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Surface Runoff Water Quality in a Managed Three Zone Riparian Buffer

Richard Lowrance* and Joseph M. Sheridan

ABSTRACT

Managed riparian forest buffers are an important conservation practice but there are little data on the water quality effects of buffer management. We measured surface runoff volumes and nutrient concentrations and loads in a riparian buffer system consisting of (moving down slope from the field) a grass strip, a managed forest, and an unmanaged forest. The managed forest consisted of sections of clear-cut, thinned, and mature forest. The mature forest had significantly lower flow-weighted concentrations of nitrate, ammonium, total Kjeldahl N (TKN), sediment TKN, total N (nitrate + TKN), dissolved molybdate reactive P (DMRP), total P, and chloride. The average buffer represented the conditions along a stream reach with a buffer system in different stages of growth. Compared with the field output, flow-weighted concentrations of nitrate, ammonium, DMRP, and total P decreased significantly within the buffer and flow-weighted concentrations of TKN, total N, and chloride increased significantly within the buffer. All loads decreased significantly from the field to the middle of the buffer, but most loads increased from the middle of the buffer to the sampling point nearest the stream because surface runoff volume increased near the stream. The largest percentage reduction of the incoming nutrient load (at least 65% for all nutrient forms) took place in the grass buffer zone because of the large decrease (68%) in flow. The average buffer reduced loadings for all nutrient species, from 27% for TKN to 63% for sediment P. The managed forest and grass buffer combined was an effective buffer system.

Both grass buffers (vegetated filter strips) and forest buffers are increasingly used as conservation practices to control nonpoint-source pollution from agriculture. These conservation practices are based on numerous studies that directly measured the water quality effects of the practice or a set of practices. The earliest studies of buffers stressed either effects of simple grass buffers on surface runoff nutrients (Dillaha et al., 1989; Magette et al., 1989) or studied shallow subsurface movement of dissolved nutrients, especially nitrate, for naturally occurring buffers (Jacobs and Gilliam, 1985; Lowrance et al., 1983, 1984; Peterjohn and Correll, 1984). More recently, knowledge of vegetated filter strips and riparian forest buffer systems has been advanced through more detailed studies in various parts of the USA and through studies of combined grass and forest buffers (Daniels and Gilliam, 1996; Hubbard et al., 1998; Lee et al., 2000, 2003). In addition, new information is available on the water quality impacts of newly established and managed buffer systems (Clausen et al., 2000; Hubbard et al., 1998; Vellidis et al., 2003; Lee et al., 2000, 2003; Lowrance et al., 2000b). Unlike some of the earliest studies of complex buffers, all of these studies reported on some aspect of surface runoff nutrient removal.

There are still very few studies that measure the effectiveness of either vegetated filter strips or riparian forest buffers under natural rainfall conditions at a scale appropriate to represent management units realistically. Clausen et al. (2000) studied nutrient transport and developed N budgets for a restored fescue (*Festuca* spp.) buffer in Connecticut. They found that loads and concentrations of nitrate-N, total Kjeldahl N (TKN) and total P were reduced in runoff compared with the control, which was an unrestored riparian cornfield. Verchot et al. (1997) found that on North Carolina Piedmont sites, forested buffers might be either sources or sinks of nutrients in surface runoff. The forest buffers were ineffective during the winter and spring when water-filled pore space exceeded 25 to 35% and infiltration was low. Infiltration was the key factor controlling N pollutant removal from surface runoff. Therefore, buffers in the clayey soils of the Piedmont may not be as effective as sandy coastal plain soils (Verchot et al., 1997). Daniels and Gilliam (1996) found that combined grass and riparian forest filters reduced runoff loads of nutrients by 50 to 80%. The reduction in the chemical load depended on the nutrient and its form. Filters reduced total P load by 50%, but 80% of the soluble DMRP arriving at the field edge frequently passed through the filters. The filters retained 20 to 50% of the ammonium-N and approximately 50% of the TKN and nitrate-N. High-volume flows commonly overwhelmed both grass and riparian filters next to cultivated fields. Forested ephemeral channels had little vegetation and were effective sediment sinks during the dry season but were ineffective during large storm events because there was little resistance to flow (Daniels and Gilliam, 1996).

