Mitigation of Nonpoint Pollution by a Riparian Forest Buffer in an Agricultural Watershed of the Mid-Atlantic Piedmont

STROUD PRESERVE WATERSHEDS NATIONAL MONITORING PROJECT

Final Report

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Table of Contents

Introduction

Riparian forest buffers have become well-established as a management practice that can reduce the surface and subsurface transport of agro-chemicals 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 reforestation. This difficulty reflects, in part, a discord between the very high nutrient and sediment removal rates that many studies have demonstrated (see reviews by Lowrance et al. 1997, Mayer et al. 2007), and the cautions that these potentials are not always achieved (e.g., Dillaha et al 1989b, Dosskey 2001, Dosskey 2002, Vidon and Hill 2004). While such cautions do not lessen the advisability of riparian reforestation, which in any case enhances 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 forestation 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 non-buffered agricultural riparian zones—it is often lands less suitable for tillage that are left in forest.

The Stroud Preserve riparian reforestation project was a demonstration of the three-zone Riparian Forest Buffer System (RFBS) developed by the U.S. Department of Agriculture (USDA) -Forest Service (FS) (Welsch 1991). The primary objectives of this project were to:

(1) evaluate the non-point source reductions of the RFBS in the relatively high-relief terrain of the Mid-Atlantic Piedmont,
(2) assess the time required after reforestation to achieve significant sediment and nutrient reduction, and
(3) establish specific guidelines for planting and managing forest buffers zones in the Mid-Atlantic region.

Initiated in 1992, the project involved three experimental agricultural watersheds in the Stroud Preserve, a southeastern Pennsylvania farm protected by conservation easements (Fig. 1). The streams lie in the drainage of the Brandywine River, which flows into the Delaware Estuary. Prior to 1992, all three watersheds were primarily in crop production (maize, soybeans, hay) under a soil conservation plan including contouring and crop rotation. Water quality was compromised by elevated nutrients and suspended sediments.

The RFBS was established between 1992 and 1994 in a 15-ha watershed (Morris Run) that is primarily in row crop production. The RFBS consists of: Zone 1, a streamside strip (~5 m) of permanent woody vegetation for stream habitat protection; Zone 2, an 18-20 m strip of managed forest upslope from Zone 2; and Zone 3, a 6-10 m wide grass filter strip. Zone 1 included existing streambank trees; Zone 2 was converted from hay and crops to hardwood seedlings; and a level-lip spreader (to disperse concentrated overland flow) was constructed in Zone 3. A second watershed (Mine Hill Run) was unaltered and maintained in agricultural production comparable to that of Morris Run, as a long-term reference watershed. The third watershed (Half Way Run) was taken out of agricultural production and 75% of its area was reforested.

The water quality monitoring design used paired watersheds supplemented by mass balance estimates of nutrient removal by the riparian forest buffer. The paired-watershed design involves comparisons between the treatment (RFBS) and reference both before and after intervention. In this project the "intervention" is represented by the maturation of the riparian forest buffer and was expected to produce a gradual rather than immediate effect.

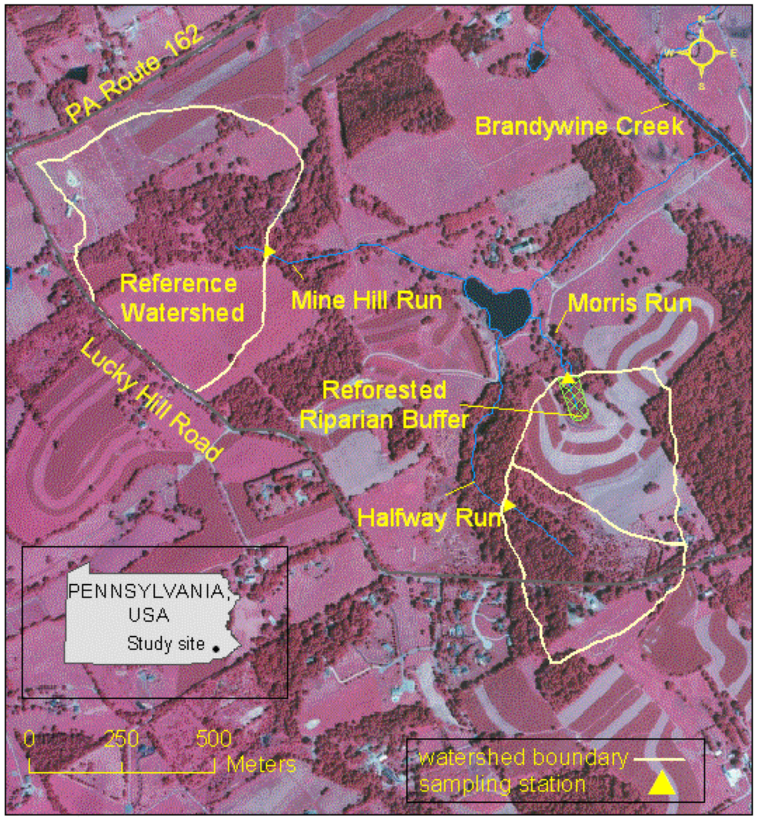

Figure 1. Project watersheds on the Stroud Preserve. The Morris Run (treatment) sampling station is located at 39º56' 41" N, 75º39'13"W.

The project received financial support from 1991 through 1995 from the USDA-FS, the Pennsylvania State Bureau of Forestry, and the Chesapeake Bay Program. From 1997 to 2007 the project was funded by the Pennsylvania Department of Environmental Protection (PA-DEP) as a Clean Water Act Section 319 National Nonpoint Source Monitoring Program (NNPSMP) project. The NNPSMP is coordinated by the U. S. Environmental Protection Agency (USEPA). At various times during the course of the project, technical assistance was provided by the USDA-FS, the Pennsylvania State Bureau of Forestry, and the USDA-Natural Resource Conservation Service (NRCS). The project also benefited greatly from extensive cooperation with the Natural Lands Trust, which owns the land, the Brandywine Conservancy, which holds the conservation easements, and Mr. Sonny Hicks, who managed the farming operation.

This report summarizes data for the entire 16 years of the project and represents the final completion report for the ten years of PA-DEP funding of the USEPA-NNPSMP project. Additional detail has been presented in previous reports, including the final report to the Chesapeake Research Consortium (Newbold et al. 1995), and the annual reports (Year 1 to Year 9) to the PA-DEP (Newbold and Sweeney 1998-2006). A brief description of the project and results appears in Newbold et al. (2008).

Site description

The study was conducted on three small watersheds located in the Piedmont province of southeastern Pennsylvania (Fig. 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. A weathered rock or saprolite extends to a typical depth of 5-7 m with a bedrock consisting mainly of fractured schist. Streamflow from each watershed was continuously gaged through a 90° V-notch weir.

The 14.9-ha treatment watershed is drained by a perennial first-order stream, Morris Run. All but a few hectares of the treatment watershed are maintained in strips (primarily corn and soybeans) under contoured crop rotation. In April of 1992, a Riparian Forest Buffer System (RFBS) surrounding Morris Run was established in accordance with the specification published by the USDA-FS (Welsch 1991). The RFBS (Fig. 2) consists of: Zone 1, a streamside strip (~5 m) of permanent woody vegetation for stream habitat protection; Zone 2, an 18-20 m strip, upslope from Zone 1, reforested in hardwoods; and Zone 3, a 6-10 m grass filter strip with a level-lip spreader between Zone 2 and the tilled field. The reforestation of Zone 2 consisted of a mix of sugar maple, red oak, tulip poplar, white ash, black walnut, and trembling aspen planted as 1-year seedlings at approximately 3-m spacing and protected by plastic (1.3-m) tree shelters. Prior to the planting, the buffer area consisted of mowed grass, some tilled area, and a narrow riparian strip (3-10 m) of hardwood trees and brush.

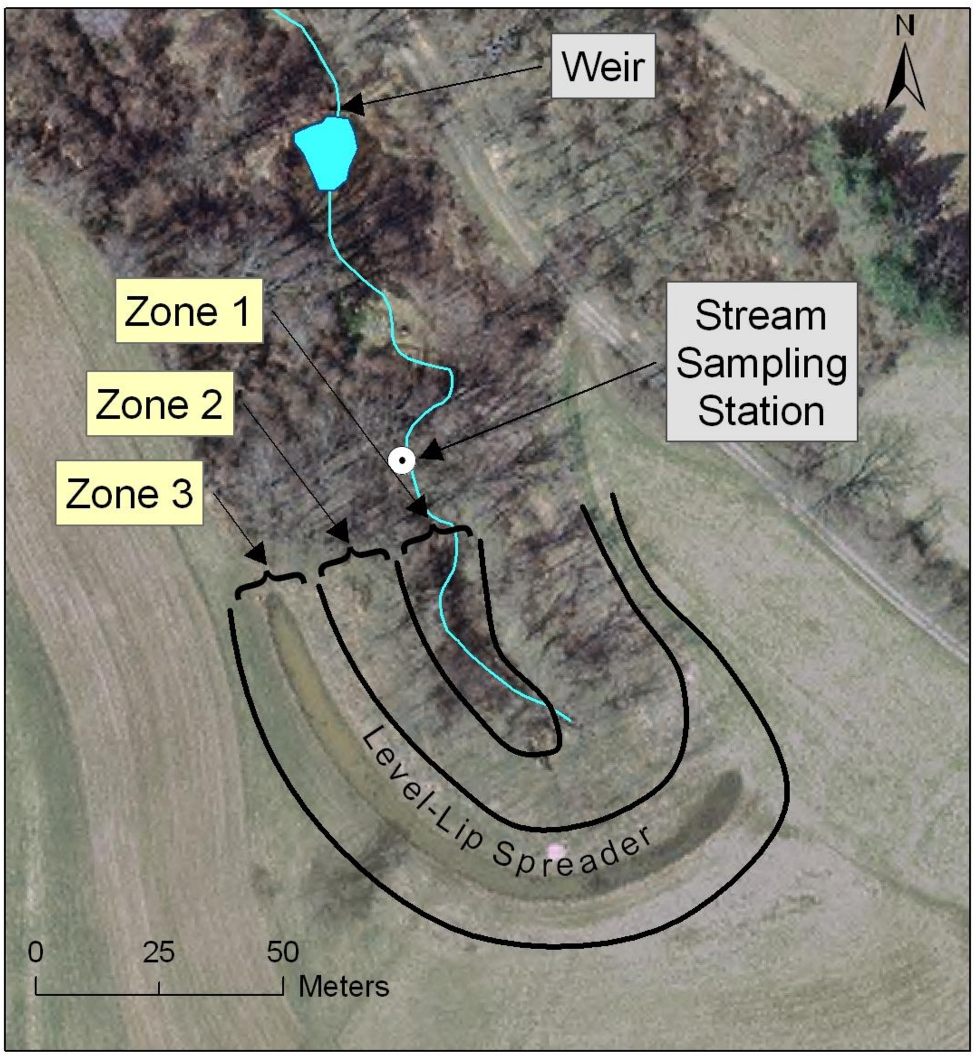

Figure 2. Morris Run (treatment) stream and the riparian forest buffer system with level-lip spreader in April 2005.

In accordance with the 3-zone buffer specification, the grassland zone was contoured in May 1994 to form a level-lip spreader, designed by the USDA-NRCS. The purpose of the spreader is to intercept surface runoff that is delivered to the buffer via grassed waterways and release the runoff to the forested buffer as dispersed sheet flow in order to minimize erosion within the buffer. The level-lip spreader consisted of a level grass strip 3 m wide by 140 m long (10 feet by 450 feet), oriented along the original-contour and bordering the upslope boundary of Zone 2. The level-lip was constructed at the original contour level, rather than as a raised berm, in order to minimize settling and thus to maintain the design level indefinitely. Actual construction required some cutting and filling, with a maximum cut of 0.5 m (1.5 feet) below the original contour and a maximum fill no more than 0.25 m (0.8 feet) above the original contour. The downslope (discharge) face of the level-lip had a maximum slope of 1:10. A swale was cut into the contour running parallel to the level-lip on the upslope side. The centerline of the swale was graded to an elevation 0.5 m (1.5 feet) below that of the level lip and was located 3.1 m (10 feet) upslope from the level-lip, giving a grade on the upslope face of the level-lip of 1.5:10. On the upslope side, the swale was graded into the original contour at a maximum grade of 1:5, which required a maximum upslope distance of 10 m (30 feet) from the level lip.

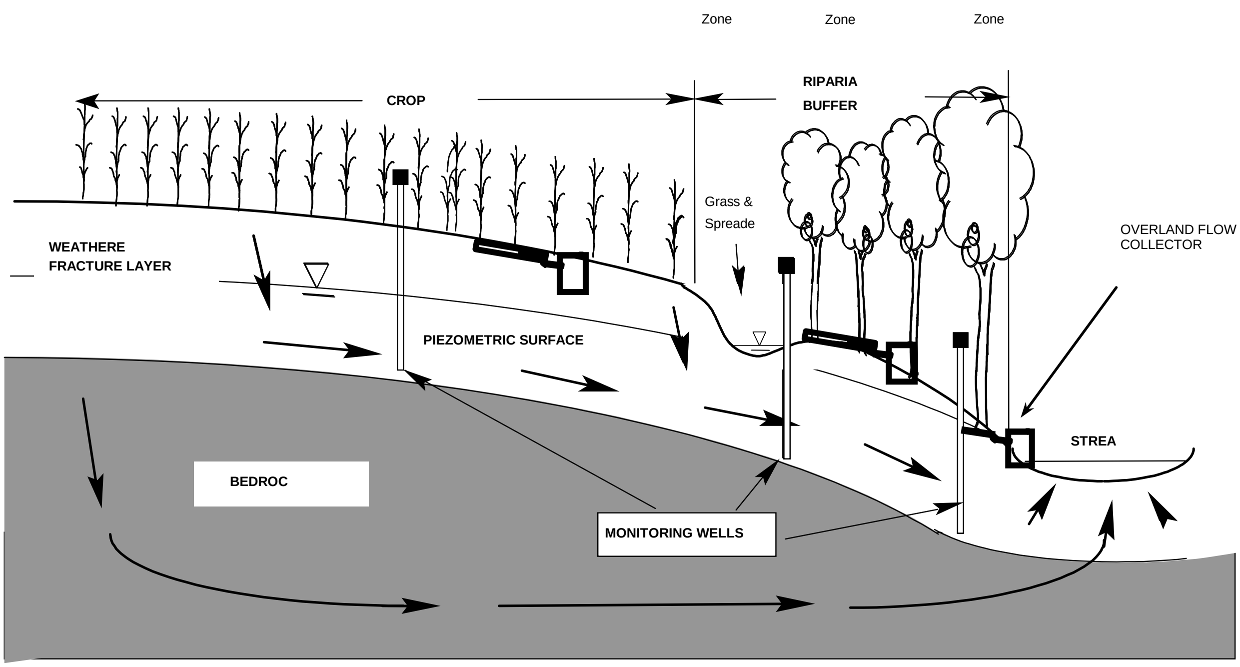

Figure 3. Schematic transect showing location of sampling wells and overland flow collectors in relation to the crop area, the three zones of the riparian forest buffer system, and stream in the Morris Run (treatment) watershed.

A survey of the spreader conducted in 2005 revealed localized subsidence of up to 0.12 m (0.4 feet). Comparison with the original survey showed that the subsidence was largely localized to the limited areas where the spreader had been constructed of fill to a level above original grade. The subsidence had allowed concentrated flow to enter Zone 2 of the buffer, contributing to the development of two head-cuts at the source of Morris Run. In June of 2006, 60 m (200 feet) of the spreader was re-leveled by removing the top thatch layer, applying soil, and regrading. The head-cuts were filled to create a swale which terminated in a drop structure that dissipates energy into a rock-lined plunge-pool at the head of the stream. A local opening of the canopy was created to support the growth of grass in the swale. Both the re-leveled portion of the spreader and the swale were stabilized with coconut fabric and seeded with a tall fescue mix. In addition, the swale was planted with deer tongue and riverbank rye, as seeds, and switchgrass, as plugs.

Nineteen sampling wells (5-7 m deep, screened in the lower 0.5-3 m) were installed in the Morris Run (RFBS) treatment watershed along six 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 tilled field. The wells in the field were placed ~ 70 m upslope from Zone 3 (Fig. 3). Three of the well transects were installed in April of 1991 and were constructed of 10-cm PVC pipe, screened below 3-6 m. The wells were back-filled with sand and sealed with bentonite and concrete. The other three well transects (9 wells) were installed in June of 1994. Each of these wells consists of a stainless steel screened well point, implanted at a depth of 6-8 m, with sampling access via a polyethylene tube, accompanied by a 5-cm diameter PVC well with a depth of 4-6 m screened over the lower 1.6 m. The 5-cm PVC wells were used only to measure depth to the water surface.

