 2005 to early 2007. All samples were analyzed for nitrate, ammonia, and SRP. In addition, overland flow samples were analyzed for TP and total suspended solids (TSS). Overland flow samples were collected within 24 h of a storm. Samples from a given event were assayed for nutrients and TSS if there were analyzable samples in at least two collectors (out of 10) both upslope of the reforestation ("Above Zone 2") and downslope of the reforestation ("Below Zone 2"). The collectors in the waterways ("Above Zone 3") normally filled even in small storms.

Nitrate (including nitrite) was determined after membrane (0.24 μm) filtration by cadmium reduction (USEPA method 353.2) (USEPA, 1993). Ammonia-N was determined by the colorimetric automated phenate method (USEPA method 350.1) (USEPA, 1993). SRP was determined on filtered (0.45 μm pore size, membrane) samples by the ascorbic acid method (USEPA method 365.1) (USEPA, 1993). TP was determined on unfiltered samples by the ascorbic acid method after digestion by ammonium persulfate (USEPA method 365.1) (USEPA, 1993). Total dissolved phosphorus (TDP) was determined as TP in membrane-filtered samples. TDP averaged 16% higher than SRP and was closely correlated (r^2 = 0.99) with SRP. Therefore, the TDP assay was suspended in 2001 and only SRP is reported. Particulate phosphorus was calculated as the difference between TP and SRP. TSS was determined by filtering an aliquot (100-3,200 ml, as filter capacity permitted) of sample onto a preweighed 47 mm Whatman GFF glass-fiber filter (0.7 μm nominal pore size), drying at 105°C for 24 h and reweighing the filter (American Public Health Association, American Water Works Association and Water Environment Federation, 1992). We assayed samples for TSS concentration rather than as suspended-sediment concentration from whole samples (Gray *et al.*, 2000). To minimize bias from settling of particles, all samples, both from overland flow collectors and streams, were inverted and vigorously shaken during or immediately prior to sub-sampling for TSS.

Tree Growth Monitoring

To monitor forest growth in Zone 2 of the RFBS, the diameter of each tree was measured at breast height (DBH) once or twice annually from 1998 through 2006. Basal area was calculated from DBH. Canopy cover in the RFBS Zone 2 was estimated annually in late summer from 2002 through 2006. Each of 358 points of a 3 × 3-m grid encompassing Zone 2 of the RFBS was scored as either lying directly below tree canopy or below open sky.

Data Analysis

Analysis of variance (ANOVA) in conjunction with Tukey's multiple comparison test was used to analyze both year-to-year variation in nitrate and phosphorus concentrations in streamwater and groundwater and within-year spatial variation along the field-to-stream well transects. For the paired watershed comparisons, the differences between paired samples (RFBS less control) were analyzed by one-way ANOVA, using year as the main effect. Temporal trends in the paired differences were analyzed by Tukey's multiple comparison test. Our use of ANOVA on paired differences is derived from the general analysis-of-covariance (ANCOVA) approach to paired watersheds as presented by USEPA (1993) and applied, for example, by Clausen *et al.* (2000) and Meals (2001). The ANCOVA can detect a change (pre-to-post intervention) in the relationship between the two watersheds (i.e., in the slope of a regression of concentrations from one watershed on those of the other watershed), or a change in a constant difference between the two watersheds (a pre-to-post change in the regression intercept). If the regression slope has a value of 1 and remains constant, so that the pre-to-post change involves only the intercept, then the ANCOVA result is mathematically equivalent to one-way ANOVA of the paired differences. For this study, the regression slope for streamwater nitrate concentrations was 1.00 ± 0.06 (standard error) over the entire record (1992-2007), so the use of paired differences in the ANOVA appeared justified. The simpler ANOVA approach allowed multiple comparisons among individual years, an advantage because the transition from calibration to post-intervention depended on tree growth and was expected to be gradual.

Sediment and nutrients transported in overland flow were analyzed by first log-transforming the analyte concentration from each collector for each storm date and then computing the mean transformed concentration for each of the three collector positions (Above Zone 2, Below Zone 2, and Above Zone 3) on each date. The effect of collector position on concentration was tested by a single two-way (date $\times$ position) ANOVA with one observation per cell, followed by a Tukey's test. Data were back-transformed to geometric means for tabular reporting. All effects were tested at the $p < 0.05$ significance level.