This study was a test of the three zone buffer system proposed as a USDA practice by Welsch (1991) and Lowrance (1991). The three zone buffer consists of a grass buffer (Zone 3) adjacent to the crop field; a managed forest (Zone 2) where trees can be clear-cut or thinned; and a permanent forest (Zone 1) where only selective harvesting of trees to correct drainage problems is allowed. The USDA–Natural Resources Conservation Service (NRCS) practice standards provide for this combination of vegetated filter strips and riparian forest buffer at the edge of field where control of nutrient and sediment movement to streams is needed. Although the three zone buffer system is based on scientific principles developed from studies of mature buffers, it has received few tests under field conditions. The studies reported here provide one of the first tests of surface runoff nutrient control by managed buffers of a scale

Southeast Watershed Research Lab., 2379 Rainwater Road, Tifton, GA 31794. Received 26 July 2004. *Corresponding author (Lorenz@tifton.usda.gov).

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677 S. Segoe Rd., Madison, WI 53711 USA

Abbreviations: DMRP, dissolved molybdate-reactive phosphorus; GFS, Gibbs Farm site; LIFE, Low Impact Flow Event sampler; TKN, total Kjeldahl nitrogen; TVU, Tifton–Vidalia Upland.

and complexity typical of real-world conditions. We studied the performance of a grass filter strip and a down slope managed riparian forest buffer under natural rainfall conditions and along an entire stream reach that encompassed three management treatments for the forested buffer. This study was designed to provide information on both concentrations and loads of N, P, and chloride in direct surface runoff moving through a managed riparian buffer system. This is a companion study to previously published studies from the same site on sediment and water transport (Sheridan et al., 1999); subsurface hydrology (Bosch et al., 1994, 1996); herbicide transport (Lowrance et al., 1997); subsurface nutrient and chloride transport (Hubbard and Lowrance, 1997; Lowrance et al., 2000b); soil ecology (Lowrance, 1992; Ettema et al., 1999a, 1999b); and model testing (Inamdar et al., 1999a, 1999b; Lowrance et al., 2000a).

The specific objectives of this study were to (i) determine the effects of harvest of a part of the mature riparian forest on the movement of N, P, and chloride in surface runoff; (ii) determine the spatial variability of N, P, and chloride movement in surface runoff in a grass filter strip, a mature riparian forest, and a managed riparian buffer; and (iii) determine the concentrations and loads of N, P, and chloride in surface runoff in a three zone buffer system managed according to USDA-NRCS practice standards.

MATERIALS AND METHODS

Gibbs Farm Study Site

The study was done at a research farm (Gibbs Farm Site, GFS), which is part of the University of Georgia Coastal Plain Experiment Station near Tifton, GA. The GFS is located in the Tifton–Vidalia Upland (TVU) portion of the Gulf–Atlantic

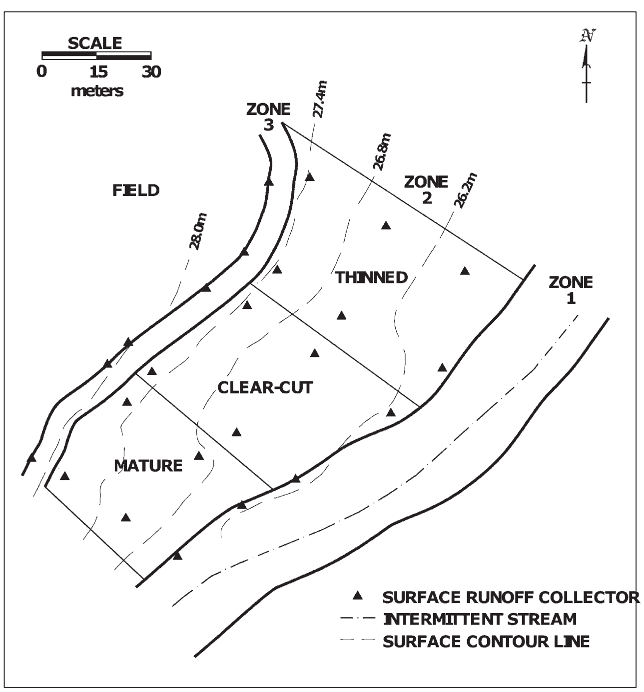

Fig. 1. Gibbs Farm Site showing location of surface runoff collectors; buffer Zones 3, 2, and 1; and location of Zone 2 treatments.