Overland flow collectors (Fig. 4) were installed in the Morris Run watershed in locations shown schematically in Fig. 3. The collectors were modifications of the Low Impact Flow Event sampler described by Sheridan et al. (1996). A major feature of the modification used for this study was the use of a bypass-overflow that diverted water from the collector once it had reached capacity (see Fig. 4). This averted a large bias that could arise if water overflowed the collector while continuing to trap sediment. The upslope collector width (15-25 cm) was set so that most collectors typically captured sufficient volume to analyze without reaching capacity (about 18 L). However, due to the large spatial and temporal variability in overland flow volumes, some filling to capacity could not be avoided. Because at least some collectors did fill to capacity during most of the larger storms, no attempt was made to convert collection volumes to flow estimates or to estimate overland fluxes. The filling may have introduced some bias in concentration estimates arising from variations in concentration during the overland flow event. This bias remains unevaluated.

Two collectors ("Field") were installed in each of two waterways that convey overland flow from the tilled field to Zone 3, the grass area with the level spreader. Overland flow entered the reforested area (Zone 2) only after filling a swale that borders the level-lip spreader in the grass buffer (Zone 3). Once the swale filled, water flowed over the level-lip spreader into Zone 2. Ten overland flow collectors ("Grass") were positioned downslope of the spreader at the interface between Zones 3 and 2. Another ten collectors ("Forest") were positioned downslope from the reforested Zone 2, at the interface with Zone 1 near the stream.

The reference watershed is 34.1 ha in area, drained by Mine Hill Run. Most of the watershed is planted in alfalfa, corn, and soybeans, and also uses a soil conservation plan that includes contouring and crop rotation. A sparsely forested, brushy zone extends 50-200 m from the stream. Land use in this watershed is being maintained without alteration. A residual portion of the watershed along the northern border remains in private ownership. This land, however, is under conservation easement, restricted to no more than two dwelling units, and the potential impact on Mine Hill Run is considered minimal.

The third watershed (14.5 ha) is drained by Half Way Run, which is surrounded by a mature forest extending at least 30 meters from the stream. The remainder of the Half Way Run watershed within the Stroud Preserve was taken out of agricultural production and planted with mixed hardwood seedlings in the spring of 1991. A paved road passes within this watershed near its southern border and a portion of the land lying south of this road is privately held. The private land is used for horse grazing and supports two dwelling units. Potential impacts from this portion of the watershed on Half Way Run are considered minimal.

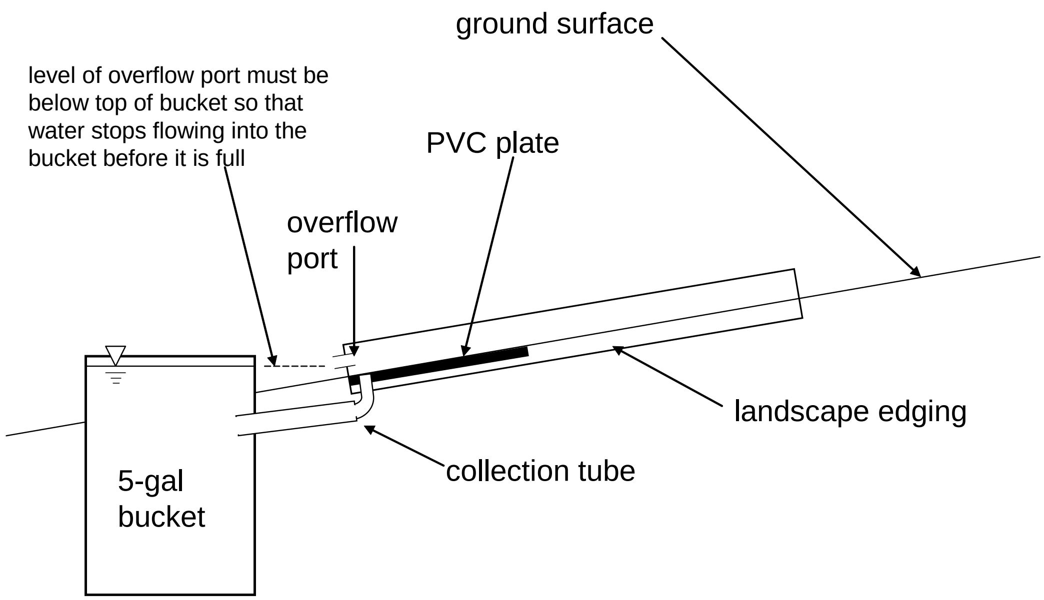

Figure 4. Schematic for overland flow collector as developed by S. J. Alberts (2000) based on the Low Impact Flow Event sampler designed by Sheridan et al. (1996).

Methods

The water quality monitoring program was based on a paired watershed design (Wilm 1944, USEPA 1993). Under a paired watershed design, a treatment is applied to one of two similar watersheds while the other serves as a control or reference. A calibration period precedes application of the treatment in order to establish a relationship in water quality characteristics between the two watersheds. A treatment effect is inferred if the relationship changes after the treatment is applied. In the present study, the treatment involved the planting of seedlings that were not expected to have any measureable influence on water quality for several years. Thus, the paired-watershed approach was modified to allow the calibration period to run concurrently with the period of seedling establishment. In addition, the statistical analyses were adapted, as detailed below, to detect an emerging trend in the absence of a pre-specified time of onset for post-treatment effects. To supplement the paired watershed design, nutrient and sediment retention by the riparian buffer are estimated by mass balance, using data from groundwater monitoring wells and overland flow collectors.

Streamwater was sampled from each stream just upstream from the weir at regularly scheduled periods (1-to-3-week intervals from 1992 to 1997 and at 2-week intervals from 1997 through March 2007). Automated water samplers (ISCO®) were used to collect hourly streamwater samples from Morris Run (RFBS) and Mine Hill Run (reference) during 8 storms per year from 1997 through 2001 and again from 2005 through early 2007. Four samples spanning peak flow were selected from each storm for analysis. Storm-generated overland flow was collected from the overland flow collectors after each of fifteen storms from 1998 through 2001 and eight storms from 2005 through early 2007. The groundwater monitoring wells were sampled quarterly from 1992 through March 2007 except in 1996, when no groundwater samples were collected. Sampling of storms from both streamwater and overland flow were suspended for three years (2002-2004) because tree growth through 2001 had lagged expectations and the value of continued intensive sampling prior to significant tree growth was deemed limited. Storm sampling was resumed in 2005 after rapid tree growth was evident.

Nitrate (including nitrite) was determined after membrane (0.24 µm) filtration by cadmium reduction (U.S. EPA method 353.2, U.S. EPA 1993). Ammonia-N was determined by the colorimetric automated phenate method (U.S. EPA method 350.1, U.S. EPA 1993). Soluble reactive phosphorus (SRP) was determined on filtered (0.45-µm pore size, membrane) samples by the ascorbic acid method (U.S. EPA method 365.1, U.S. EPA 1993). Total phosphorus (TP) was determined on unfiltered samples by the ascorbic acid method after digestion by ammonium persulfate (U.S. EPA method 365.1, U.S. EPA 1993). Total dissolved phosphorus was determined as total phosphorus in membrane-filtered samples. Total suspended solids concentration (TSS) was determined by filtering an aliquot (100-to-3200 mL, as filter capacity permitted) of sample onto a pre-weighed 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 based estimates of sediment transport on TSS concentrations, rather than on suspended-sediment concentrations as assayed from whole samples (Gray et al. 2000). To minimize bias from settling of coarser (sand-size) particles, all samples, both from overland flow collectors and streams, were inverted and vigorously shaken during or immediately prior to sub-sampling for TSS.

Forest growth in Zone 2 of the RFBS was monitored by measuring the diameter at breast-height (1.5 m) (dbh) of each tree 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 grid point of a 3 x 3-m grid within the RFBS was scored as either lying directly below tree canopy or below open sky.

Analysis of variance (ANOVA) in conjunction with Tukey's multiple comparison test was used to identify year-to-year variations in nitrate and phosphorus concentrations in streamwater and groundwater, and for within-year spatial variations along the field-to-stream well transects. Formal paired-watershed comparisons involving the entire record, 1992 to 2007, could be applied only to the regularly scheduled sampling for nutrients because storm-intensive sampling of nutrients and TSS was not initiated until 1997 and TP was irregularly sampled prior to 1997. Two approaches were used for the paired-watershed comparison for nitrate and SRP. The first used a regression design suggested by Grabow et al. (1999) for the detection of gradual change. The model for this design is given by:

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + e$$

Where Y is the nutrient concentration in the treatment (RFBS) stream, β_0 is the regression intercept, X_1 is the simultaneously sampled concentration in the reference stream, β_1 is the regression parameter reflecting the covariance between the two watersheds, X_2 is the time elapsed after the end of the calibration period, β_2 is the rate of change per unit time in Y that is independent of the reference watershed (X_1), and e is the random error. For this study, the time, X_2, was measured from 1 January 1997, which allows a 5-year calibration period and occurred prior to rapid tree growth (see below). Prior to 1 January 1997, X_2=0. Results from this model, however, were treated with caution because, as presented below, there were complicating trends in streamwater nutrient concentrations. Therefore, an alternate approach was used, which was intended to provide a simpler more flexible interpretation of paired-watershed dynamics. For this second approach, the differences between paired samples (treatment minus reference) were analyzed by one-way ANOVA, using year as the main effect. Reference-corrected year-to-year differences were identified by Tukey's multiple comparison test. Galeone (2000) used paired differences in conjunction with a non-parameteric test of pre- to post-treatment periods. Our use of the paired differences is also related to the general analysis-of-covariance (ANCOVA) approach to paired watersheds as presented by USEPA (1993). The ANCOVA can detect a change in the relationship between the two watersheds (the slope of Y on X_1), or in a constant difference between the two watersheds (a pre-to-post change in the regression intercept). If the slope of Y on X_1 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. The simpler ANOVA formulation provides the advantage that it can be easily extended to multiple periods, such as individual years. 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 intensive storm sampling of nutrients and TSS was conducted in two periods, the first running from 1997 to 2002, and the second from 2005 to early 2007. Rapid tree growth began near the end of the first phase. Therefore, the paired watershed approach was used to test whether tree growth induced reductions in storm-flow nutrient and TSS concentrations. In this case, there were two clearly defined before-and-after periods, so the ANCOVA approach (USEPA 1993), as described above, was applied. For each storm, the flow-weighted average concentration was calculated from the four available samples. These samples spanned the peak of the hydrograph and so characterized the bulk of the export from each storm. The natural log of the flow-weighted concentrations from the RFBS stream (Y) were regressed on the respective paired reference-stream concentrations (X_1) and analyzed by ANCOVA for a change in the slope and intercept between the two periods.

Data from sediment and nutrients transported in overland flow were analyzed by log-transforming the analyte concentration from each collector on each storm date, then computing the mean transformed concentration for each of the three collector positions (Field, Grass, and Forest) on each date. Differences in analyte concentration from Field-to-Grass and Grass-to-Forest were tested by a single two-way (date × position) ANOVA with one observation per cell. Thus, it is the between-storm error rather than the between-collector error that was used to test the hypothesis of a buffer effect. Differences between positions were evaluated by Tukey's test. Data were back-transformed to geometric means for tabular reporting. All effects were tested at the $P<0.05$ significance level. All statistical analyses were conducted using the SAS Version 9.1.

A mass-balance estimate of nitrogen removal by the RFBS was calculated as the difference between subsurface nitrate flux into the buffer from the tilled field and baseflow export of nitrate via the stream, as described further in the Results section. Although we did not characterize subsurface flow pathways, we are confident that the streamwater exports captured nearly all of the groundwater flow both because the piezometric surface conformed reasonably with surface topography and because annual water yields agreed well with regional watershed water balances of similar geology (Vogel and Reif 1993).

Results

Riparian forest growth

Tree growth in the RFBS was slow from 1992 to 1998, with significant annual mortality from drought and deer damage. Much of the initial planting stock was replaced during these years. The deer damage included both browsing, which stunted growth as the trees emerged from the 4-foot shelters, and rubbing by bucks in rut, which caused mortality among those trees that had grown above the browse level. In 1998 it was recognized that more aggressive measures were needed to assure vigorous forest development. These included permitting more deer hunting on the farm, annual herbicide (glyphosate) treatment of each tree, the use of taller (5-foot) tree protectors (both plastic and wire mesh) as the trees matured, and the planting of relatively mature trees to replace mortality, especially into critical remaining gaps. Replacement trees included river birch and green ash, as riparian-adapted species.

Between 1998 and late 2006, woody basal area increased 20-fold to 3.1 m^2/ha with an average increase of 40% per year (Table 1, Figure 5). This increase was nearly all due to growth, with very little attributable to replacement planting. The 2006-basal area represents ~5-15% of that typical of a mature forest (20-60 m^2/ha; Shure et al. 1998, Cooper-Ellis et al. 1998, Compton and Boone 2000). In the most-recent two years (2004 to 2006) the growth in basal area averaged 26% per year. Average diameter (breast height) increased from 1.1 cm in 1998 to 7.5 cm in October 2006, while stem density declined from 617 stems/ha to 477 stems/ha (Table 1). Canopy cover increased from 41% in 2002 to 67% in 2005, although a somewhat lower figure of 59% was recorded in 2006 (Figure 6). The small decline from 2005 to 2006 is attributed to measurement error because it occurred despite a 24% increase in basal area and an 8% increase in litter fall (see below) over the same time period.

Litter fall in Zone 2 (the reforested buffer) was 357 g/m^2 in 2006, having increased by 195% since 2002 (Table 2). Litter fall in Zone 2 at Morris Run was 64% of that recorded in the mature forest of Half-Way Run despite the fact that basal area remains only ~5-15% of that expected of a mature forest.

Table 1. Basal area and density of trees in reforested buffer (Zone 2) at the Stroud Preserve. Area of planting is 0.30 ha (0.74 acre). Original planting density was 750 stems per hectare (approx. 3.7-m centers).

Date	Density (stems/ha)	Average Diameter (DBH, cm)	Basal Area (m^2/ha)
15-Aug-98	617	1.1	0.15
15-Jun-99	593	1.2	0.15
15-Jun-00	570	1.7	0.30
15-Sep-01	593	2.7	0.65
15-Jun-02	637	3.0	0.80
15-Oct-02	567	3.7	0.98
15-Jun-03	600	3.7	1.08
20-Oct-03	577	4.5	1.46
24-May-04	600	4.4	1.45
7-Oct-04	553	5.5	1.95
21-Jun-05	507	6.2	2.15
24-Oct-05	483	7.0	2.49
16-Jun-06	477	7.5	2.82
3-Oct-06	477	7.5	3.09

Table 2. Litter fall inputs (leaves, wood, and other) expressed as dry weight (g/m^2) collected from buckets suspended above the stream surfaces (zone 1) and from the Morris Run Reforested Buffer (Zone 2).

	Morris Run (Zone 1)	Morris Run (Zone 2)	Mine Hill Run (Zone 1)	Halfway Run (Zone 1)
2002	408	121	no data	no data
2003	407	180	433	527
2004	386	198	430	473
2005	508	332	587	654
2006	481	357	860	559

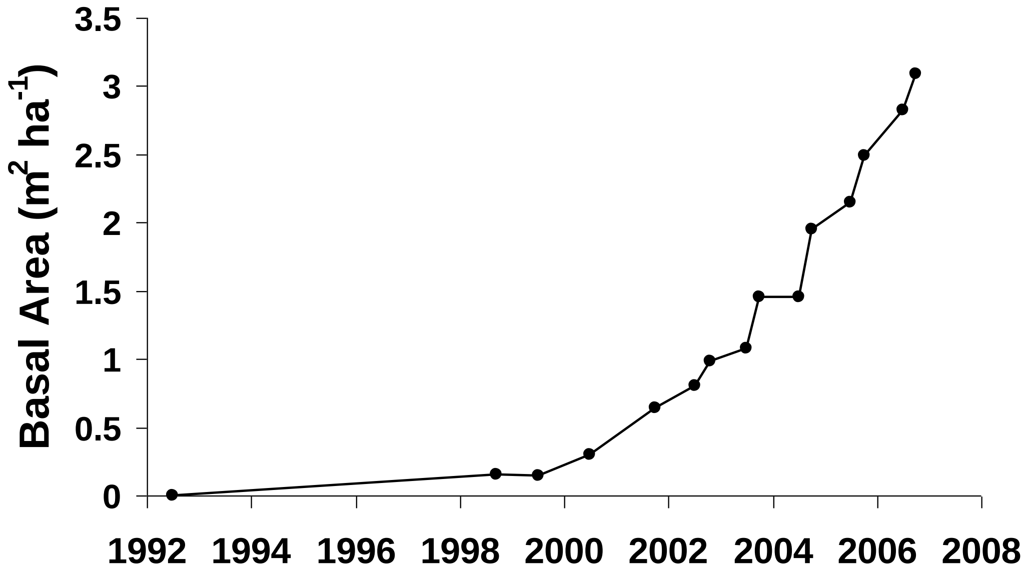

Figure 5. Basal area of trees in Zone 2 of the Morris Run Riparian Forest Buffer System.