Mass-Balance Estimate of Nitrogen Removal

A mass-balance estimate of subsurface nitrate removal by the RFBS was computed for the 10-year period 1997 through 2006 based on the difference between inputs to the buffer from upslope and baseflow exports from the watershed measured at the sampling station. This approach was possible because the RFBS entirely surrounded the stream above the sampling station and all exports necessarily passed through the buffer. Stream exports integrate deep as well as shallow flow pathways, whereas the use of streamside wells alone may not reflect the entire flux of water to the stream (Böhlke and Denver, 1995; Puckett *et al.*, 2002). We are confident that the stream exports captured nearly all of the groundwater flow both because the piezometric surface conformed reasonably well with surface topography and because annual water yields agreed well with regional watershed water balances of similar geology (Vogel and Reif, 1993). Annual subsurface export from the buffer was calculated as the product of the annual mean of base flow and of nitrate concentration in samples taken at or within 20% of base flow. Subsurface input to the RFBS from upslope was calculated as the product of groundwater flow into the buffer and average nitrate concentration in wells upslope from the RFBS. An additional input to subsurface water within the buffer was estimated as the product of nitrate concentration in soil lysimeters at 1 m depth (0.88 mg/l as N) and the groundwater recharge within the buffer. To estimate inputs of upslope water and groundwater recharge within the buffer, we assumed that groundwater recharge was equal to base flow and uniform throughout the basin. The RFBS occupied 5.1% (0.72 ha) of the watershed area above the sampling station (14.0 ha). Thus groundwater recharge to the RFBS was estimated as 5.1% of the base flow, and subsurface input from upslope was estimated as 94.9% of the base flow. The annual base flows used for these calculations were adjusted to 94% of that measured at the weir because 6% of the 14.9 ha watershed drained to the stream between the sampling station and the weir. The mean annual removal of nitrate from subsurface flow was then estimated as the difference between the mean annual inputs from upslope groundwater and recharge within the buffer, and mean annual outputs via baseflow export.

RESULTS AND DISCUSSION

Precipitation and Streamflow

Annual precipitation measured at the site averaged 1.11 m/year, ranging from 0.84 to 1.73 m/year (Figure 2). Annual streamflows, normalized for watershed area, averaged 0.37, 0.34, and 0.22 m/year for the RFBS, control, and reforested watersheds, respectively. The respective average base flows were 0.28, 0.31, and 0.15 m/year.

Tree Growth

Tree growth in Zone 2 of the RFBS was slow from 1992 to 1998, with significant annual mortality from drought and deer damage. Although much of the initial planting stock was replaced during these years, mortality was eventually reduced through annual application of herbicide (glyphosate) around each tree and the use of taller tree shelters (both polypropylene and wire mesh) as the trees matured. After 1999, rapid tree growth was evident and basal area increased 20-fold between 1999 and 2006 (Figure 4). Canopy cover reached 41% in 2002 and 59% in 2006 (data not shown).

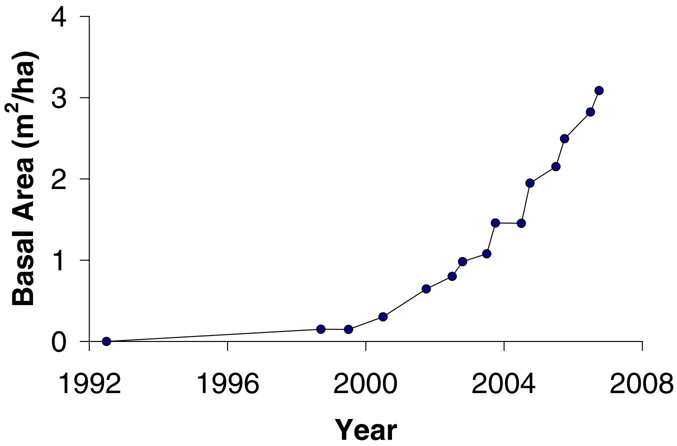

FIGURE 5. Changes in Basal Area of Trees in Zone 2 of the Riparian Forest Buffer Over the Project Period.