Coastal Plain. The climate of the TVU is humid subtropical with about 120 cm of annual rainfall and a long growing season. Because of both less permeable soil material at depth and the presence of a geologic formation (Hawthorn Formation), which limits deep recharge to the regional aquifer system, most of the excess precipitation in the TVU moves either laterally in shallow saturated flow or moves in surface runoff during storm events. The typical hydrology of the region is reflected at the GFS.

The GFS is a hillside with a 1.1-ha cultivated field draining into approximately 0.9 ha of riparian forest. A second-order intermittent stream drains the site. The cultivated field had an average slope of 2.5% and the average distance from the field to the stream was 75 m. The soil of most of the GFS riparian forest is an Alapaha loamy sand (fine-loamy, siliceous, acid, thermic Typic Fluvaquents). The soil of the adjacent upland area is a Tifton loamy sand (fine-loamy, siliceous, thermic, Plinthic Kandiudult). The upland soil extends approximately 10 m into the buffer system and included the grass buffer established for this study. Although permeabilities of the Alapaha and Tifton soils are similar, the Alapaha soil has a high water table for much of the year while the Tifton soil does not.

A three zone riparian buffer system was established at an existing riparian forest site for this research project in 1992 (Fig. 1). The upper part of the site at the field edge was steeper than the lower part of the site near the stream (Fig. 1). The site extended 120 m across the hillside (perpendicular to the slope). The buffer consisted of three zones. Zone 3 was an 8 m wide strip of common bermudagrass [*Cynodon dactylon* (L.) Pers.] and bahiagrass (*Paspalum notatum* Flugge.). The grass strip was interplanted with perennial ryegrass (*Lolium perenne* L.) during its establishment. Zone 2 (before timber harvest) was a 45- to 60-m wide band of slash pine (*Pinus elliottii* Engelm.) and long leaf pine (*Pinus palustris* Mill.). Zone 1 was a 15-m wide band of trees with mostly hardwoods including yellow poplar (*Liriodendron tulipifera* L.) and swamp black gum (*Nyssa sylvatica* var. *biflora* Marsh.). The entire buffer averaged 75 m in width (range 68–83 m) along an intermittent second-order stream channel. The distance across the site was divided into three equal 40-m sections in which the Zone 2 forests received different treatments (Fig. 1). In early November 1992, one section of Zone 2 forest was clear-cut and one section was selectively cut (thinned) to one-half of the original tree basal area. A third Zone 2 forest block was left as a mature forest (control) area (Fig. 1). The mature forest of Zone 2 and all of Zone 1, with average tree ages of about 50 yr, were considered to be in a steady state condition with very little net increase in biomass. The timber harvest was done with a feller-buncher equipped with floatation tires. After harvest, all branches greater than approximately 2.5 cm (1 inch) diameter were removed from the harvested sites. Any branches <2.5 cm diameter were redistributed by hand within the plot to provide a relatively uniform cover of debris. There was limited rutting of the plots and no intentional soil–litter disturbance such as occurs when branches and other debris are windrowed. The harvest was done very carefully to limit increases to spatial variability in the harvested sections. The clear-cut Zone 2 was replanted with improved slash pine in winter of 1993 and naturally occurring vegetation was allowed to grow with no attempt at control. No seedlings were planted in the thinned Zone 2 area. The timber harvest practices are typical of BMPs applied in riparian zones except for the absence of windrowed debris and attention to minimizing soil disturbance. It is likely that our experimental forest harvests caused much less disturbance than typical harvests. All Zone

3 and Zone 1 areas received uniform treatment throughtout. No timber was harvested from any of the Zone 1 areas.