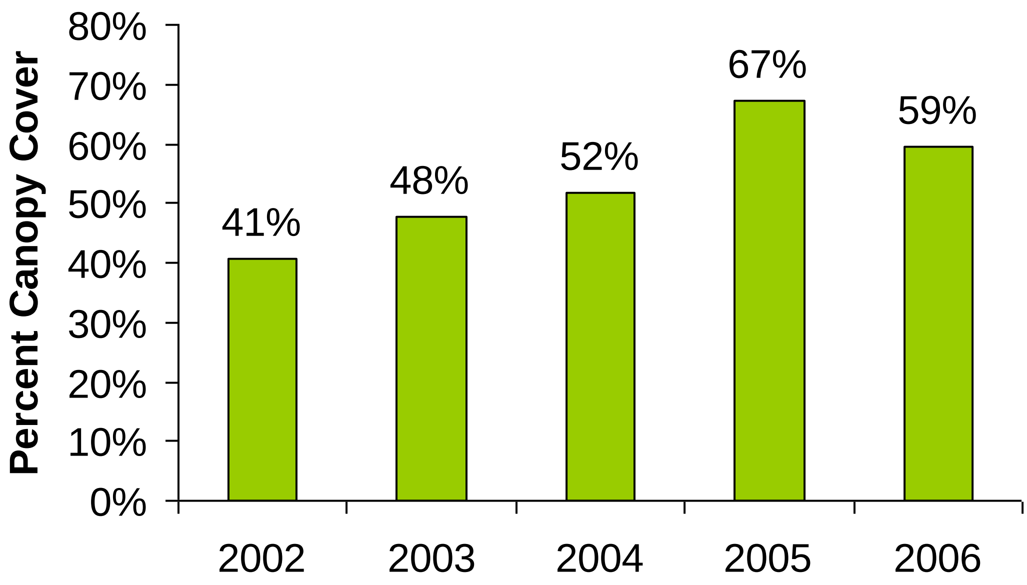

Figure 6. Canopy cover in Zone 2 of Morris Run Riparian Forest Buffer System.

Streamwater 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 7, Table 3). 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 pre-existing pool of groundwater nitrate from the watershed. Over this period, the nitrate concentration declined at an exponential rate of 0.30 y^{-1} (nonlinear regression, r^2=0.88), suggesting 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 the groundwater in the watershed (the inverse of the flushing rate) was 3.3 y.

In Morris Run, the stream draining the RFBS, nitrate concentration was expected to decline relative to that in the reference watershed (Mine Hill Run), as the riparian forest buffer matured. This trend was tested using a paired-watershed regression model designed to test for a gradual trend beginning after an initial calibration period, as suggested by Grabow et al. (1999) and

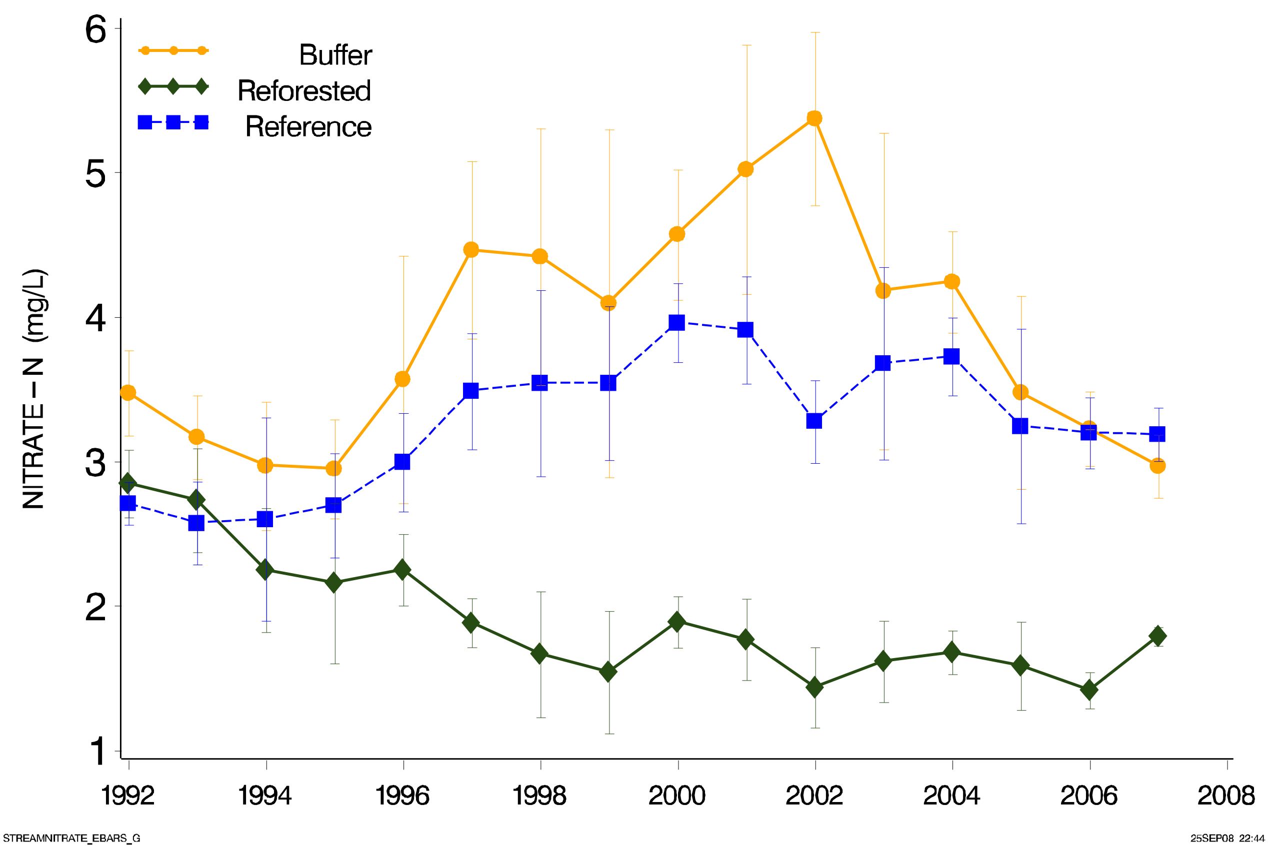

Figure 7. Streamwater nitrate concentrations in three streams on the Stroud Preserve sampled at regular (2- or 3-week) intervals. Morris Run drains the treatment stream with the reforested buffer planted in 1992. Mine Hill Run is the reference stream. Half Way Run drains a watershed that was largely reforested between 1991 and 1993. The lines join the annual mean concentrations, also presented in Table 3. The symbols are as follows: circle and solid line—Morris Run (buffer); square and dashed line—Mine Hill Run (reference); diamond and dashed line—Half Way Run (reforested).

presented in the previous Methods section. The calibration period was taken to be the first five years (1992-1996) of monitoring during which the seedlings became established but accrued relatively little biomass and were not expected to exert a measureable effect on streamwater nutrient concentrations. This test yielded a significant ($P<0.05$) treatment effect indicating a linear decline in nitrate concentration in the RFBS-stream relative to the reference stream beginning in 1997. The estimated rate of decline (β_2=-0.028 mg L^{-1} y^{-1}), however, was small, equivalent to a decrease in nitrate concentration of 0.28 mg/L over ten years. The result of this test should be treated with caution because, as Fig. 7 illustrates, the actual trends in streamwater nitrate concentrations were complex and perhaps inadequately represented by the paired-watershed regression model. A more detailed description and analysis of the nitrate trends follows.

Streamwater nitrate concentration in Morris Run (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 (Fig. 7, Table 3). 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. One-way ANOVA verified significant differences among years ($F_{15,340}$=25.9, $P<0.05$). 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). The large and steady decline in nitrate concentration that began in 2002 coincided with the onset of rapid tree growth in the reforested buffer and is therefore consistent with a buffer effect. However, the increase in nitrate concentration that occurred prior to 2002 was clearly not consistent with a buffer effect. Thus, trends based on the RFBS stream alone were inconclusive.

As Fig. 7 shows, the nitrate trends in the reference 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. The RFBS concentrations were adjusted for trends in the reference stream by computing ΔNO_3-N, which represents the difference (RFBS-reference) between paired (same day) samples. Use of the paired differences represents an adaptation of the USEPA (1993) method of paired-watershed analysis, as previously described (Methods). A one-way ANOVA of ΔNO_3-N showed a significant year effect ($F_{15,336}$=20.7, $P<0.0001$). Tukey's test was used to distinguish between individual years.

As a result of the parallel trends between the RFBS and reference streams, ΔNO_3-N changed little between 1992 and 2000 (Fig. 8, Table 3). These years were statistically equivalent ($P>0.05$). There was a sharp increase in ΔNO3-N that began in 2001 and peaked in 2002 when ΔNO_3-N was higher than in all other years ($P<0.05$). Immediately thereafter, ΔNO_3-N fell sharply and then continued to decline, falling below the initial value of 1992 ($P<0.05$) in 2005, and below zero (RFBS<reference, $P<0.05$) by the end of the monitoring in 2007. Despite having fallen below the level of the beginning of the study, however, ΔNO_3-N did not fall significantly ($P>0.05$) below the levels of 1993-1996, which were also calibration years. Thus the results of this alternative analysis, while supporting the conclusion that forest growth reduced streamwater nitrate in the RFBS stream, remain less than fully conclusive. This analysis makes apparent, however, that while the reference stream accounted for much of the complicating dynamics of the raw RFBS stream nitrate concentrations (Fig. 7), the adjustment was not entirely successful in this regard. The significant peak in ΔNO_3-N of 2001 and 2002 remains unexplained and so limits interpretation of the results. The next section presents evidence that the streamwater nitrate dynamics in the RFBS stream were strongly influenced by varying concentrations of groundwater nitrate entering the buffer from upslope, and argues that a riparian buffer effect can be inferred from the timing of these variations.

Table 3. Average annual streamwater nitrate concentration (mg/L as N) from samples taken at regular sampling intervals. Means are based on samples taken every three weeks before 1 April 1997 and every two weeks from 1 April 1997 to early 2007.

	Morris Run (Reforested Buffer)	Mine Hill Run (Reference)	Half Way Run (Reforested Watershed)	ΔNO_3N: Difference between paired samples (Reforested Buffer – Reference)
1992	3.47	2.71	2.85	0.76
1993	3.17	2.57	2.73	0.60
1994	2.97	2.61	2.25	0.36
1995	2.95	2.7	2.16	0.25
1996	3.57	2.99	2.25	0.58
1997	4.46	3.49	1.88	0.97
1998	4.43	3.54	1.67	0.89
1999	4.09	3.54	1.54	0.55
2000	4.57	3.96	1.89	0.61
2001	5.02	3.91	1.77	1.11
2002	5.37	3.28	1.44	2.09
2003	4.18	3.68	1.62	0.50
2004	4.24	3.73	1.68	0.51
2005	3.48	3.24	1.58	0.24
2006	3.23	3.20	1.42	0.03
2007 (Jan-Mar)	2.97	3.19	1.79	-0.22

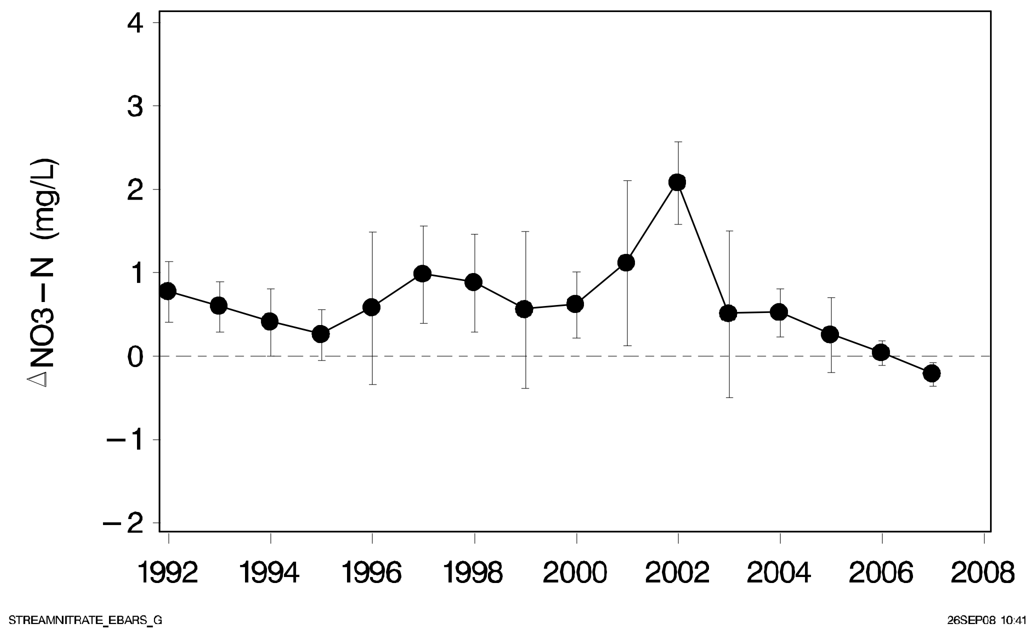

Figure 8. Annual mean difference in nitrate concentration, ΔNO_3-N, between Morris Run (RFBS planted in 1992) and Mine Hill Run (reference stream), from regularly scheduled samples. Differences were calculated between samples taken the same day, normally within two hours of one another. Error bars are ±1 standard deviation. The dashed line is a zero reference.

Groundwater nitrate

Trends in concentrations of groundwater nitrate within the RFBS watershed provide an alternate means of assessing the effectiveness of the riparian buffer that complements the paired-watershed approach presented above. As illustrated in Figure 3, the groundwater monitoring wells in the Morris Run treatment watershed were arranged in relation to the three zones of the riparian forest buffer system so that each transect follows the down-slope pathway of water from the tilled field, through the riparian buffer zones, to its emergence in the stream. The six transects converged on a relatively short reach of stream that was sampled at a single point (upstream from the weir) near its exit from the watershed.

Figure 9 shows the mean annual nitrate concentration (the average of all six transects) at each location along the down-slope pathway for selected years (years not shown can be found in earlier annual reports). In the first year (1992), nitrate concentration increased slightly along the down-slope transects from field, through the buffer, to the stream, but in 1993 there was no clear trend along the down-slope transect. By 1997, a declining down-slope trend had developed, with lower nitrate concentrations in the streamwater than in groundwater from the tilled field in each year through 2004 ($P<0.05$, ANOVA, Tukey's test). In 2005 and 2006, however, no down-slope trend was evident and the difference in nitrate concentration between the tilled field and the stream was no longer significant ($P>0.05$).

The declining down-slope trends during the mid years of the study suggest that nitrate was removed from the groundwater as it passed beneath the buffer area en route to the stream. But interpretation is complicated by large variability in the concentration of nitrate entering the buffer from the upslope field. Groundwater nitrate in the field increased from 2.5 mg/L in 1992 to a peak of 7.5 mg/L in 1997 (Fig.10). The nitrate concentration remained near 7 mg/L through 2003, and then declined steadily to ~3 mg/L in 2007. The increase, first observed in 1993, may reflect an increase in the rate of fertilizer application to the tilled fields about the time the riparian buffer was installed, i.e., in 1992. At this time farm management changed hands and no records of fertilizer application prior to 1992 are available. The subsequent decline in upslope groundwater nitrate that began in 2004 was probably related to a reduction in 2002 in the rate of nitrogen application to the hay strips in the field. As Fig.10 shows, nitrate concentrations within the RFBS (Zone 3 and the reforested portion of the buffer or Zone 2), as well as in the streamwater draining the RFBS, increased in parallel with the increase in upslope nitrate, but with a 3-4 year time lag. This delay is consistent with the response to the cessation of nitrogen application observed in the reforested watershed (Fig. 7) and strongly suggests that the increases in nitrate in the buffer and stream resulted from the slow downstream movement of nitrate-laden groundwater from the field. In 2003, nitrate concentrations fell sharply in both RFBS zones as well as in the stream, while field concentration remained high before declining in 2004. That is, the declines in the buffer and stream occurred *prior* to the decline in the field, rather than lagging it by 3-4 years. The timing of these declines is consistent with the onset of rapid tree growth in the RFBS (Fig. 5). We interpret this as evidence that the observed forest growth contributed to nitrate removal within the RFBS during that period. The next section presents budgetary estimates of nitrate removal.