Stream and Groundwater Nitrate

In the stream draining the reforested watershed (Half Way Run), mean annual nitrate-N concentration decreased by 44% from 2.7 mg/l in 1992 to 1.5 mg/l in 1999 and remained near this level (averaging 1.6 mg/l) into the first months of 2007 (Figure 5). Because agricultural nitrogen application ceased when the watershed was reforested in 1991, the decline in nitrate between 1992 and 1999 appears to represent the flushing of the preexisting pool of groundwater nitrate from the watershed. During this period, the nitrate concentration declined at an exponential rate of 0.30/year toward an asymptotic concentration of 1.55 mg/l as described by the equation $C(t) - 1.55 = 1.8\exp(-0.30t)$, where $C(t)$ is the average nitrate concentration in year t after 1999 (nonlinear regression, SAS Proc NLIN, $r^2 = 0.88$). This exponential decline suggests a relatively simple mixing and replacement of the original high-nitrate groundwater with more recent recharge from unfertilized soil. If this view is correct, it implies that the residence time of groundwater nitrate in the watershed (the inverse of the flushing rate) was 3.3 years.

Streamwater nitrate concentration in Morris Run, draining the RFBS stream, changed little during the first four years of monitoring and seedling establishment (1992-1995), but then increased, almost steadily, to a peak in 2002 (Figure 6). After the 2002 peak, nitrate concentration declined sharply, returning to the 1992-1995 levels by 2005 and continuing to decline into 2007 when the monitoring ended. Nitrate concentration was higher between 1997 and 2004, than during either the first (1992-1995) or final (2005-2007) years of monitoring ($p < 0.05$, Tukey's multiple comparison test). Nitrate trends in the

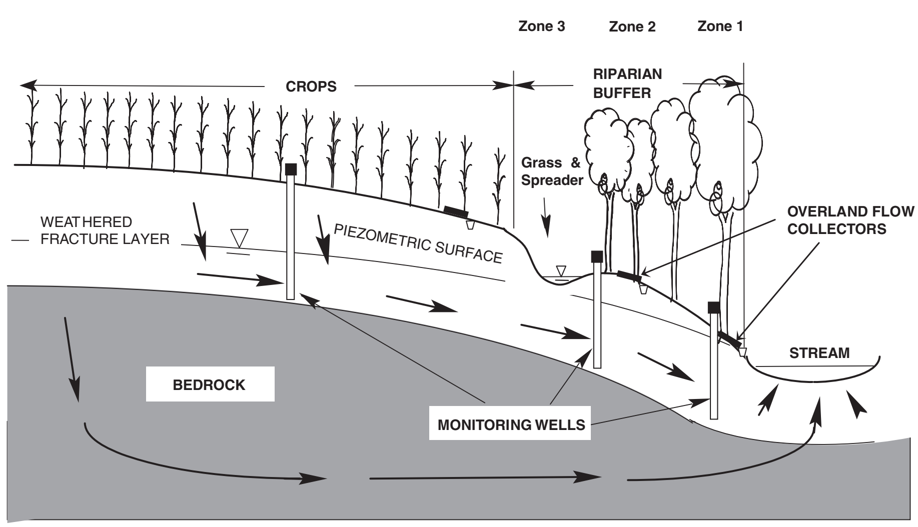

FIGURE 4. Location of Sampling Wells and Overland Flow Collectors Along a Transect Running From the Crop Area Through the Three Zones of the RFBS to the Stream in the Morris Run Watershed. Not drawn to scale.