The field above the buffer system on the west side of the stream was in continuous corn (*Zea mays* L.) for the first 3 yr of this study (1992–1994). In 1995, the field was planted in peanut (*Arachis hypogea* L.). In 1996, the field was planted in millet (*Pennisetum glaucum* L). All crops were grown using conventional tillage and conventional fertilizer and pesticide treatments. Fields were disk-harrowed and mold-board plowed for all the crops. The plowed fields were bedded for the peanut crop. Rows were oriented at an angle to the upslope edge of the buffer system.

Sample Collection and Handling

Surface runoff was collected from December 1992 through December 1996 using the Low Impact Flow Event sampler (LIFE sampler; Sheridan et al., 1996, 1999). Two types of LIFE samplers were used to collect either 10 or 1% of the flow through a 30.5-cm wide "dustpan"—a collection apron mounted flush with the soil surface. The 10% collection was made by splitting the flow into 10 pathways at the back of the collector and collecting the flow from one pathway. The 1% sample was collected by connecting two 10% samplers in series. The water flowed into a buried sample receptacle made from a 1 m long piece of 10-cm diameter PVC pipe with capped ends. One of each type sampler was located at each of four positions in the buffer. The positions were defined by the zonal interfaces (six samplers per zonal interface) (Fig. 1). In addition, six samplers were located in the middle of Zone 2.

Surface runoff samples were collected, volumes were measured, and subsamples collected for nutrient analysis on the work-day following each rainfall event. Samples from all collectors that had volumes >100 mL were used for each surface runoff event. Samples were taken in chemically clean glass bottles with Teflon-lined caps. Samples were collected by pumping the receptacles with a peristaltic pump while agitating the sample by mixing with the inlet line of the pump. Samples were stored in coolers in the field and then transported to lab refrigerators (4°C) within 2 h of collection.

In the lab, samples were filtered through Whatman 934 AH filters for determination of suspended sediment (Sheridan et al., 1999). An aliquot of the filtrate was stored for dissolved nutrient analysis. In addition an aliquot of the unfiltered sample was stored for analysis of TKN and total P in a digestate. The filtered sample was analyzed for nitrate-N, ammonium-N, dissolved molybdate-reactive P (DMRP), and chloride using USEPA approved colorimetric techniques (Clesceri et al., 1998) on a Lachat Flow Injection Analyzer. Both the filtered and unfiltered sample were analyzed for TKN and total P using digestion and colorimetric techniques adapted from USEPA-approved methods (Clesceri et al., 1998). The TKN and total P sediment fractions were calculated by subtracting filtered concentrations from unfiltered concentrations for a sample. Total N was calculated as the sum of unfiltered TKN and nitrate-N.

Data Analysis

Flow-weighted concentrations and unit area loads were calculated from the flow volumes and the laboratory data on concentrations. Flow-weighted concentrations were calculated for each collector and event based on the (Event concentration × Event volume)/Total volume for the collector for the entire study. The sums of these event flow-weighted concentrations are the mean flow-weighted concentrations for the entire study. Loads were calculated for each collector and event as Concentration (mg L^{-1}) × Volume (converted to L m^{-1} of collector edge). Loads were summed for the entire study and converted to units of g m^{-1}. The total load changes within the overall buffer were used to estimate the percentage load reduction by Zones 3 and 2 of the managed buffer system. The runoff water enters the buffer at Position 1, so this is the entering load. Load reductions were calculated as the [(Position 1 load − Downslope load)/Position 1 load] × 100. The load reduction for the entire buffer was calculated as [(Position 1 load − Position 4 load)/Position 1 load] × 100.