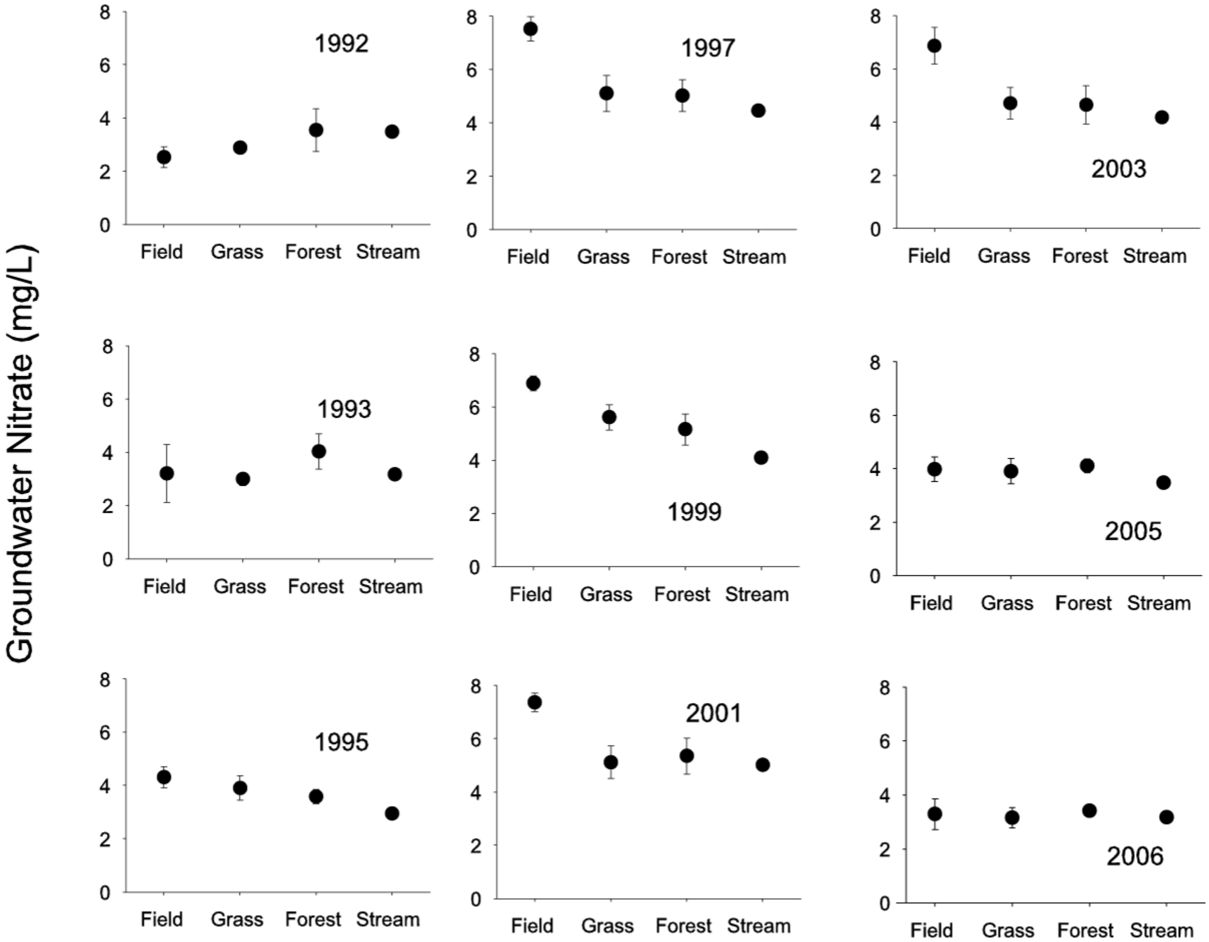

Figure 9. Mean annual nitrate concentrations in groundwater and streamwater in the treatment watershed. Sampling locations are as illustrated in Figure 3. The category "Field" represents wells located near the down-slope end of the tilled area; "Grass" represents wells near the down-slope boundary of Zone 3 (which includes the level spreader); "Forest" represents wells down-slope from the reforested buffer (Zone 2) near the up-slope boundary of Zone 1. The "Stream" is Morris Run, which originates within the buffer and was sampled at the outflow from the buffered area. Error bars represent ± 1 standard error. Sample sizes ranged from 15 to 28.

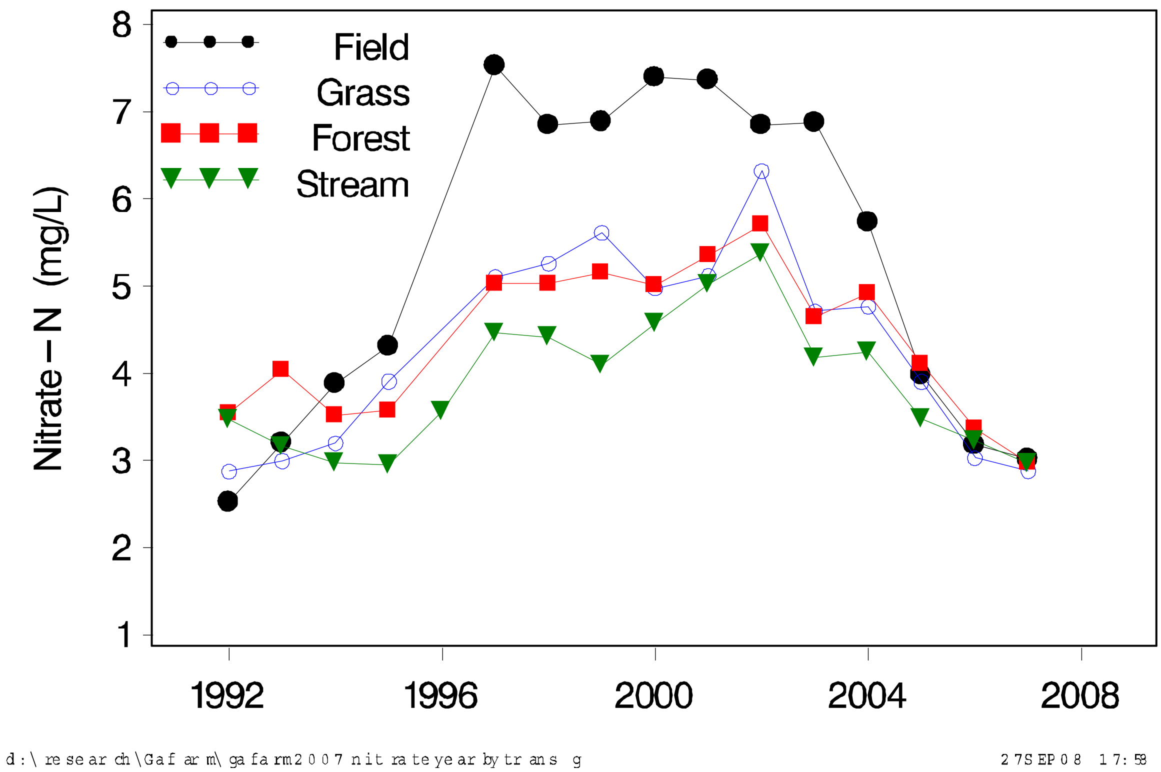

Figure 10. Mean annual nitrate concentrations in groundwater and streamwater in the treatment watershed. These are the same data as presented in Fig. 9, but including all sampling years. Standard deviations for individual points averaged 1.35 (range: 0.41 to 2.0) mg/L for groundwater and 0.61 (range: 0.25 to 1.2) mg/L for streamwater. Sample sizes ranged from 15 to 28.

Subsurface nitrate budget for the treatment watershed

As argued in the previous section, groundwater appeared to take several years to move through the RFBS. Thus, a budget of nitrate inputs and outputs that is based on a single-year "snapshot" is of relatively little value because it can not account for changes in nitrate storage within the RFBS. A meaningful budget must cover a period substantially longer than the time required for groundwater to pass through the buffer. Table 4 presents annual inputs and outputs of water and nitrate for the subsurface or groundwater-saturated zone of the Morris Run RFBS.

The length of the stream, from source to weir, lies entirely within Zone 1 of the buffer. Therefore, groundwater outputs from the buffer were assumed to equal the annual baseflow of the stream, which was estimated from the continuous flow record at the weir using a simple hydrograph separation procedure (Lesack 1993). The 1.0 ha of buffer area lies between the stream and the upslope remainder (13.9 ha) of the 14.9 ha-watershed. Thus, inputs to the groundwater in the buffer consist of the groundwater drainage from the 13.9 ha of tilled land plus the groundwater recharge occurring within the buffer itself. It was assumed that the areal rate of groundwater recharge was the same in the field (tilled area) as in the buffer. Thus subsurface input of groundwater from upslope was calculated as 93.3% (13.9/14.9) of the groundwater output and the remaining 6.7% was assigned to recharge from within the buffer. Both infiltration and evapotranspiration may have been somewhat higher (on an areal basis) in the buffer than in the field but the net difference to groundwater recharge, relative to the field, was probably small.

The nitrate input via subsurface flow from upslope was estimated as the product of the influent water volume and the mean concentration in wells in the tilled field (the concentrations shown in Figures 9 and 10). Nitrate carried to the groundwater via soil drainage within the buffer was estimated as the product of the groundwater recharge and the average nitrate concentration (0.88 mg/L) measured in soil lysimeters during the non-growing season (Watts 1997). Although the soil lysimeter measurements were conducted only in 1994, the contribution from this pathway (~1% of inputs) is so small that any reasonable error would not affect the overall budgets. Nitrate output from the buffer was assumed equal to the baseflow export of nitrate in the streamwater, calculated as the baseflow concentration of nitrate multiplied by the respective volume of baseflow water. The baseflow nitrogen concentrations were computed from all the regularly scheduled samples taken at flows less than 50% over baseflow. For all years, these were within ±7% of the values shown in Table 3, which did not exclude samples from elevated flows.

Over the ten-year span of the budget, the difference between inputs and outputs represents the nitrate apparently removed from subsurface flow within the buffer. Table 4 shows these input-output balances on an annual basis but only to represent their temporal variability. They should not, as noted above, be taken as valid estimates of the nitrate removal in any given year. The ten-year averages from Table 4 are shown in a consolidated budget in Figure 11. The estimated average removal rate was 69 kg/yr representing 26% of the upslope inputs. The area of the buffer at Morris Run, including all three zones, is ~1 ha. Thus the subsurface removal of nitrate by the buffer can be expressed as 69 kg per hectare of buffer area per year. The 69 kg originated from a tilled area of 13.9 ha, yielding a removal rate of 5.0 kg per hectare of tilled land draining into the buffer per year. If, as the paired watershed results suggest, the growing forest began to reduce streamwater nitrate concentration only as of 2003, then this average removal rate represents an average of removal that occurred prior to an influence of the reforestation, i.e., in a herbaceous (largely grass) buffer, together with a higher rate of removal effected by the reforestation.

Table 4. Subsurface water and nitrate budgets for the riparian buffer in Morris Run Watershed. The area of the buffer (Zones 1, 2, and 3) is 1.0 ha. The entire watershed area is 14.9 ha.

	Water (m^3/y)			Nitrate (kg/y)				
	Input		Output	Input		Output	Balance	
	Subsurface from upslope	Recharge from within buffer	Stream discharge	Subsurface from upslope	Vertical drainage from soil water	Export to stream	(Input-Output)	
Year							kg/y	%
1997	45306	3259	48565	341	2.6	219	124	36
1998	29949	2155	32104	205	2.6	146	62	30
1999	24712	1778	26490	183	2.6	115	70	38
2000	52072	3746	55819	385	3.3	258	130	34
2001	27066	1947	29013	199	1.7	146	55	28
2002	16063	1156	17219	110	1.0	92	19	17
2003	66194	4762	70956	455	4.2	316	143	31
2004	65605	4720	70325	376	4.2	301	79	21
2005	45012	3238	48250	179	2.8	169	13	7
2006	44276	3185	47462	141	2.8	154	-10	-7

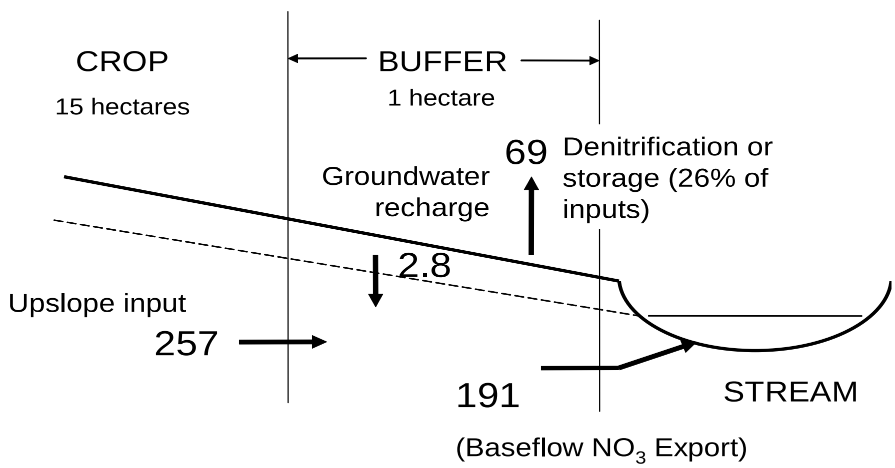

Figure 11. Subsurface nitrate budget (kg/y) for the Morris Run treatment watershed 1996-2006.

Streamwater phosphorus

This section reports trends in streamwater phosphorus concentrations from the regularly scheduled samples in the same way that trends in nitrate concentrations were reported in the previous section. There is, however, an important difference in interpretation. The regularly scheduled samples were taken predominantly at or near baseflow, and the analyses, which are based on concentration data that are not flow-weighted, do not account for the influence of storms on the export of nutrients from the watershed. In the case of nitrogen, this is appropriate because baseflow nitrate accounted for >90% of nitrogen exports (see Sediment and Nutrient Exports). For phosphorus, however, the majority of exports (74 %, Appendix B) occurred during stormflow so that the utility of analyzing trends in baseflow concentrations is limited. The influence of the RFBS on phosphorus exports is considered in a subsequent section.

Streamwater concentration of soluble reactive phosphorus (SRP) in regularly scheduled samples (Figure 12) declined in all three streams between 1992 and 2007 ($P<0.05$, linear regression). Although the declines were significant, they occurred within the context of relatively large sample-to-sample and year-to-year variability, so that linear regressions of concentration versus time explained <10% of the variance in each stream. As described previously for the nitrate concentrations, a paired-watershed regression model was used to test for a gradual change in SRP in the RFBS stream relative to that in the reference stream, beginning after 5 years of tree growth. The data were log-transformed to reduce skew. This test yielded a significant ($P<0.05$) treatment effect indicating a linear decline in SRP concentration in the RFBS-stream relative to the reference stream beginning in 1997. The estimated rate, β_2, of decline in SRP was 0.023/y, or equivalent to a 20% reduction (or about 0.01 mg/L) over ten years.

As in the case of the nitrate concentrations, the results of the test for a gradual decline may not be definitive. As Figure 13 shows, ΔSRP (the paired differences between the RFBS and reference streams) showed no net change over the entire period of monitoring (1992-2007). One-way ANOVA showed significant year-to-year differences ($F_{15,340}=2.35$, $P<0.05$, $R^2=0.09$), but only the year 1999 differed from any of the other years (Tukey's test).On the other hand, a linear regression of ΔSRP versus time for the period 1997-2007 was significant ($P<0.05$), in agreement with the gradual-change model. Thus although SRP in the RFBS stream did decline, both absolutely and relative to the reference stream, this decline was preceded by, and essentially reversed, an unexplained increase that occurred during the calibration period.

Streamwater concentration of total phosphorus (TP) from the regularly scheduled samples (Figure 14) were, on average, about 50% higher than SRP, but unlike SRP did not trend downward through the course of the study. The paired differences (ΔTP, Fig. 15) showed no consistent temporal trends ($P>0.05$). Thus it remains inconclusive, from the regularly scheduled samples, whether the growing riparian forest reduced streamwater phosphorus concentrations As noted above, however, the regularly scheduled samples characterized baseflow concentrations, rather than export loads, which occur primarily as stormflow.