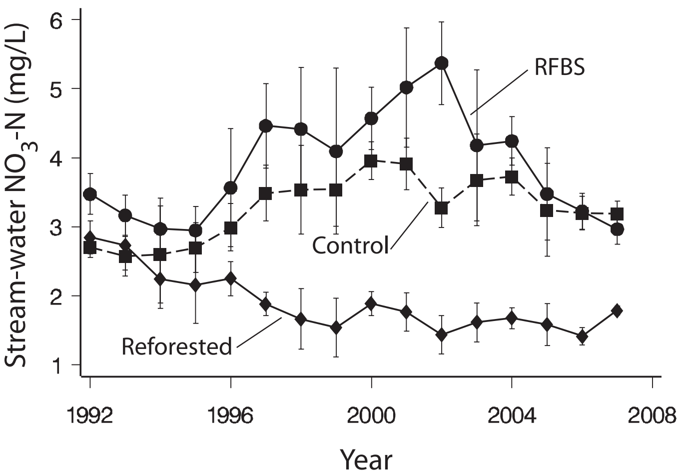

FIGURE 6. Mean Annual Streamwater Nitrate Concentrations Sampled at Regular (two or three week) Intervals. Error bars are ±1 standard deviation. The means of samples taken within 20% of base flow (approximately 80% of the total number of samples) averaged 2% higher (6% maximum) than those shown.

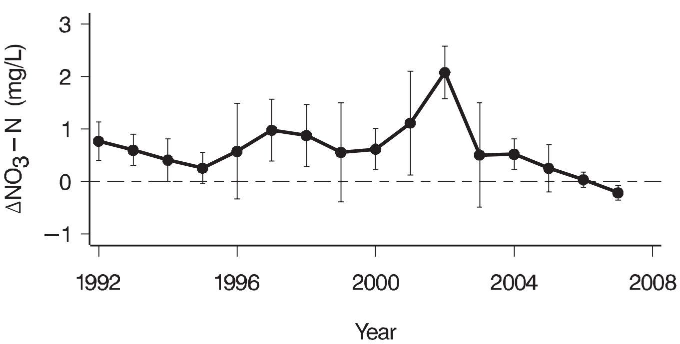

FIGURE 7. Mean Annual Average of Differences in Streamwater Nitrate Concentration Between Paired Same-Day Samples From the RFBS and Control Watersheds. Error bars are ±1 standard deviation.

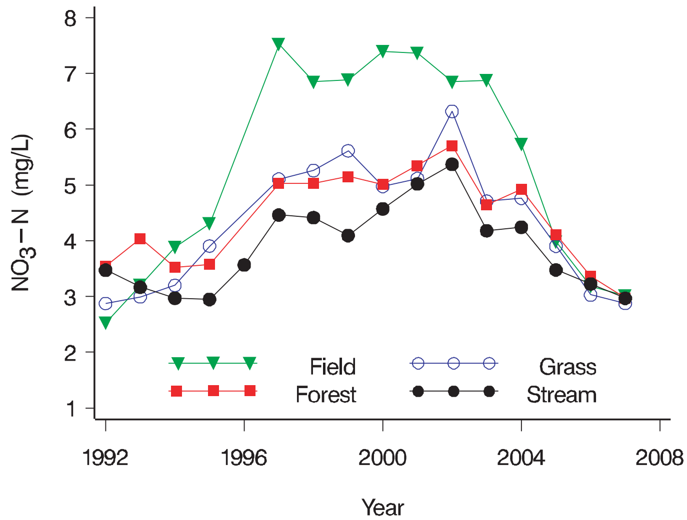

FIGURE 8. Mean Annual Nitrate Concentrations in Groundwater and Stream Water in the RFBS Watershed Over the Project Period. "Field" refers to wells located upslope of the RFBS; "Grass" refers to wells located at the interface of Zones 3 and 2; "Forest" refers to wells at the interface of Zones 2 and 1. Standard errors for individual points averaged 0.50 (range: 0.30 to 1.92) mg/l for groundwater and 0.13 (range: 0.05 to 0.24) mg/l for stream water. Sample sizes for individual points ranged from 15 to 28.

control stream were broadly similar to those of the RFBS stream and, like the RFBS stream, showed a large increase in nitrate concentration between the years 1995 and 2000. Trends in nitrate concentration in the RFBS stream relative to the control stream are shown in Figure 7 as ΔNO_3-N, which represents the difference (RFBS less control) between paired (same day) samples. ΔNO_3-N showed relatively little trend until 2001 when it increased sharply to a peak in 2002, which was significantly higher than the initial value in 1992 ($p < 0.05$, Tukey's test). After 2002, ΔNO_3-N declined steadily, falling below its initial 1992 value in 2005 ($p < 0.05$) and below zero (RFBS < reference, $p < 0.05$) by the end of the monitoring in 2007. While the results of this paired comparison support the conclusion that the RFBS reduced stream-water nitrate in the RFBS stream, they remain less than fully conclusive. The significant peak in ΔNO_3-N of 2002 was not consistent with a buffer effect and illustrates that the control stream could not fully compensate for trends in the RFBS stream that were unrelated to RFBS implementation.