Data were tested for normal distribution using the Univariate Procedure of the Statistical Analysis System (SAS Institute, 1999). The concentration data were not normally distributed, so typical analysis of variance was not used. Instead, the NPAR1WAY procedure of SAS with the Kruskal-Wallis test was used. The NPAR1WAY procedure is a nonparametric procedure that tests whether the distribution of a variable has the same location parameter across different groups. The Kruskal-Wallis procedure tests the null hypothesis that the groups are not different from each other by testing whether the rank sums are different based on a Chi-square distribution (Sokal and Rohlf, 1981). Data were analyzed to determine if there were differences among positions within a treatment (mature, clear-cut, or thinned) and differences among treatments within a position. Data were also analyzed to determine if there were differences among positions for all data pooled.

Data will be presented both for positions within the individual treatments and for the entire riparian buffer. Although there were differences both within the treatments and as the water entered the buffer system, the overall average concentrations and sums of loads provide an understanding of the entire buffer system. This is particularly relevant to the management of buffers along streams because—on a given stream reach—the forest buffer managed according to USDA-NRCS practice standards would typically be in various stages of growth from immediately post clear-cut to mature. The average concentrations and sums of loads are the values that could be expected from this average buffer. All samples are reported based on their position within the buffer. The four landscape positions are: Position 1, field edge (water entering Zone 3, the grass buffer); Position 2, entering Zone 2 (after water has moved through the grass buffer); Position 3, middle Zone 2 (after water has moved through half of the Zone 2 forest buffer); and Position 4, entering Zone 1 (after water has moved through all of the Zone 2 buffer). Entering Zone 1 was as close to the stream channel as samplers could be located because Zone 1 was typically inundated during high stream flow events several times a year. Therefore, the samples collected in this study do not reflect the final filtering that takes place in the Zone 1 portion of the buffer.

RESULTS AND DISCUSSION

Flow-Weighted Concentration Differences among Treatments and among Positions

There were significant differences in flow-weighted concentrations both among treatments within a position and among positions within a treatment for all N species, for all P species, and for chloride (Tables 1 and 2). Concentrations exiting the field above the buffer were significantly different in different parts of the field. Concentrations of ammonium, total N, total P, and sediment P exiting the field were significantly lower above the thinned treatment (Position 1). The DMRP was significantly lower above the clear cut treatment. No concen-

Table 1. Flow-weighted concentrations of N species in surface runoff. All values are actual N (mg N L^{-1}). Values are means followed by number of observations and standard error. Total N = Nitrate + TKN unfiltered.

	Treatment														
	Nitrate			Ammonium			TKN unfiltered			Sediment TKN			Total N		
Landscape position	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned
Position 1	NS†1.56‡	1.66‡	1.15‡	§1.41NS	1.70NS	1.16‡	NS3.55	3.43‡	3.66‡	NS1.23	0.92NS	1.28‡	§4.86NS	4.91‡	4.72‡
(Field edge)¶	(181, 0.37)	(142, 0.34)	(191, 0.16)	(176, 0.24)	(140, 0.44)	(193, 0.17)	(169, 0.38)	(132, 0.41)	(175, 0.49)	(130, 0.24)	(100, 0.17)	(139, 0.34)	(166, 0.56)	(127, 0.65)	(173, 0.57)
Position 2	NS1.22	2.09	1.60	NS1.26	1.35	1.71	NS3.58	3.14	4.93	NS1.01	1.05	1.26	NS3.93	4.93	6.48
(Entering Zone 2)¶	(129, 0.25)	(117, 0.60)	(115, 0.38)	(131, 0.46)	(117, 0.36)	(114, 0.56)	(125, 0.80)	(107, 0.60)	(109, 1.15)	(104, 0.52)	(87, 0.33)	(83, 0.64)	(120, 0.74)	(105, 1.12)	(108, 1.46)
Position 3	§0.83	0.89	1.75	§0.65	1.45	1.02	§2.80	5.22	6.83	§0.79	2.68	0.58	§3.42	6.07	8.49
(Middle Zone 2)	(111, 0.10)	(101, 0.32)	(51, 0.54)	(110, 0.12)	(98, 0.49)	(50, 0.24)	(97, 0.47)	(92, 2.13)	(48, 3.75)	(77, 0.24)	(74, 2.00)	(32, 0.94)	(94, 0.50)	(92, 2.18)	(46, 3.97)
Position 4	§1.20	1.07	0.99	NS1.26	1.28	1.29	§4.81	3.75	5.08	NS2.07	0.80	1.30	§5.55	4.35	6.01
(Entering Zone 1)	(110, 0.27)	(72, 0.27)	(138, 0.23)	(108, 0.39)	(74, 0.52)	(137, 0.29)	(106, 1.24)	(67, 0.63)	(132, 1.08)	(86, 0.96)	(59, 0.24)	(98, 0.79)	(100, 1.37)	(62, 0.67)	(131, 1.14)