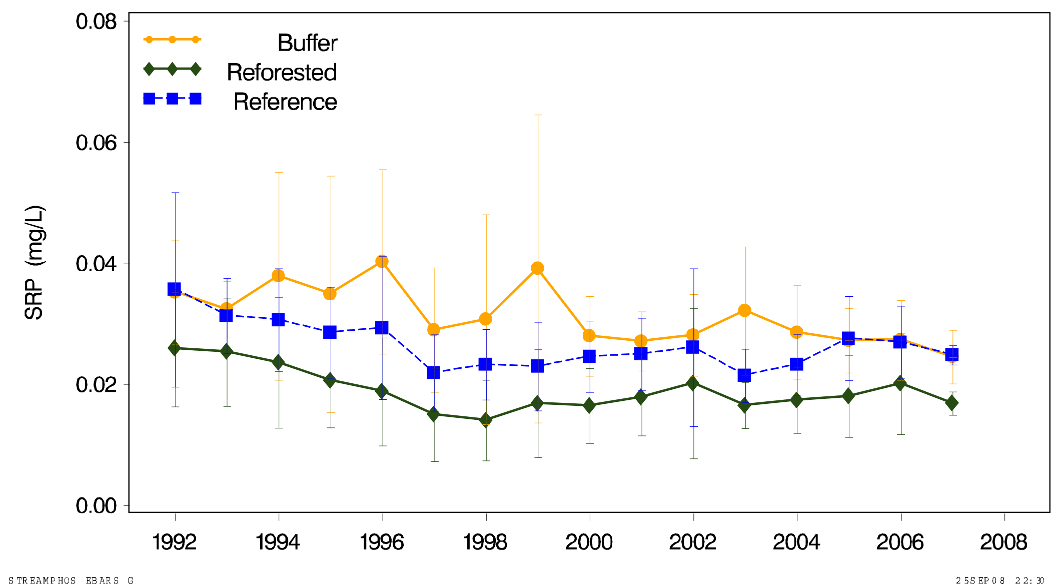

Figure 12. Streamwater soluble reactive phosphorus (SRP) concentrations in three streams on the Stroud Preserve sampled at regular (2- or 3-week) intervals. Morris Run drains the reforested buffer planted in 1992. Mine Hill Run is the reference stream. Half Way Run drains a watershed that was reforested between 1991 and 1993. The lines join the annual mean concentrations. The symbols are: circle and solid line: Morris Run (buffer); square and dashed line: Mine Hill Run (reference); diamond and short-dashed line: Half Way Run (reforested watershed).

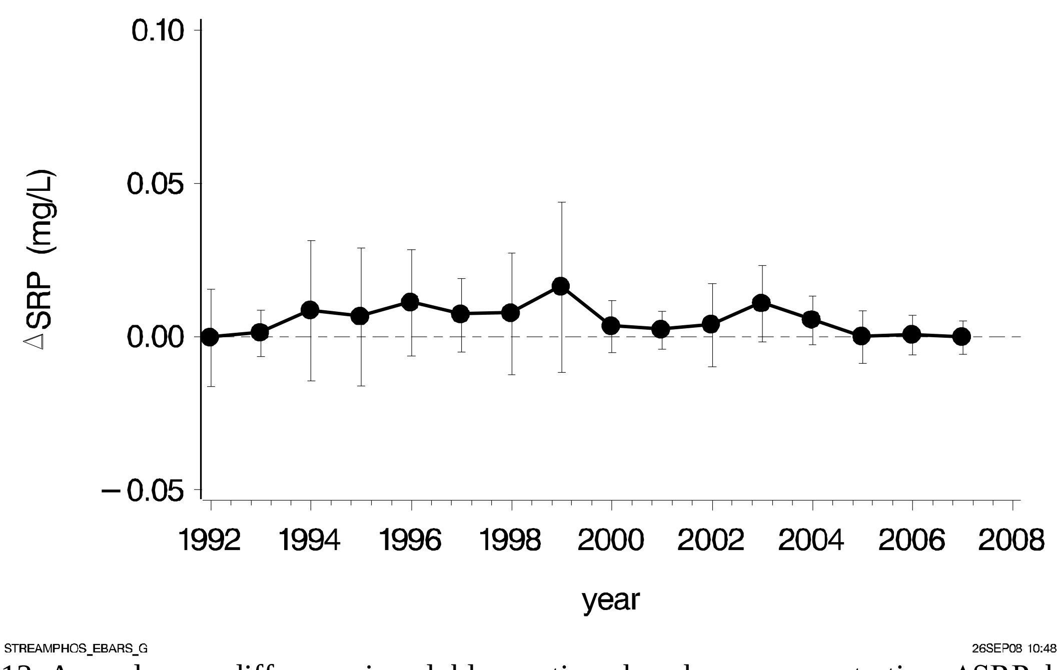

Figure 13. Annual mean difference in soluble reactive phosphorus concentration, ΔSRP, between Morris Run (RFBS planted in 1992) and Mine Hill Run (reference stream), from regularly scheduled samples. Differences were calculated between samples taken the same day, normally within two hours of one another. Error bars are ±1 standard deviation. The dashed line is a zero reference.

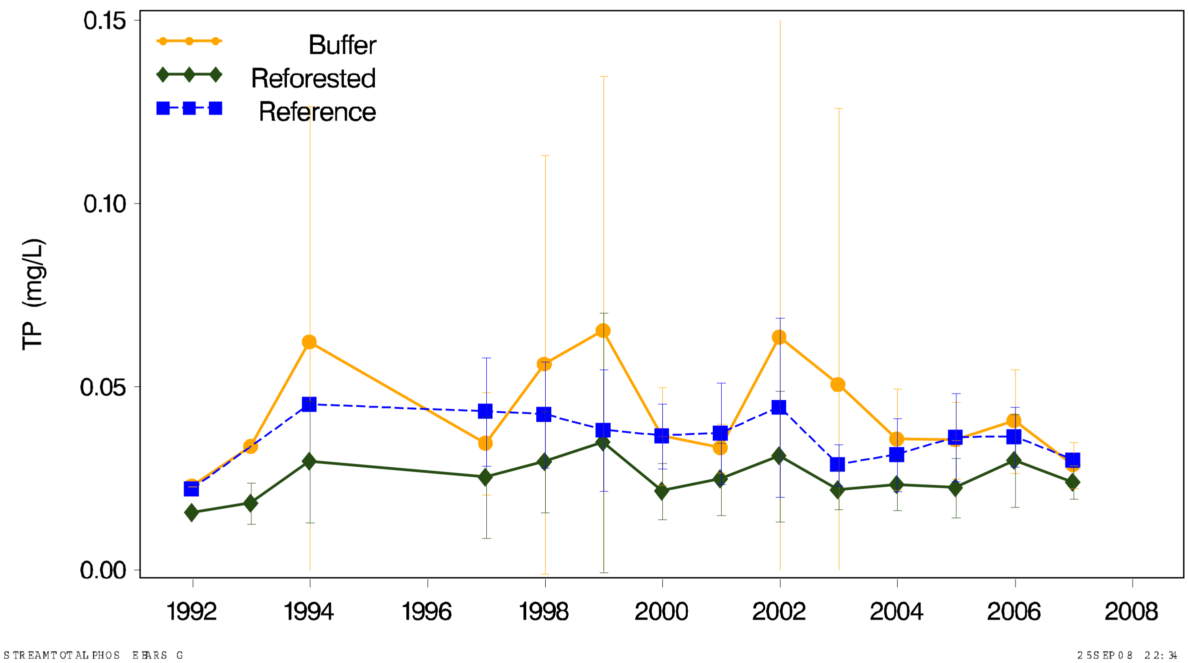

Figure 14. Total phosphorus concentrations in regularly scheduled streamwater samples. The symbols are: circle and solid line: Morris Run (Buffer or RFBS); square and dashed line: Mine Hill Run (reference); diamond and short-dashed line: Half Way Run (Reforested watershed).

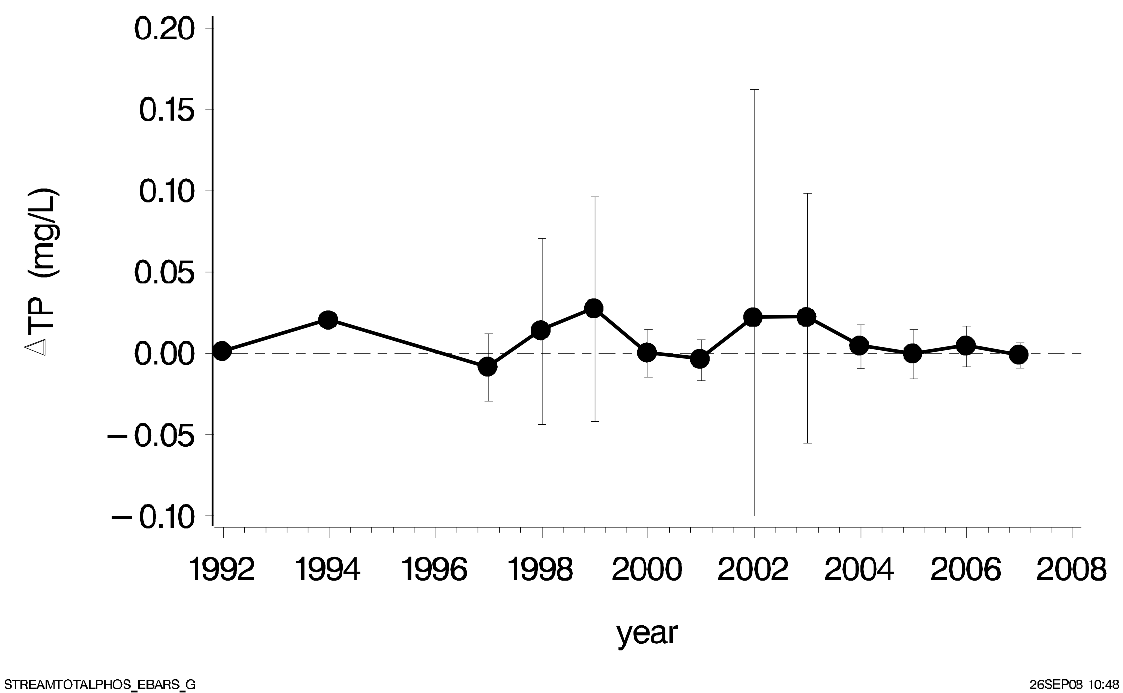

Figure 15. The difference in total phosphorus concentration, ΔTP, between Morris Run (RFBS planted in 1992) and Mine Hill Run (reference stream), from regularly scheduled samples. Differences were calculated between samples taken the same day, normally within two hours of one another. Error bars are ±1 standard deviation. The dashed line is a zero reference.

Groundwater phosphorus

At the beginning of the study (1992-1993), groundwater SRP concentrations (Fig. 16) in the tilled field and at the down-slope boundaries of Zone 3 (grass) and Zone 2 (reforested) of the treatment watershed were all relatively similar and all of these concentrations were similar to the streamwater concentration. Groundwater SRP in the field and in the buffer then declined until 1998 while streamwater SRP fluctuated with no clear trend. After 1998, groundwater SRP in the tilled field rose steadily until the end of the study in 2007, exceeding that of the stream after 2004. Groundwater SRP within the buffer (grass and reforested) remained approximately constant after 1998 at concentrations below those of both the field and the stream ($P<0.05$, two-way ANOVA, Tukey's test).

The increasing divergence, since 1998, of ground-water SRP concentrations within the buffer from those in the field is evidence that the buffer intercepts SRP moving toward the stream from the field. This result has not been widely observed in other studies. Peterjohn and Correll (1984) reported a net release of both SRP and TDP within a forested riparian zone. Osborne and Kovacic (1993) and Hoffmann et al. (2006) both reported alternate periods of release and sequestration. Phosphorus tends to adsorb strongly to soil particles and probably migrates very slowly through the buffer. Thus it is likely that most of the SRP applied to the field was adsorbed within a short distance of entering the buffer, i.e., within Zone 3, where SRP concentration tended to be higher than within the reforested portion of the buffer. The decline in groundwater SRP concentration within the buffer since 1992 may be a long-term response to the cessation of direct fertilizer application within the area converted to buffer. The low concentrations within the forested zone of the buffer suggest that it has a high capacity to adsorb phosphorus and thus forestall "breakthrough" of groundwater phosphorus to the stream.

The relatively low groundwater SRP concentrations observed in the near-stream forested zone of the buffer did not yield similarly low SRP concentrations in the stream, implying that some other source of phosphorus maintained the relatively higher stream concentrations. One possibility is that phosphorus desorbs from sediments delivered to and deposited in the stream via overland flow during storms.

From these results, it appears that the buffer did intercept phosphorus in subsurface flow that entered the buffer, but there is not clear evidence that the buffer reduced streamwater concentrations. However, as noted above, baseflow accounts for relatively little of the total streamwater phosphorus exports. Subsequent sections address phosphorus removal in overland flow and long term trends in annual phosphorus exports.

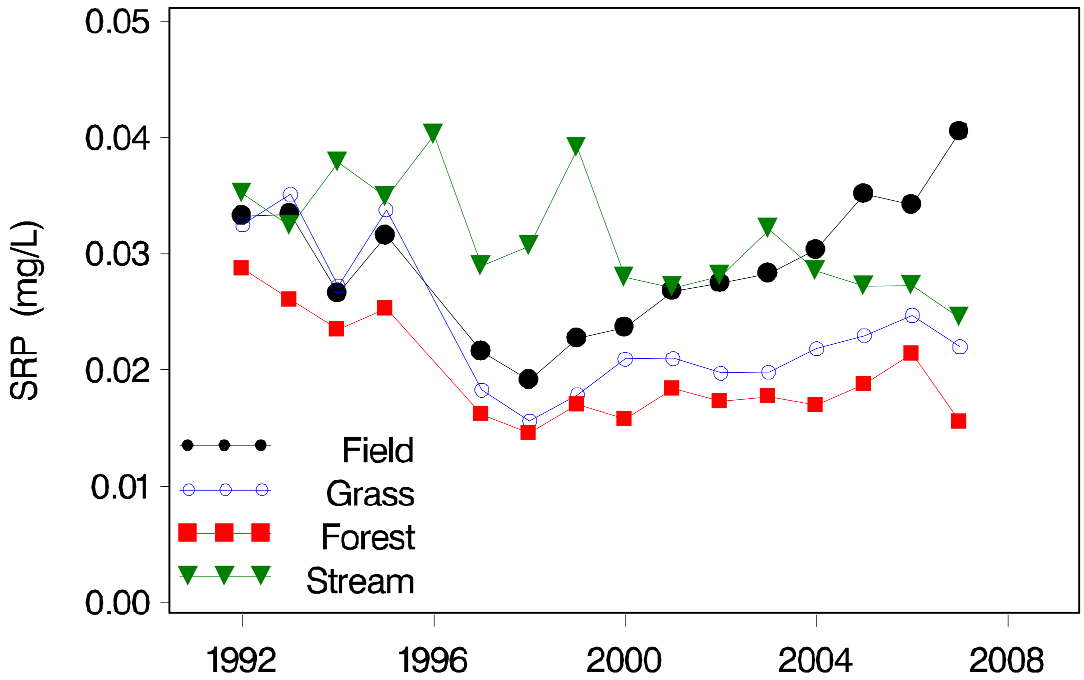

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Figure 16. Mean annual groundwater and stream soluble reactive phosphorus (SRP) concentrations in the Morris Run watershed. Each groundwater value represents the means of 3-4 wells (1992-1993) or 6-7 wells (1994-2007). Each well was sampled at least four times a year, except that no well samples were analyzed in 1996 and wells were sampled only once in 2007 (26 Feb – 2 Mar). The stream concentrations represent the annual means of 18-26 samples. Standard deviations for individual points averaged 0.014 (range: 0.008-0.056) mg/L for groundwater and 0.011 (range: 0.005-0.025) mg/L for streamwater. Sample sizes ranged from 15 to 28.

Sediment and nutrients in overland flow

Overland flow is sampled in the Morris Run watershed in order to assess the effectiveness of the riparian forest buffer in removing sediments and nutrients from surface runoff before it enters the stream.

A series of ten overland flow collectors was positioned at the upslope boundary of the reforested buffer zone (Zone 2), and a second series of ten was positioned down-slope from the reforested zone, near the stream. Overland flow entered the reforested zone only after passing into Zone 3 ("Grass") and filling the swale in the grass buffer that borders the level-lip spreader in Zone 3. When the swale filled, water flowed over the level-spreader into Zone 2 ("Forest"). Overland flow reached Zone 3 via two waterways, each with two overland-flow collectors (Field). The positions of the overland collectors are shown in Fig. 3.

The swale detained the overland flow from small storms, especially during drought periods when the swale holds no standing water. Thus, only relatively large storms generated significant and collectable overland flow through the buffer. The number of collectors that received a volume sufficient for analysis depended on the size of the storm. Storms were considered suitable for inclusion in the statistical analysis, if there were analyzable samples in at least two collectors (out of 10) both upslope of the reforestation (Grass) and down-slope of the reforestation (Forest). The collectors in the waterways (Field) normally fill even in small events.