The complex trends in streamwater nitrate in the RFBS watershed may be attributable to the combined dynamics of groundwater nitrate, tree growth, and variations in precipitation. Groundwater nitrate-N in the field (just upslope from the RFBS) increased from 2.5 mg/l in 1992 to a peak of 7.5 mg/l in 1997 that was significantly higher ($p < 0.05$, Tukey's test) than in all previous years (Figure 8). Groundwater in the field remained near 7 mg/l through 2003 and then declined steadily, with concentrations in 2005–2007 being significantly ($p < 0.05$) lower than in the 1997-2003. The 1992-1997 increase likely reflects an increase in fertilizer application that occurred at the beginning of the study when farm management changed hands. However, the increase in application cannot be verified because application rates prior to 1991 were not recorded. The decline in upslope groundwater nitrate that began in 2004 was probably related to the reduced nitrogen applications in the latter years of the study (Figure 2), although it remains unclear why the decline in groundwater concentration was abrupt rather than gradual. This may have been related to the location of fertilizer application, which varied with crop rotation. High precipitation in 2003 and 2004 (Figure 2) may also have been a factor but it should be noted that similarly high precipitation in 1997 was followed by a peak in groundwater nitrate concentration. Groundwater nitrate concentration downslope of Zone 3 of the RFBS (the grassed portion with the level lip spreader) generally increased from 1992 to a peak in 2002, when it was higher ($p < 0.05$) than in 1992-1994. The nitrate concentration of the groundwater downslope of Zone 2 (reforested) also followed an upward trend to 2002, but differences between years were not significant ($p > 0.05$). For groundwater nitrate from both zones, the trend in increasing concentration appeared to lag the increase in the cultivated field by two to three years, as did nitrate concentration in the stream water draining the RFBS. That is, the apparent upward trend that began in the field after 1992 did not begin until after 1994 in Zone 3, and until after 1995 in Zone 2 and in the stream (Figure 8). The lag, if real, is consistent with the response to the cessation of nitrogen application observed in the reforested watershed (Figure 6). These results suggest that groundwater nitrate concentrations within the RFBS and stream were strongly influenced by variations in the groundwater nitrate that entered the RFBS in subsurface flow from the upslope fields. By contrast, the subsequent decline in RFBS groundwater and stream-water nitrate concentrations that apparently began in 2003 cannot be fully explained as a response to upslope inputs because they preceded the decline in upslope nitrate by one year, rather than lagging it. Two factors, both related to buffer function, may have contributed to the nitrate declines. The first is the rapid tree growth in the RFBS, which began about 2000 (Figure 4). Tree growth, however, fails to explain why the decline of 2003 occurred downslope of Zone 2, the grassed portion of the buffer, as well as downslope of the reforested Zone 3. It is possible that the wells at the interface between Zones 2 and 3 were influenced by uptake from the adjacent trees, but if tree growth had been a major influence we should also have seen lower groundwater nitrate concentrations downslope of Zone 2, relative to Zone 3, as the study progressed. The second factor that may have reduced groundwater nitrate in the RFBS in 2003 was the high precipitation that year, the highest recorded during the study. The wet year produced exceptionally high groundwater elevations (data not shown) which may have contributed to nitrate removal via both denitrification and plant uptake. As with tree growth, however, precipitation cannot fully explain the nitrogen declines in the RFBS and the stream. The wet year of 1996 was associated with nitrate concentration increases, rather than declines, and declines in nitrate continued in 2006 and 2007 under near-normal precipitation. We suggest the possibility that vegetative growth, of both grasses and trees, in the absence of tillage, with associated changes in soil properties, such as reduction in bulk density and accumulation of soil organic matter (Maloney *et al.*, 2008), may have enhanced the nitrate removal capabilities of the RFBS over time.