† NS, no significant difference among positions (columns) and/or treatments (rows).
‡ Values are significantly different among positions (within a treatment) based on the Kruskal-Wallis test (0.05 level).
§ Values are significantly different among treatments (within a position) based on the Kruskal-Wallis test (0.05 level).
¶ Upslope of Zone 2, not directly affected by Zone 2 treatment.

Table 2. Flow-weighted concentrations of P species, chloride, and sediment in surface runoff. All P values are actual P (mg P L^{-1}). Values are means followed by number of observations and standard error.

	Treatment											
	DMRP			Total P unfiltered			Sediment total P			Chloride		
Landscape position	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned	Mature buffer	Clear cut	Thinned
Position 1	†0.70NS‡	0.67NS	0.69§	†0.87NS	0.84NS	0.79§	†0.23§	0.21NS	0.13§	NS5.46§	5.18§	5.22§
(Field edge)¶	(181, 0.09)	(143, 0.07)	(193, 0.08)	(182, 0.11)	(145, 0.10)	(193, 0.08)	(152, 0.06)	(113, 0.07)	(153, 0.04)	(173, 0.70)	(136, 0.84)	(184, 0.77)
Position 2	NS0.72	0.60	0.87	NS0.78	0.75	1.12	NS0.06	0.17	0.11	NS6.04	5.56	6.01
(Entering Zone 2)¶	(131, 0.13)	(118, 0.09)	(113, 0.24)	(133, 0.14)	(118, 0.12)	(117, 0.29)	(108, 0.01)	(88, 0.07)	(91, 0.05)	(127, 0.65)	(114, 0.97)	(112, 0.90)
Position 3	†0.36	1.20	2.30	†0.49	1.33	2.34	†0.12	0.06	0.29	†6.88	7.12	10.74
(Middle Zone 2)	(114, 0.08)	(98, 0.43)	(50, 1.02)	(114, 0.09)	(100, 0.43)	(51, 1.02)	(87, 0.04)	(78, 0.02)	(36, 0.16)	(108, 1.02)	(97, 1.67)	(52, 2.79)
Position 4	†0.75	0.44	0.50	†0.80	0.64	0.59	NS0.10	0.12	0.13	NS10.39	10.65	11.49
(Entering Zone 1)	(118, 0.18)	(74, 0.09)	(138, 0.08)	(118, 0.18)	(76, 0.15)	(139, 0.09)	(98, 0.04)	(61, 0.06)	(118, 0.04)	(114, 1.67)	(77, 1.76)	(136, 1.38)

† Values are significantly different among treatments (within a position) based on the Kruskal-Wallis test (0.05 level).
‡ NS, no significant difference among positions (columns) and/or treatments (rows).
§ Values are significantly different among positions (within a treatment) based on the Kruskal-Wallis test (0.05 level).
¶ Upslope of Zone 2, not directly affected by Zone 2 treatment.

trations were significantly lower above the mature forest treatment. The differences among treatments at Position 1 reflect the differences in flow-weighted concentrations leaving the field with concentrations generally lower at the corner of the field above the thinned treatment. There were no treatment differences within Position 2 (after the grass buffer) but there were significant treatment differences at Position 3 (middle Zone 2) and Position 4 (entering Zone 1). For Position 3, significantly lower concentrations of nitrate, ammonium, TKN, total N, DMRP, total P, and chloride were found in the mature buffer. Significantly lower sediment N was found in the thinned Zone 2 and significantly lower sediment P was found in the clear cut. Position 4 (entering Zone 1) had significantly lower nitrate, and total P in the thinned and significantly lower TKN, TN, and DMRP in the clear cut. Overall, the main differences among treatments were lower concentrations leaving the field above the thinned Zone 2 (not a treatment effect) and lower concentrations of most nutrients in the mature buffer at Position 3.