Overland flow was collected from 23 storms between 1997 and 2007. For all storms combined, the geometric mean concentration of total suspended solids (TSS) in the overland flow as it entered the RFBS from the grass waterways was 105 mg/L (Table 7, “Above Zone 3" or "Field”). The concentration was reduced to 72 mg/L in water flowing from the level-lip spreader into the reforested Zone 2 (“Above Zone 2" or "Grass”), and to 60 mg/L in overland flow as it exited Zone 2 (“Below Zone 2" or "Forest”) toward the stream. Concentrations of TSS delivered from the field varied widely among storms (Figure 17), as did the respective changes in concentration as the overland flow passed through the grass and the forest. However, in 18 of the 23 storms there was at least some reduction in TSS concentration across the entire buffer (i.e., through grass-and-forest combined. The geometric mean concentrations imply that the RFBS as a whole reduced the concentration of TSS transported from the field by 43%. In Zone 3 alone, the concentrations declined by 30%. Both of these reductions were significant ($P<0.05$), but the incremental amount (13%) removed by Zone 2 alone was not significant (Tukey’s multiple comparison test, as shown by underlines in Table 7). These estimates of removal were based on log-transformed data, effectively giving equal weight to the percent (rather than absolute) reduction in each storm. The percent reduction correlated positively ($r=0.53$, $P<0.05$) with the TSS concentration entering the buffer, suggesting that the removal efficiency increased as the sediment load increased (although the actual sediment load was not quantified). Thus the effective removal efficiencies may be higher than the estimates given here.

Table 7. 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). The numbers in parenthesis correspond to the lower and upper 95% confidence limits of the geometric mean. Means with a common underline were not significantly different (P>0.05, Tukey's test).

	Above Zone 3 (Field)	Above Zone 2 (Grass)	Below Zone 2 (Forest)
Total Suspended Solids	104.9	73.2	59.9
	(64 – 172)	(45 – 120)	(39 – 92)
Nitrate-N	0.099	0.081	0.261
	(0.056 – 0.176)	(0.052 – 0.127)	(0.183 – 0.372)
Ammonia-N	0.057	0.035	0.077
	(0.039 – 0.082)	(0.025 – 0.048)	(0.057 – 0.105)
Soluble Reactive Phosphorus (SRP)	0.34	0.30	0.43
	(0.23 – 0.51)	(0.21 – 0.43)	(0.34 – 0.55)
Particulate Phosphorus	0.30	0.22	0.23
	(0.21 – 0.43)	(0.16 – 0.30)	(0.17 – 0.33)
Chloride	2.03	1.82	1.87
	(1.44 – 2.85)	(1.40 – 2.36)	(1.46 – 2.41)

As described above (Methods), the overland collectors reached capacity with some frequency so that overland flow volumes, and hence potential infiltration, could not be estimated. Infiltration can account for a large fraction of the total sediment entrapment within vegetated filter strips (e.g., Schmitt et al. 1999, Borin et al. 2005). However, it seems unlikely that infiltration was significant at the Stroud Preserve RFBS site. As a riparian buffer, near the stream, it is located in soils (aquic fragiuldults) that may have less infiltration capacity than the upland typic hapludults that supply the overland flow (although the grass and forest vegetation may substantially compensate for this). Of potentially more significance, the area of the buffer (~1 ha) is far less than the supplying area of upland tillage (~14 ha). Finally, the swale of the level spreader fully detains the overland flow from smaller storms and assures that overland flow through Zones 2 and 1 occurs only after a substantial, saturating rain has already occurred. For these reasons, it is assumed that the percent concentration reduction estimates are also reasonable estimates of the percent mass removal of sediments by the buffer. As a caveat to this inference, overland flow from the field during small storms was entirely trapped by the swale in Zone 3, and the effect of this trapping, on an annual basis, would add to the removal estimates reported here.

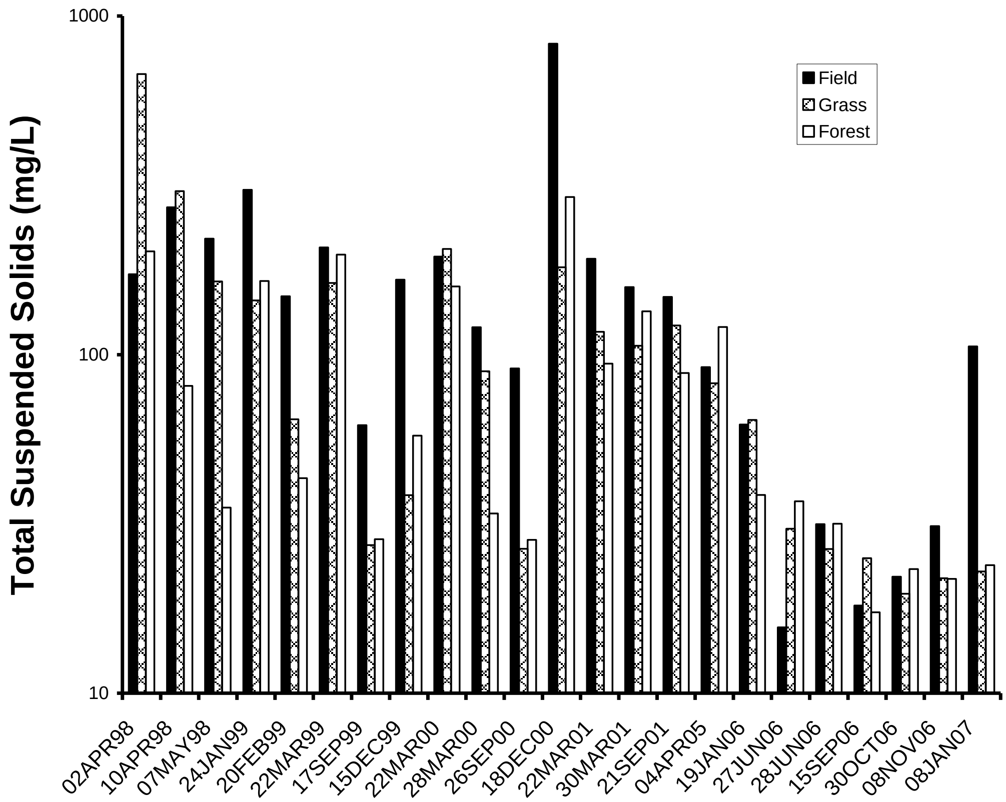

Figure 17. Suspended solids in overland flow from storms collected from 1998 to 2007 in the Morris Run treatment watershed.

Nitrate concentration in overland flow did not change significantly in Zone 3, but increased in Zone 2 to a concentration averaging 2.6 times higher than that entering Zone 3 ($P<0.05$). Nitrate concentrations in the overland flow, however, were very low, averaging <0.1 mg/L in the water transported from the field and 0.26 mg/L in the water leaving Zone 2, still well below typical (baseflow) stream concentrations of >3 mg/L. Thus the addition of nitrate to overland flow contributed negligibly to annual watershed exports of nitrogen. This conclusion is supported by the annual export estimates (see Appendix) which show that stormwater export of nitrate accounted for <7% of total annual nitrate export. The RFBS had a positive (though not significant) overall effect on ammonia-N concentration. Ammonia concentrations were also very low (<0.1 mg/L) and did not affect annual nitrogen exports. SRP did not change in Zone 3, but increased in its passage through Zone 2 to concentrations averaging about 26% higher than those entering either Zone 3 or Zone 2 ($P<0.05$). Particulate phosphorus concentration declined by 22%

across the whole buffer ($P<0.05$), but did not change significantly in Zone 2. The removal of particulate phosphorus roughly matched the release of dissolved phosphorus yielding no net effect on total phosphorus. Chloride in overland flow was not significantly altered by the RFBS.

The apparent lack of removal by the buffer of total (dissolved plus particulate) phosphorus contrasts with other reports of high (~75%) removal of total phosphorus from overland flow in reforested buffers (Clausen et al. 2000, Vellidis et al. 2003). Similarly, Meals and Hopkins (2002) reported 20-50% reductions in total P exports in response to streambank fencing. In this study, there was no net phosphorus removal partly because the entrapment of particulate phosphorus was small (22%), and partly because dissolved phosphorus was released. The low rate of particulate removal may be in part attributable to unmeasured upslope removal in the grass waterways. Phosphorus has been shown to adsorb preferentially to smaller particles, while larger particles are preferentially deposited over short travel distances. Thus the particulate phosphorus that reached the buffer, after traversing the grassed waterways, may have been largely associated with the remaining clay-sized particles that then passed through the buffer with relatively little deposition. Release of dissolved phosphorus by riparian buffers has been observed in a number of studies (e.g., Dillaha et al. 1989a, Peterjohn and Correll 1984, McKergow et al. 2003), although it typically is less than the particulate removal.

As noted above, the overland flow collectors measured concentrations rather than overland fluxes. To convert the removal to an estimate of the mass of suspended solids removed, we used the annual average storm-export of TSS from the watershed of 208 kg ha^{-1} y^{-1}. (Total annual exports are presented below. The partitioning of total export into baseflow and storm exports is presented in the previous annual reports and appears in Appendix B. Assuming that 43% of the total quantity of TSS leaving the field was removed, the 208 kg ha^{-1} y^{-1} of storm export represents 57% of this total which, then, is 365 kg ha^{-1} y^{-1}. The estimated quantity sequestered by the buffer would therefore be 157 kg ha^{-1} y^{-1} or 2340 kg/y deposited in the buffer. This estimate is clearly speculative, but it can be compared to a similarly rough estimate of the accumulation of sediments within the swale of the level spreader. Sediments in the spreader were not surveyed or sampled directly, but measurements of the soil elevation were recorded on a staff gage in the swale whenever the swale was dry at the time of biweekly sampling. Between 1994 and 2007, the soil elevation in the swale increased by 0.22 m, equating to a volumetric accumulation of 1.5 m^3/y. Assuming a bulk density of 1.27, this equates to 1900 kg/y. The previous estimate of 2340 kg/y removal refers to the entire buffer. The proportion inferred to trapped in Zone 3 would be 70% (30%/43%), or 1630 kg/y. Thus the accumulation estimated from the concentration reductions (1630 kg/y) agrees reasonably well with the accumulation estimated from the filling of the swale (1900 kg/y). An additional 4400 m^3 of sediments will fill the swale to capacity, i.e., to the level of the spreader. At the estimated rates, capacity will be reached in 70-85 years.

Annual exports of sediments and phosphorus

Streamwater nutrients and suspended solids were sampled intensively from roughly 8 storms per year between 1997 and 2001 and again in 2005 through early 2007. Previous annual reports describe the dynamics of nitrate, ammonium, total and dissolved phosphorus, and suspended solids during storms and present annual export estimates for each of these constituents. These results showed that, because streamwater nitrate concentration declined greatly during storms, and because ammonium concentration was at all times far lower than nitrate concentration, more than 90% of the export of inorganic nitrogen from the watersheds occurred as nitrate during baseflow. These baseflow nitrate exports were presented and discussed in previous sections and the stormflow dynamics and exports of nitrogen are not further considered here. However, the majority of exports of both phosphorus and suspended solids occurred during stormflows and the results of storm sampling for these constituents are presented in this section.

This aspect of the study began five years after the RFBS was established in the Morris Run watershed. At that time, the trees planted in the RFBS remained small and the reforested area (Zone 2) was effectively a grass buffer. Mine Hill Run, in the reference watershed, had a relatively wide, brushy buffer that changed little throughout the study. Thus, comparisons of export estimates from the RFBS to the reference watershed over the ten year period (1997-2006) cannot quantify the overall effectiveness of the buffer. Instead, this component of the study was intended to address the question of whether the RFBS increased in effectiveness as the initially grassy area of Zone 2 converted, through tree growth, to a young forest.

As described fully in the first annual report (Newbold et al. 1998), annual exports were estimated by first developing an annual rating curve for each constituent that related concentration to streamflow. Then the rating curve was applied to the flow record for the respective year to compute and sum individual export estimates for each 15-min interval of the hydrograph record. Two storms sampled early in 2007 were included in the 2006 rating curve because a complete export year could not be estimated for 2007.

Annual exports of sediments, estimated as exports of TSS, were similar in the treatment and reference streams, averaging 296 kg $ha^{-1}y^{-1}$ in Morris Run (RFBS) and 312 kg $ha^{-1}y^{-1}$ in Mine Hill Run (Fig. 18). Annual exports from Morris Run appeared generally to decline between 1997 and 2006, but this decline was mirrored by the exports from the reference stream, and neither decline was significant ($P>0.05$, Spearman rank correlation test). Thus, there was no apparent effect of forest growth on the export of sediments between 1997 and 2006. This result is not in conflict with the sediment removal rate observed in overland flow, which represents an average measured over nearly the same 1997-2007 period. Rather, it suggests that the removal that was observed has not yet been significantly affected by the transition of Zone 2 from grass to young forest. The RFBS forest, as of 2006, was 15 years old and growing vigorously, but remained far from mature. The soil had not yet developed a litter layer characteristic of a forest. Thus, changes in sediment sequestration rates may occur in the future, over a span of several decades.

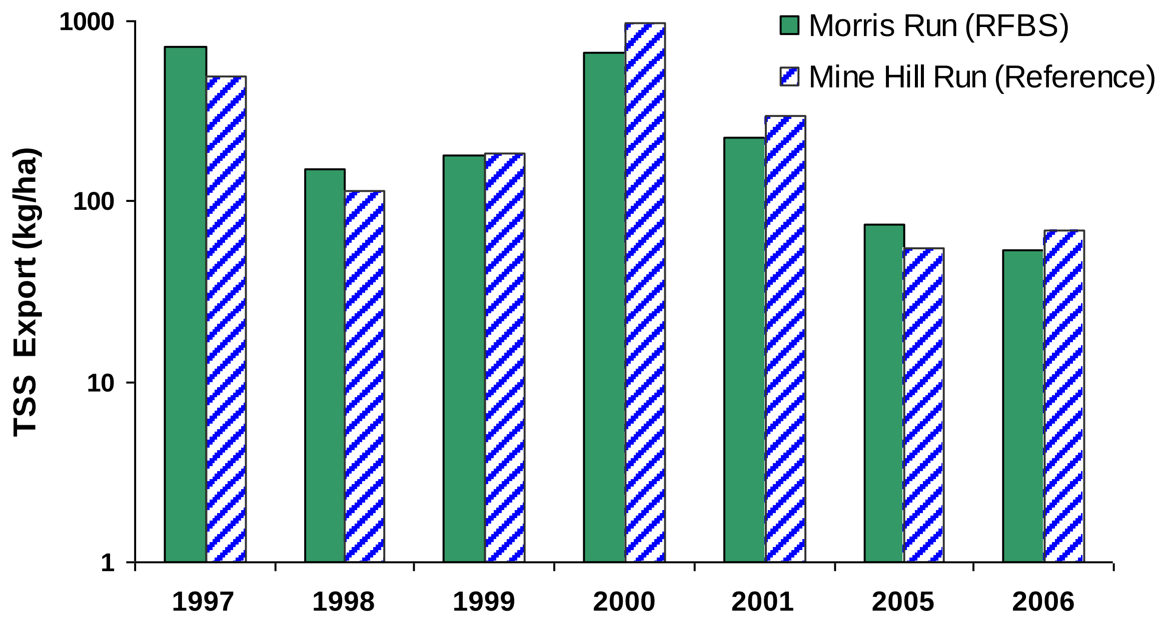

Figure 18. Annual export of TSS from Morris Run (riparian buffer) and Mine Hill Run (reference) watersheds. Exports were not monitored 2002 to 2004.

Exports of total phosphorus (Fig. 19) averaged 0.59 kg $ha^{-1}y^{-1}$ in Morris Run (RFBS) and 0.39 kg $ha^{-1}y^{-1}$in Mine Hill Run (reference), and were higher in Morris Run than in Mine Hill Run in each of the seven years. As noted above, this comparison does not evaluate buffer effectiveness because the reference stream had a pre-existing buffer of brushy vegetation. If the growing forest in the RFBS increased the buffer's effectiveness, we would expect to see a decline in phosphorus exports between 1997 and 2006, in Morris Run relative to the exports from Mine Hill Run. As Fig. 19 illustrates, there was not a clear trend. Exports from the RFBS in 2005 and 2006 were lower than all other years except 1998. However, the exports from the reference stream were also low in 2005 and 2006, so that the decline cannot be attributed to the RFBS. As noted in the preceding discussion of sediment exports, the forest remained relatively young as of 2006 and may affect phosphorus exports more strongly as it matures.