Despite its variability, nitrate concentration in streamwater draining the RFBS was lower than groundwater concentration in the tilled field during all years between 1995 and 2004 ($p < 0.05$, Tukey's test, Figure 8). The initial differences may have simply reflected delays between the upslope loading and export in the stream, whereas the sustained differences clearly suggest that the buffer removed a substantial amount of nitrate within this period. Although budgetary comparisons between inputs and outputs are not meaningful on an annual basis, the effects of a long residence time diminish with longer budgeting periods. Based on a 10-year averaging period (1997-2006), we estimated that the buffer removed 65 kg/year of N, or 90 kg/ha of buffer area per year (Figure 9). This removal was 26% of the upslope input of subsurface nitrate, substantially lower than the average removal efficiency of 72% for nitrate in forested buffers reported by a recent meta-analysis (Mayer *et al.*, 2007). The low removal efficiency of the RFBS cannot be attributed to a low areal rate of N removal, because the 90 kg/ha/year of this study was in the range reported by other studies with higher efficiencies. For example, studies by Peterjohn and Correll (1984), Vellidis *et al.* (2003), and Hoffmann *et al.* (2006) reported removal rates of 44, 83, and 260 kg/ha/year and removal efficiencies of 89, 78, and 64%, respectively. Instead, the efficiency may have been low simply because inputs from upslope were high. The RFBS received an average water flux of 0.37 m^2/day (i.e., 0.37 m^3/m of Zone-3 upslope

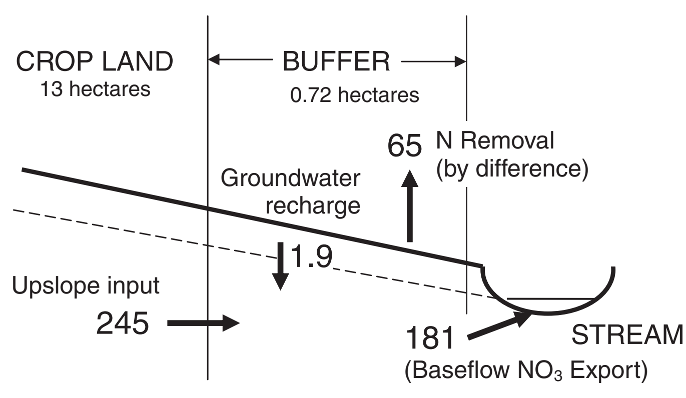

FIGURE 9. Subsurface Nitrate Budget (kg/year) for the Morris Run (RFBS) Watershed 1997-2006.

boundary per day) and an average subsurface nitrate input of 2.7 g N/m/day (245 kg/year from Figure 9 passing through 275 m of upslope boundary). Both of these fluxes are among the highest reported from studies of riparian buffers (Vidon and Hill, 2006). Two factors may explain why they were high. First, the buffer received the convergent flow from an entire basin to the origin of a first-order stream, whereas most previous riparian buffer studies have focused on incremental inputs from lateral flow paths. Headwater streams typically lie in convergent basins with larger contributing area per unit length than do higher order streams (Bren, 1998; McGlynn and Seibert, 2003; Burkart *et al.*, 2004). Thus, the apparently high water and nutrient input fluxes that we observed may actually be characteristic of much of the landscape. Second, many studies of riparian buffers have been conducted at sites where flow is constrained to shallow pathways in hydric soils (Hill, 2000). Such sites are conducive to denitrification and have a high potential for nitrogen removal (Simmons *et al.*, 1992; Gold *et al.*, 2001; Hefting *et al.*, 2004), but these same characteristics may limit the flux of water that passes through them (Vidon and Hill, 2006). In the Piedmont setting of the present study, it is likely that most of the water flow was preferentially constrained to the approximately 6 m of saprolite, while a small fraction passed through the underlying fractured bedrock (Rose, 1992). Most of the subsurface flux probably passed below the upper organic soil horizons where denitrification may be most intense (e.g., Clément *et al.*, 2002). However, somewhat deeper water may have been subject to less intensive denitrification, as has been observed to occur deeper in riparian soils, where subsurface water flowing through long pathways may interact with heterogeneously distributed organic-rich sediments (Hill *et al.*, 2004). Plant uptake may also have removed significant nitrate from the water in shallower pathways (Lowrance *et al.*, 1984; Peterjohn and Correll, 1984), although we quantified neither plant uptake nor denitrification. Water following the deepest pathways may have been subject to little or no nitrogen removal (Böhlke and Denver, 1995; Puckett *et al.*, 2002), explaining in part the relatively modest removal efficiency that we observed. Nonetheless, the relatively robust areal removal of 90 kg/ha/year suggests that the RFBS, despite its high water and nutrient loading, functioned as an effective nitrogen sink.

Stream and Groundwater Phosphorus

TP concentrations in streamwater varied from year to year without consistent trends in any of the three streams – averaging 0.045, 0.037, and 0.26 mg/l in the RFBS, reference, and reforested streams, respectively, between 1992 and 2007 (data not shown).

SRP averaged approximately 67% of TP and similarly showed no temporal trends. SRP in groundwater in the cultivated field of the RFBS watershed averaged 0.028 mg/l without long-term trends ($p > 0.05$). Groundwater concentrations within the buffer (Zones 2 and 3), however, were initially similar to those of the cultivated field, but between 1997 and 2007 averaged 0.019 mg/l which was less ($p < 0.05$) than in the cultivated field. These within-buffer concentrations were lower than the average (0.045 mg/l) in the stream draining the RFBS watershed. This result is consistent with observations that streamwater phosphorus in agricultural streams is controlled less by groundwater supply than by inputs of sediments from overland flow (Taylor and Kunishi, 1971). Thus, although the buffer may have removed phosphorus from subsurface flow, this removal does not appear to have influenced stream water concentrations significantly. Other studies (Peterjohn and Correll, 1984; Osborne and Kovacic, 1993; Clausen *et al.*, 2000) have similarly reported an absence of significant phosphorus removal from groundwater flow.

Overland Flow

Overland flow was collected from 23 storms between 1997 and 2007. The geometric mean sediment concentration of water was 105 mg/l as it entered the RFBS from the grass waterways (Table 1, Above Zone 3), and was reduced to 72 mg/l as it flowed from the level lip spreader into Zone 2 (Above Zone 2), and to 60 mg/l as it exited Zone 2 (Below Zone 2) toward the stream. Both of these reductions were significant ($p < 0.05$), relative to the concentration entering the RFBS, but the apparent incremental reduction of 12 mg/l as the water flowed through Zone 2 was not significant ($p > 0.05$). Assuming that infiltration of water during storm flow was negligible, these concentrations imply that the RFBS removed 43% of the sediment transported from the field, while Zone 3 and the level lip spreader alone removed 32% although, as noted above, the 11% difference was not significant. The absence of significant additional removal within Zone 2 does not necessarily imply that the reforested area was a less effective filter than the grass-spreader combination because water reaching Zone 2 had already been filtered through Zone 3. Preferential deposition of coarser, more rapidly settling particles typically produces enhanced removal efficiency within the first few meters of a

TABLE 1. Geometric Mean Concentrations (mg/l) for Overland Flow Collections From All Runoff Events Collected From 1997 Through 2007 (23 events for suspended solids, 19 for other analytes).

	Above Zone 3 (Field)	Above Zone 2 (Grass)	Below Zone 2 (Forest)
Total suspended solids	104.9 (64-172)	73.2[1] (45-120)	59.9[1] (39-92)
Nitrate-N	0.099[1] (0.056-0.176)	0.081[1] (0.052-0.127)	0.261 (0.183-0.372)
Ammonia-N	0.057[1] (0.039-0.082)	0.035[1] (0.025-0.048)	0.077[1] (0.057-0.105)
Soluble reactive phosphorus	0.34[1] (0.23-0.51)	0.30[1] (0.21-0.43)	0.43 (0.34-0.55)
Particulate phosphorus	0.30 (0.21-0.43)	0.22[1] (0.16-0.30)	0.23[1] (0.17-0.33)

Note: Numbers in parenthesis correspond to the lower and upper 95% confidence limits of the geometric mean.
[1]Means not significantly different ($p > 0.05$, Tukey's test).

filter strip, regardless of vegetation (Cooper *et al.*, 1987; Daniels and Gilliam, 1996; Syversen and Borch, 2005). The 43% removal observed in this study, while substantial, was lower than removal rates of 60 to >90% reported by several other studies of riparian buffers (e.g., Peterjohn and Correll, 1984; Sheridan *et al.*, 1999; Lee *et al.*, 2003; Schoonover *et al.*, 2006). This study's lower removal rate may be partially explained by the role of other conservation measures practiced on the study site. Overland flow reached the buffer only after leaving contoured strips and traversing grassed waterways which themselves have been shown to remove much of the filterable sediments (Fiener and Auerswald, 2003).