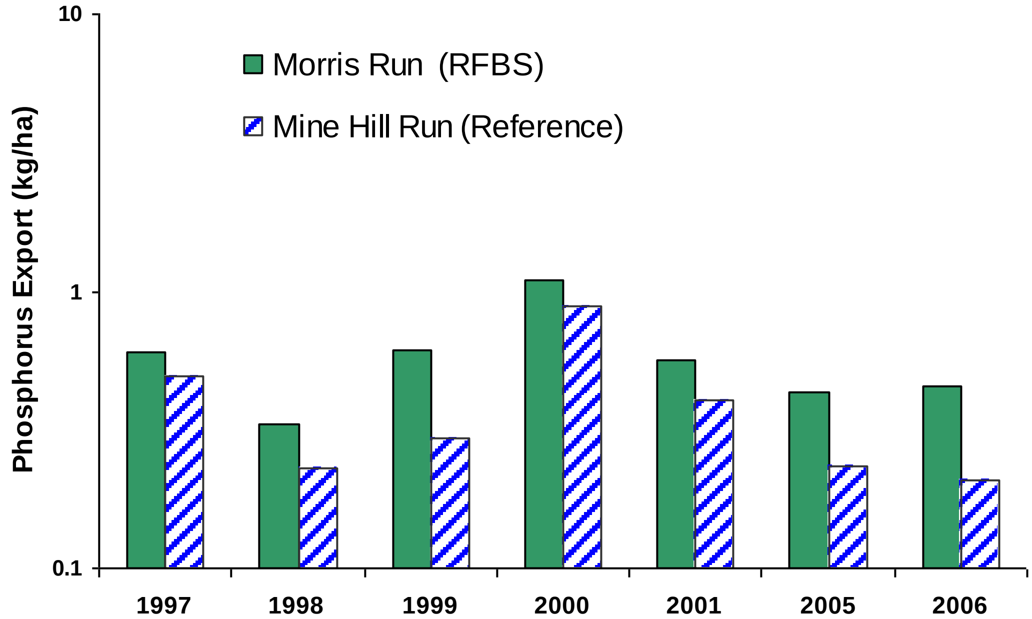

Figure 19. Annual exports of total phosphorus from Morris Run (RFBS) and Mine Hill Run (reference). Exports were not monitored in years 2002 to 2004.

Summary and Conclusions

A reforested riparian zone was established in an agricultural field in the Mid-Atlantic Piedmont in 1992. This study found that a 35-m wide 3-zone riparian forest buffer system removed 26% of the subsurface nitrate and 43% of the suspended sediments delivered from upslope. Total phosphorus was not removed by the buffer. The influence of tree growth on nitrate removal became apparent approximately ten years after planting. The grass filter strip between the forest and the cultivated field, contoured to disperse concentrated overland flow into the reforested area, also 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).

<u>Success in meeting objectives</u>

The results of this project are perhaps best summarized in terms of the initial objectives repeated here for convenience:

(1) Evaluate the non-point source reductions of the Riparian Forest Buffer System (RFBS) in the relatively high-relief terrain of the Mid-Atlantic Piedmont.
(2) Assess the time required after reforestation to achieve significant sediment and nutrient reductions.
(3) Establish specific guidelines for planting and managing forest buffers zones in the Mid-Atlantic region.

Objective 1: How much reduction of nonpoint source pollutants was achieved by the RFBS?

The 3-zone riparian forest buffer system (RFBS), occupying a total width of approximately 32 m (100-110 feet), removed 26% of nitrate entering the buffer from upslope tilled field via subsurface flow. This reduction was measured over a 10 year period beginning 5 years after buffer establishment. The mass rate of nitrogen removal over this period was 69 kg N per hectare of riparian buffer per year (Fig. 11), or 5.0 kg N per hectare of tilled land. Over the same period, the RFBS removed 43% of the suspended sediment from overland flow delivered from the tilled field. The mass of sediment removed was estimated to be 2340 kg per hectare of riparian buffer per year, or 168 kg per hectare of tilled land per year.

The study found no significant net removal of phosphorus from upslope sources. Although the RFBS did remove 22% of particulate phosphorus delivered from tilled field in overland flow, it released a compensating quantity of dissolved phosphorus. Groundwater phosphorus dynamics suggested that the buffer reduced subsurface flow of phosphorus from the field to the stream, but streamwater phosphorus concentrations remained higher than near-stream groundwater concentrations, so effects on subsurface flow remain inconclusive. Streamwater phosphorus concentrations at baseflow declined throughout the study but a parallel decline in the reference stream leaves this result, again, inconclusive.

A recent meta-analysis of nitrogen removal by riparian buffers (Mayer et al. 2007) reported an average removal rate of 72% for forested buffers, substantially higher than found in the present study. High removal rates have typically been observed in settings where the subsurface flow is constrained to shallow pathways rich in organic carbon and/or in contact with the root zone (Peterjohn and Correll 1984, Simmons et al. 1992, and Hill et al. 2000). In contrast, buffers may be relatively ineffective where water flows through deeper pathways to reach the stream (Böhlke and Denver 1995, Vidon and Hill 2004). In the relatively high relief Piedmont setting of the present study, it is likely that flow is preferentially constrained to the shallow saprolite, but that flow through the underlying fractured bedrock is significant (Rose 1992). This mix of shallow and deep pathways may explain the modest rates of nitrogen removal found in this study.

Given that this study found a lower rate of nitrogen removal than the average of 72% cited above, the question arises as to whether the Stroud Preserve site was atypical and hence understates the removal that might generally be expected from the implementation of an RFBS. As noted above, a large number of studies have demonstrated that while subsurface removal rates of nitrogen can be very high, the conditions under which such high rates occur are relatively restrictive. In particular, sites that both have the conditions necessary for high nitrogen removal (either by denitrification or root uptake) and receive significant delivery via the groundwater of nitrogen from upslope sources may be relatively localized, occurring as patchy "hotspots" within a watershed (e.g., Vidon and Hill 2004, Hill et al 2000). The great majority of studies that have shown high nitrogen removal rates in riparian zones, regardless of vegetation, have been based on the analysis of well transects, similar to those of the present study. Very few, however, have been conducted in gaged watersheds where it was possible to relate nitrate concentrations in the groundwater passing along the transects to concentrations and exports of streamwater. Where the groundwater flows cannot be related to watershed exports it is very difficult if not impossible to verify that the processes observed in the transect represent those affecting the bulk of the water that reaches the stream. Of the studies cited by Mayer (2007), there are only three that relate the groundwater studies to watershed export: Lowrance et al. 1984, Peterjohn and Correll (1984), and Vellidis et al. (2003). In each case, the nitrogen removal rate was high (>75%), but all three studies involved relatively wide (38-85 m) buffers and reported within-buffer areal rates of nitrogen removal (40-82 kg ha^{-1} y^{-1}) comparable to those of this study (69 kg ha^{-1} y^{-1}). These rates are only somewhat higher than rates typical of natural riparian forests (Lowrance et al. 1995) but far below the highest rates reported from sites with ideal conditions for denitrification (e.g., Fustec et al. 1991: 475 kg ha^{-1} y^{-1}, Pinay and Decamp 1988: 194 kg ha^{-1} y^{-1}, Hoffmann et al. 2006: 340 kg ha^{-1} y^{-1}). In fact, if nitrogen removal rates are typically in the range of 40-80 kg ha^{-1} y^{-1}, then to achieve removal rates of greater than, for example, 50% would require either buffers substantially in excess of 30 m, or upslope nitrogen loads lower than are typically observed in agricultural settings. These considerations suggest that the nitrogen removal rate of 26% found in this study may be a realistic estimate for RFBS implemented in agricultural watersheds of the Mid-Atlantic Piedmont.

The sediment removal rates we observed, while substantial, were lower than the rates approaching or exceeding 90% reported by several other studies of riparian forest buffers (e.g. Peterjohn and Correll 1984, and Lee et al. 2003). Other conservation measures practiced on our study site may offer a partial explanation. The cropped area upslope of the 3-zone RFBS followed a conservation plan with contours and crop rotation including grasses, which is designed to minimize soil losses to tolerable levels. Overland flow reached the buffer only after leaving contoured strips and traversing grassed waterways which themselves may have removed much of the filterable sediments (Feiner and Aurswald 2003).

Objective 2: How much time is required after reforestation to achieve significant nutrient and sediment reduction?.

Some of the water quality benefits of the RFBS appeared early in the study, soon after reforestation and before any effects of a growing forest could have been realized. Reduction in streamwater nitrate concentration was observed in 1993, one year after the RFBS was established. This essentially immediate improvement is interpreted as the result of terminating fertilizer applications near the stream. Similarly, the ability of the riparian buffer to remove sediments from overland flow appears to have begun prior to 1997 when overland flow monitoring was initiated. Most of the sediment removal occurred in the grass-vegetated Zone 3, probably through deposition in the swale formed by the level spreader. The level spreader was constructed in 1994 and, after establishment of a grass turf, its function as a sediment trap would not be expected to have changed appreciably (except that some studies show a decline in buffer effectiveness over time because of clogging, but these were plot studies like Dillaha 1989). Between 1997 and 2007 there was no significant trend in the export of sediments from the RFBS, relative to that in the reference watershed (except that 2005 and 2006 sediment export were lower than export in beginning of study at RFBS).

There is strong, but not conclusive, evidence that the nitrate removal by the riparian buffer increased substantially beginning ten years after buffer establishment, corresponding to the onset of rapid tree growth within the reforested Zone 2 of the RFBS. It is difficult to separate the influence of the growing forest from nitrogen removal that occurred during seedling establishment (i.e., prior to rapid growth) because meaningful annual mass balance estimates of removal could not be made. Such annual estimates were prevented by the multi-year travel time of groundwater through the buffer together with large annual variations in the inputs of groundwater nitrate from the tilled field. Nonetheless, the sharp decline in streamwater nitrate that began in 2002 was most likely the result of tree growth. The alternative explanation, that decline was the result of reduced upslope inputs, is not consistent with the timing of that decline, which should have reduced streamwater concentrations until about 2005 or later.

Objective 3: What guidelines for establishing and maintaining forested riparian buffers arose from this study?

The experience of this study suggests guidelines in two major areas: (1) survival and growth of the planted trees; and (2) design of level spreader and management of overland flow.

Tree survival and growth—The RFBS at the Stroud Preserve suffered from excessive tree mortality and slow growth through the first several years (1992-1999), some of which was attributable to drought conditions, but much of which might have been averted. The initial tree planting and maintenance followed some practices that undoubtedly improved success and are recommended. These included (1) the use of tree shelters which were shown in another study to greatly enhance both survivorship and growth in the first several years (Sweeney et al. 2002); (2) the use of a mix of native hardwood species, some of which offer higher growth potential while others are more resistance to herbivore damage (Sweeney et al. 2004); and (3) annual inventory of survivorship and replacement of dead trees.

Another practice—herbicide control of competing vegetation—was not practiced until 1998, and this may have been the single most important factor behind the early mortality and slow growth. Sweeney et al. (2002) showed that herbicide (glyphosate) application to individual trees, increased survivorship approximately 2.5-fold over control by mowing, and approximately 50% over control by tree mats. Thus herbicide control is strongly recommended.

Deer browse became important in stunting growth after 2-5 years when trees emerged from the 4-foot tree shelters that were initially used. Replacement with 5-foot shelters largely eliminated this problem and, at sites where deer browse may be high, 5-foot shelters are recommended. As trees reached diameters of over 5-10 cm (2-4 inches) a significant number were lost to rubbing by deer. The plastic tree shelters were insufficient to protect against this damage. Because of this problem, the plastic shelters were replaced by 5-foot wire mesh (3-inch mesh) protectors as trees reached this size. The latter practice eliminated the rubbing problem and is recommended.

Level-spreader design and maintenance--Most of the area in the grassed Zone 3 of the RFBS was occupied by the level spreader. The spreader proved to be an important component of the buffer system, demonstrating two major functions. First, it collected concentrated surface runoff from the grass waterways of the tilled fields and dispersed it, allowing the runoff to enter the reforested Zone 2 as uniformly distributed sheet flow. This function was the original design purpose. Second, the spreader, together with its associated upslope swale, performed a major function in trapping sediments and particulate phosphorus. Although it was anticipated that the grassed Zone 3 would remove suspended sediments and phosphorus from overland flow, its relative contribution to total removal (74% and effectively 100% of the total sediment and particulate phosphorus removal that occurred in the RFBS as a whole) was unexpected.

The dispersal function of the level spreader was, for the most part, successful. However, the experience of the project demonstrated that it is critical to construct and maintain uniform elevation of the spreader along its entire length. It further demonstrated the importance of constructing the spreader on the original contour with minimal use of fill. Beginning in 2004, ten years after construction, it was noted that overland flow from the spreader had begun to concentrate in two areas where settling of the spreader surface was evident. The concentrated flow exacerbated gully erosion and the formation of a head cut at the point of origin of perennial flow in the stream. A survey of the spreader confirmed that the spreader had subsided in these locations by as much as 0.4 foot (0.12 m). Further, the subsidence was largely localized to the limited areas where the spreader had been constructed of fill to a level above original grade. In June of 2006, the spreader level was restored to a tolerance of ±0.05 feet (0.015 m) and reseeded in native grasses.

Lessons Learned

The experience of implementing and evaluating the RFBS at the Stroud Preserve provided a number of "lessons learned" that fall roughly into two categories: (1) RFBS installation and maintenance, and (2) monitoring project design and execution. Lessons from the first category resulted in the guidelines described in the previous section and so are only summarized here.

RFBS installation and maintenance

- The buffer should be planted with a mix of native hardwoods that provide a range of growth potential, resistance to deer damage, and adaptation to less well-drained riparian soils.
- The planting site for each tree should be cleared of competing vegetation and be kept free of competing vegetation for several years.
- Tree shelters are recommended to enhance early growth and survivorship. The shelters should be five feet high where deer browsing may be a problem. Where deer pressure is severe and bark-stripping may occur, wire mesh tree protectors should be considered.
- Tree survival should be monitored annually, with replacement of excess mortality.
- To minimize gully erosion, it is important that the forested portions of the RFBS (Zones 1 and 2) be protected from concentrated overland flow delivered from grass waterways or natural swales. A level-lip spreader, located in the grass portion of the buffer (Zone 3) is recommended to convert the concentrated flow to sheet flow as it enters the buffer.
- It is important that the spreader remain level indefinitely. This can be facilitated by constructing the spreader along the original contour, keeping the spreader lip at the level of the contour.

Design and execution of the monitoring program

- Many years may be required to observe the effects of a BMP improvement. This lesson is not new and, in the case of the Stroud Preserve RFBS, the extended time involved tree growth, which was well understood in advance. What was learned was that, where the time-rates of change are unknown at the outset, the long term monitoring plan should be designed to adapt flexibly to the emerging time line.
- Estimates of riparian buffer removal of nutrients from subsurface flow must account for time lags in subsurface flow pathways. This lesson was not well understood at the outset, as the time lags were unknown, and a number of previously published studies had estimated buffer removal from relatively short term (1-2 years) analyses in which steady state is assumed. The 3-4 lag time, observed in the present study, between upslope inputs and streamwater response was inconsistent with the assumption of steady state and led to the use of a much longer averaging time (12 years) for the calculation of a subsurface budget.
- All human activities within the paired watersheds, with the exception of BMP implementation, should be maintained with as little change as possible. In this study, the ability to detect an effect of the growing riparian forest on streamwater nitrate concentrations was seriously limited by large temporal variations that resulted, at least partially, from changes in fertilization rates. Farming operations during this study were

monitored, but not controlled. The paired watershed design substantially compensated for the temporal variations, but could not remove their effects entirely.

- Multiple monitoring strategies can be complementary and be of great benefit. The project used two approaches in evaluating the effect of the RFBS, one based on paired-watershed monitoring, the other on mass balance within the treatment watershed. The mass balance study, particularly the groundwater monitoring, proved critical in explaining the paired-watershed trends. Conversely, the paired-watershed study yielded information regarding temporal lags that improved the assessment of mass balance.

State and local application of results

The project targeted both professionals involved in development of nonpoint source control strategies and the public at large. Over the life of the project, it has hosted site visits for groups of professionals involved in local conferences, workshops, and training sessions. Typically, there have been several such field tours each year. A description of The Stroud Preserve Reforested Riparian Buffer and results of this study, including a downloadable brochure are posted on the Stroud Water Research Center website at http://www.stroudcenter.org/research/StroudPreserve/index.htm.

The results of this project have provided support and guidance to the PA Department of Environmental Protection (PA-DEP) for the development of both regulations, under Pennsylvania's Clean Streams Law, and technical guidance, concerning the protection of riparian zones and the implementation of the 3-zone RFBS concept. The regulations and guidance remain in development as of the writing of this report (February 2009).

The project has also had substantial educational impact. It receives considerable exposure, through the Stroud Water Research Center's educational program, which reaches thousands of students and adults annually. The Stroud Preserve, and the RFBS demonstration also receive regular field visits from classes in ecology and hydrology taught at nearby West Chester University. One Master's thesis (Alberts 2000) and one Ph.D. thesis (Watts 1997) were written as part of this project.

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APPENDIX A

Stroud Preserve Riparian Buffer Brochure

http://www.stroudcenter.org/research/projects/StroudPreserve/StroudPreserveBufferBrochure_Oct2007.pdf

Project Background

The Stroud Riparian Reforestation Project was initiated in 1991 as a demonstration of the 3-zone Riparian Forest Buffer System developed by the U.S. Forest Service. Previous research had shown trees nearest the stream protect habitat by controlling water temperature, providing leaf litter as a food supply for aquatic organisms, and stabilizing stream banks.