Nitrate concentration in overland flow did not change significantly in Zone 3, but increased ($p < 0.05$) with passage through Zone 2 (Table 1). Despite this increase, the average concentration of nitrate-N exported from Zone 2 toward the stream (0.26 mg/l) remained below average streamwater and groundwater concentrations (>2 mg/l, Figures 6 and 8). On an annual basis, storms accounted for <10% of total nitrogen export from the watershed (based on intensive storm sampling not reported here). Thus the nitrate supplied by the RFBS to overland flow detracted negligibly from the overall performance of the buffer. Ammonia concentrations in overland flow were not significantly ($p > 0.05$) affected on passage through the buffer and, like those of nitrate in overland flow, were too low to be a factor in buffer performance.

SRP did not change ($p > 0.05$) in its passage through Zone 3, but increased in Zone 2 to concentrations averaging 26% higher than those entering Zone 3 ($p < 0.05$) (Table 1). Particulate phosphorus, in contrast, declined by 22% across the whole buffer ($p < 0.05$), but did not change significantly in Zone 2. The decline in the concentration of particulate phosphorus was comparable to the increase in SRP concentration, yielding no net effect of the buffer on TP in overland flow. This result contrasts with other reports of high (~75%) removal of TP from overland flow in reforested buffers (Clausen *et al.*, 2000; Vellidis *et al.*, 2003). The absence of removal in this study may be in part attributable to unmeasured upslope removal in the grass waterways.

CONCLUSIONS

A reforested riparian zone that surrounded the headwaters of a perennial stream was established in an agricultural field in the Mid-Atlantic Piedmont in 1992. This study found that a 35 m-wide three-zone RFBS removed 26% of the subsurface nitrate flux and 43% of the suspended sediment concentration delivered from upslope. TP was not removed by the buffer. Although the nitrogen removal efficiency was lower than reported by a number of other studies of riparian buffers, the areal removal of 90 kg/ha/year was well within the range of reported removal rates. Relatively high subsurface fluxes of water and nitrate into the buffer explain the discrepancy, but we note that high fluxes may be more common in small headwater basins than along larger streams that have been more commonly studied. There was evidence that the effectiveness of the buffer in removing nitrate increased approximately 10 years after the buffer was established, corresponding to the onset of rapid tree growth. However, temporal variations in nitrate input fluxes and precipitation prevented a conclusive assessment of the role of tree growth. The grass filter strip between the forest and the cultivated field included a swale with a level lip spreader to disperse concentrated overland flow into the reforested area which functioned effectively to remove suspended sediments. It is important to recognize that this study did not address the indirect influences of riparian reforestation on water quality that arise from habitat improvements, including enhancement of habitat area within the stream (Sweeney *et al.*, 2004). These improvements, in turn, enhance the ability of the stream to take up and process nutrients through processes such as in-stream denitrification that are critical to the protection of downstream ecosystems (Mulholland *et al.*, 2008).

ACKNOWLEDGMENTS

Originally funded by the USDA Forest Service, the Pennsylvania Department of Forestry, and the USEPA Chesapeake Bay Program, the project became a USEPA 319 National Nonpoint Source Monitoring Program project in 1997, supported by the Pennsylvania Department of Environmental Protection. Support was also provided by the Stroud Endowment for Environmental Research. At various times during the course of the project, technical assistance was provided by the USDA Forest Service, the Pennsylvania State Bureau of Forestry, and the USDA Natural Resource Conservation Service. The project also benefited greatly from extensive cooperation from the Natural Lands Trust, which owns the land, the Brandywine Conservancy, which holds the conservation easements, and Mr. Sonny Hicks, who manages the farming operation. We thank Carl Rohr for supportive project management and Bernard Anderson for his dedicated and meticulous field and laboratory work throughout the project. We also thank Don Meals and two anonymous reviewers for their insightful reviews of the manuscript.

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