Other research showed that a somewhat wider zone of streamside forest can filter out agrochemicals and eroded soil that would otherwise reach the stream. The Forest Service's 3-zone buffer concept integrated this research into a strategy for reforesting streams in agricultural areas to protect water quality while maintaining agricultural productivity.

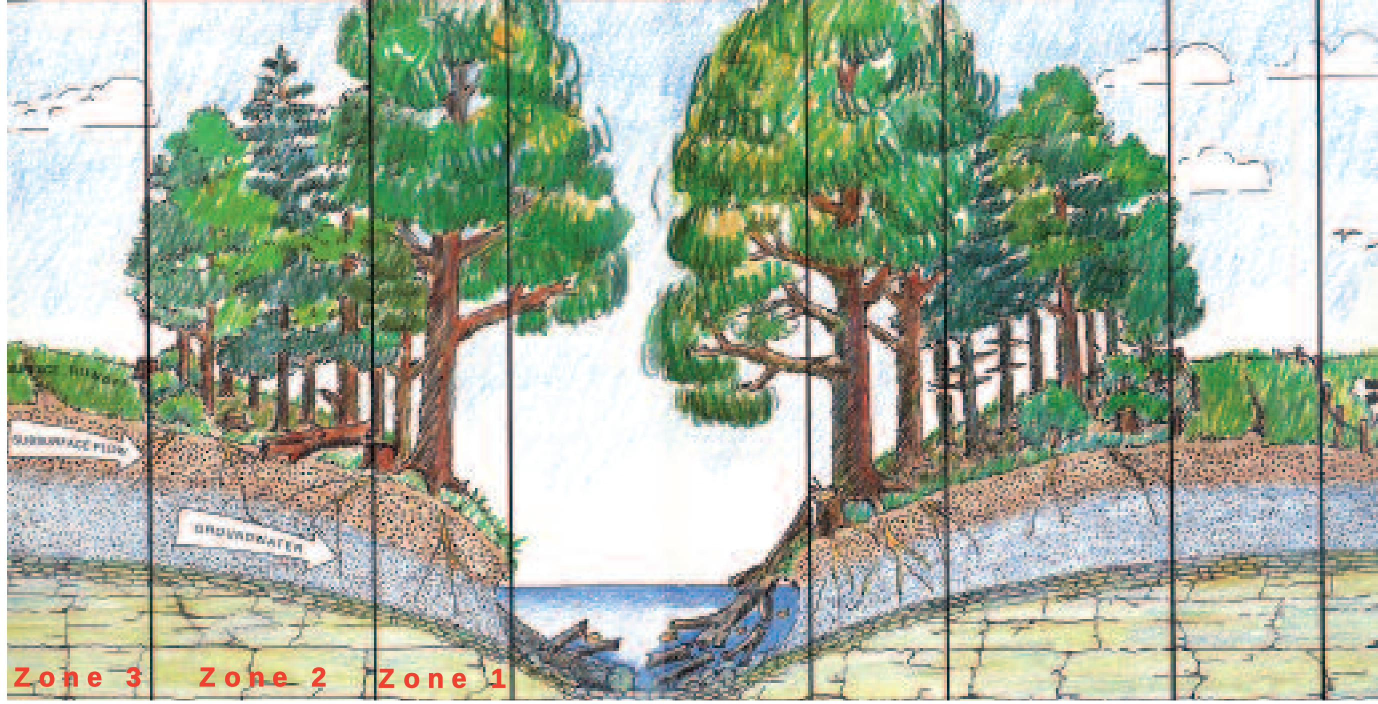
The Natural Land Trust's Stroud Preserve, near West Chester, PA, provided an ideal site for implementing the first example of the 3-zone system and collecting the long term data necessary to evaluate its effectiveness.

Project goals

- Demonstrate the use of the 3-zone Riparian Forest Buffer System developed by the U.S. Forest Service.
- Demonstrate the ability of streamside—or riparian—reforestation to improve water quality
- Assess the time needed to achieve full benefit of restoration
- Establish guidelines for riparian buffer planting, maintenance and management
- Transfer lesson learned to the general public, land-use professionals and the research community.

More information

www.stroudcenter.org/research/StroudPreserve/index.htm



The 3-zone Riparian Forest Buffer System

Zone 1— protects stream habitat. Undisturbed mature forest extending at least 15 feet from the streambank. At the Stroud Preserve, this was in place prior to 1992.

Zone 2—filters nutrients and sediments. At least 60 ft of forest that can be managed for timber production.

Zone 3—disperses concentrated runoff. At least 20 feet of non-forested buffer contoured to spread overland flow into sheet flow before it enters Zone 2. At the Stroud Preserve, a level-lip spreader was constructed in this zone.

The level-lip spreader

Overland flow during storms often runs off crop and pasture lands as concentrated flow in grass waterways. A forested buffer, although capable of filtering sediments from overland flow, is vulnerable to erosion from this concentrated flow. A level spreader intercepts the concentrated flow and spreads it out so that it flows evenly as thin sheet flow into the forest.

At the Stroud Preserve, a level spreader was constructed in 1994. The berm, or level-lip, over which the water flows lies at the original contour. The hillslope above the contour was excavated to provide a narrow, extended basin that diverts and distributes the water so that it flows uniformly over the level "lip" or contour.

The Reforestation…

Zone 2 was planted in 1992 with seedlings of red oak, white ash, tulip poplar, sugar maple, black walnut, trembling aspen, sycamore, and river birch.
Tree growth was initially delayed by drought and deer-damage. Beginning in 1998 more aggressive measures were instituted to assure vigorous forest development. These included annual herbicide (glyphosate) treatment of each tree, gradual replacement of plastic tree shelters with wire mesh tree enclosures, and replanting critical gaps with larger balled and burlapped trees. Since 1999 the trees have grown rapidly and the total basal area has increased more than tenfold.

1992

1999

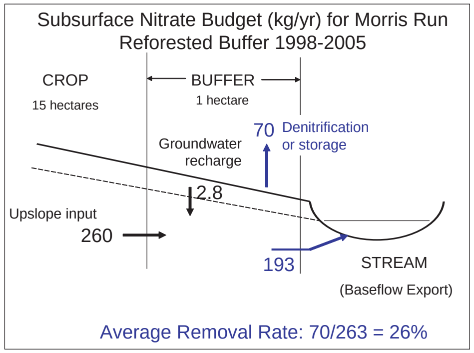
2005

What the project is monitoring

The project monitors water quality (primarily nitrogen, phosphorus, and suspended sediments) in the streams and in overland flow during storms. Nitrogen and phosphorus are also measured in the groundwater at varying distances from the stream.

What the project has shown

- the riparian buffer removes an average of 27% of nitrate and 52% of the sediment that would otherwise reach the stream
- the level-lip spreader minimizes erosion within the reforested area by dispersing concentrated surface runoff from the cropland into sheet flow as it enters the reforested area
- successful establishment of the riparian forest buffer required control of non-woody vegetation and protection of trees from deer damage
- plastic tree shelters followed by 5-foot wire-mesh cages around individual trees have proved the most effective means to protect saplings from deer damage



Project Support

Since 1997, the project has been a US-EPA National Monitoring Program Project,funded by the Pennsylvania Department of Environmental Protection and the US EPA through Section 319(h) of the federal Clean Water Act. Other support has come from: USDA Forest Service, Pennsylvania Department of Conservation and Natural Resources Bureau of Forestry, Chesapeake Bay Program, Pennsylvania State Bureau of Forestry, USDA Natural Resource Conservation Service, Stroud Foundation, Pennswood No.2 Research Endowment, and the Stroud Endowment for Environmental Research.

STROUD
Water Research Center

Protecting Water Quality and Stream Habitat…

The Stroud Preserve Reforested Riparian Buffer

A U.S.-EPA National Monitoring Program Project

APPENDIX B.

Annual water, nutrient, and suspended solids exports

Exports are presented for Years 1998-2001 and 2005,2006. No storm samples were taken in 2002-2004 and so export estimates were computed. Tables A1-A10 were compiled from the project Annual Reports (Newbold and Sweeney 1999, 2001a, 2001b, 2002, 2006). Tables A11 and A12, representing year 2006 are newly presented here.

Table A1 Water, nutrient and sediment exports for Morris Run 1998.

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			0.92
Streamflow (m)	0.19	0.036	0.226
Nitrogen (kg/ha)			
Nitrate-N	8.4	0.32	8.8
Ammonia-N	0.017	0.019	0.036
Dissolved Organic Nitrogen	0.252	0.25	0.502
Sum of Nitrogen Exports:	8.7	0.59	9.3
Phosphorus (kg/ha)			
Total Dissolved Phosphorus	0.06	0.10	0.15
Particulate Phosphorus	0.030	0.146	0.175
Sum of Phosphorus Exports:	0.09	0.24	0.33
Total Suspended Solids (kg/ha)	11	141	152

Table A2 Water, nutrient and sediment exports for Mine Hill Run for 1998.

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			0.92
Streamflow (m)	0.20	0.04	0.24
Nitrogen (kg/ha)			
Nitrate-N	7.37	0.41	7.78
Ammonia-N	0.03	0.02	0.05
Dissolved Organic Nitrogen	0.24	0.24	0.48
Sum of Nitrogen Exports:	7.64	0.68	8.31
Phosphorus (kg/ha)			
Total Dissolved Phosphorus	0.047	0.025	0.072
Particulate Phosphorus	0.043	0.117	0.160
Sum of Phosphorus Exports:	0.09	0.14	0.23
Total Suspended Solids (kg/ha)	27	86	113

Table A3. Water, nutrient and sediment exports for Morris Run 1999.

	Baseflow	Stormflow All Storms	Stormflow Floyd	Total
Water				
Precipitation (m)				1.36
Streamflow (m)	0.161	0.124	0.042	0.286
Nitrogen (kg/ha)				
Nitrate-N	7.07	1.56	1.23	8.63
Ammonia-N	0.017	0.033	0.012	0.050
Dissolved Organic Nitrogen	0.19	0.57	0.28	0.76
Sum of Nitrogen Exports:	7.3	2.16	1.52	9.46
Phosphorus (kg/ha)				
Total Dissolved Phosphorus	0.06	0.24	0.13	0.30
Particulate Phosphorus	0.02	0.29	0.07	0.31
Sum of Phosphorus Exports:	0.08	0.53	0.20	0.61
Total Suspended Solids (kg/ha)	10	170	53	180

Table A4. Water, nutrient and sediment exports for Mine Hill Run for 1999.

	Baseflow	Stormflow All Storms	Stormflow Floyd	Total
Water				
Precipitation (m)				1.36
Streamflow (m)	0.150	0.105	.028	0.253
Nitrogen (kg/ha)				
Nitrate-N	5.47	1.85	0.96	7.32
Ammonia-N	0.02	.05	.02	.07
Dissolved Organic Nitrogen	0.15	0.65	0.001	0.66
Sum of Nitrogen Exports:	5.64	2.55	0.98	8.19
Phosphorus (kg/ha)				
Total Dissolved Phosphorus	0.034	0.053	0.015	0.087
Particulate Phosphorus	0.018	0.190	0.066	0.208
Sum of Phosphorus Exports:	0.052	0.243	0.081	0.295
Total Suspended Solids (kg/ha)	9	175	72	184

Table A5. Water, nutrient and sediment exports for Morris Run 2000.

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			1.08
Streamflow (m)	0.26	0.08	0.346
Nitrogen (kg/ha)			
Nitrate-N	12.12	0.46	12.58
Ammonia-N	0.03	0.03	0.05
Dissolved Organic Nitrogen	0.23	0.51	0.73
Sum of Nitrogen Exports:	12.4	0.99	13.4
Phosphorus (kg/ha)			
Total Dissolved Phosphorus	0.22	0.22	0.45
Particulate Phosphorus	0.02	0.63	0.65
Sum of Phosphorus Exports:	0.24	0.86	1.10
Total Suspended Solids (kg/ha)	8	654	662

TableA6. Water, nutrient and sediment exports for Mine Hill Run for 2000.

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			1.08
Streamflow (m)	0.23	0.10	0.33
Nitrogen (kg/ha)			
Nitrate-N	9.21	1.19	10.40
Ammonia-N	0.03	0.04	0.07
Dissolved Organic Nitrogen	0.33	0.55	0.89
Sum of Nitrogen Exports:	9.57	1.80	11.36
Phosphorus (kg/ha)			
Total Dissolved Phosphorus	0.06	0.04	0.10
Particulate Phosphorus	0.03	0.76	0.78
Sum of Phosphorus Exports:	0.09	0.80	0.89
Total Suspended Solids (kg/ha)	17	957	974

Table A7. Water, nutrient and sediment exports for Morris Run 2001.

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			0.873
Streamflow (m)	0.179	0.068	0.247
Nitrogen (kg/ha)			
Nitrate-N	9.01	0.39	9.40
Ammonia-N	0.02	0.02	0.04
Dissolved Organic Nitrogen	0.15	0.47	0.62
Sum of Nitrogen Exports:	9.2	0.88	10.1
Phosphorus (kg/ha)			
Total Dissolved Phosphorus	0.15	0.15	0.29
Particulate Phosphorus	0.01	0.26	0.27
Sum of Phosphorus Exports:	0.159	0.408	0.567
Total Suspended Solids (kg/ha)	5	219	224

Table A8. Water, nutrient and sediment exports for Mine Hill Run for 2001.

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			0.873
Streamflow (m)	0.197	0.048	0.245
Nitrogen (kg/ha)			
Nitrate-N	7.76	0.75	8.52
Ammonia-N	0.03	0.02	0.05
Dissolved Organic Nitrogen	0.22	0.23	0.45
Sum of Nitrogen Exports:	8.01	1.00	9.02
Phosphorus (kg/ha)			
Total Dissolved Phosphorus	0.051	0.015	0.066
Particulate Phosphorus	0.024	0.313	0.337
Sum of Phosphorus Exports:	0.075	0.328	0.403
Total Suspended Solids (kg/ha)	19	279	299

TableA9. Water, nutrient and sediment exports for Morris Run 2005

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			1.1
Streamflow (m)	0.343	0.083	0.426
Nitrogen (kg/ha)			
Nitrate-N	12.23	0.23	12.46
Ammonia-N	0.02	0.02	0.04
Sum of Inorganic Nitrogen Exports:	12.25	0.25	12.5
Phosphorus (kg/ha)			
Dissolved Ortho-Phosphate	0.14	0.14	0.27
Particulate Phosphorus	0.022	0.141	0.164
Sum of Phosphorus Exports:	0.159	0.278	0.437
Total Suspended Solids (kg/ha)	25	48	73

Table A10. Water, nutrient and sediment exports for Mine Hill Run for 2005

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			1.1
Streamflow (m)	0.318	0.071	0.389
Nitrogen (kg/ha)			
Nitrate-N	10.57	0.68	11.25
Ammonia-N	0.04	0.02	0.06
Sum of Inorganic Nitrogen Exports:	10.60	0.70	11.30
Phosphorus (kg/ha)			
Dissolved Ortho-Phosphate	0.092	0.022	0.114
Particulate Phosphorus	0.025	0.095	0.121
Sum of Phosphorus Exports:	0.118	0.117	0.235
Total Suspended Solids (kg/ha)	16	38	55

Table A11. Water, nutrient and sediment exports for Morris Run 2006

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			1.13
Streamflow (m)	0.319	0.077	0.396
Nitrogen (kg/ha)			
Nitrate-N	10.10	0.28	10.38
Ammonia-N	0.03	0.03	0.05
Sum of Inorganic Nitrogen Exports:	10.1	0.31	10.4
Phosphorus (kg/ha)			
Soluble Reactive Phosphorus (SRP)	0.08	0.25	0.33
Particulate Phosphorus	0.041	0.078	0.119
Sum of Phosphorus Exports:	0.124	0.327	0.450
Total Suspended Solids (kg/ha)	23	31	55

Table A12. Water, nutrient and sediment exports for Mine Hill Run for 2006

	Baseflow	Stormflow	Total
Water			
Precipitation (m)			1.13
Streamflow (m)	0.258	0.067	0.325
Nitrogen (kg/ha)			
Nitrate-N	8.37	0.58	8.95
Ammonia-N	0.03	0.05	0.09
Sum of Inorganic Nitrogen Exports:	8.40	0.63	9.03
Phosphorus (kg/ha)			
Dissolved Ortho-Phosphate	0.07	0.03	0.100
Particulate Phosphorus	0.022	0.085	0.107
Sum of Phosphorus Exports:	0.092	0.115	0.207
Total Suspended Solids (kg/ha)	12	58	70