Although significant differences occurred for positions within treatments, the differences were not consistent and few generalities can be made except for chloride. Chloride was significantly different among positions for all treatments with a consistent pattern of increasing concentration from Position 1 to 4 (Table 2). Sediment P was significantly lower at Position 2 (entering Zone 2) for both the mature and thinned treatment. The lack of pattern among the positions within the treatments shows the effects of spatial variability both of concentrations in the surface runoff entering the buffer system from the field (Position 1) and within the buffer system. Although the mature buffer had significantly lower concentrations at Position 3 for nitrate, both the clear cut and thinned buffers had significantly lower concentrations of several nutrient species at Positions 3 and 4. Given that the mature buffer had significantly lower concentrations of nitrate, ammonium, TKN, total N, DMRP, and TP at Position 3 (middle of Zone 2), it is possible that if lower concentrations were not entering the thinned and clear cut treatments from the field that there would have been more consistent treatment differences. Largely because of the differences in flow-weighted concentrations leaving the field and because of a very careful tree harvest, position differences within treatments were difficult to detect.

Chloride was significantly different among positions for all treatments (Table 2). Chloride concentrations increased by 5 to 6 mg L^{-1} from Position 1 (field edge) to Position 4 (entering Zone 1). The increase was consistent with an increase in ground water chloride observed at the same site (Lowrance et al., 2000b). Although there are no process studies available to account for the increases in chloride concentrations, speculation has centered on the effects of evapotranspiration to increase the ground water concentration. If this is the reason for the ground water concentration increase, the surface runoff concentration increase could be due to increased ground water seepage contribution (exfiltration) to surface runoff as the water moves down slope from the field edge. If the change in chloride concentrations are due in part to exfiltration, this would be expected to change the concentrations of other constituents as well.

The lack of consistent treatment and position effects was related to high spatial variability but may also be due to the lack of true replication among the treatment blocks. Because of the scale and intensity of the sampling, replicate treatment blocks were not possible. In addition, the number of observations for a treatment position combination ranged widely from a low of 59 to a high of 193. Because of the design of the experiment to capture runoff from natural rainfall events in a real-world multi-zone buffer, there were large differences in the number of samples collected at various points in the landscape.

Concentrations and Loads Averaged across Management Treatments

Real-world buffers along a stream are likely to have various portions in different stages of development. The stages of development could include recent thinning and clear-cut, in addition to mature forest buffer. The buffer could also be receiving different inputs from different parts of the adjacent field. The Gibbs Farm riparian buffer represents these real-world conditions and the average concentrations and loads in this system can be considered representative of the average concentrations and loads passing through a managed Coastal Plain buffer.

There were significant differences among all flow-weighted concentrations with the exception of sediment total N and sediment total P (Fig. 2). Nitrate, ammonium, DMRP, and total P concentrations decreased significantly within the buffer from Position 1 to either Position 3 or 4 (Fig. 2a, 2b, 2f, 2g). Total kjeldahl N and total N increased significantly from Position 1 to Position 4 and chloride concentrations increased consistently throughout the buffer with most of the increase coming from Position 3 to 4 (Fig. 2i).

Trends in concentrations of runoff nitrate, ammonium, and chloride relative to rainfall concentrations are instructive in understanding the processes that occur to produce the observed surface runoff. Although mass balances are not used here, on an annual basis, the volume of rainfall falling in the riparian buffer is similar to the volume of runoff entering (Lowrance et al., 2000a). Mean rainfall concentrations of ammonium, nitrate, and chloride measured at National Atmospheric Deposition Program (NADP) stations (GA50 and GA 99) within 10 km of the Gibbs Farm site for 1992–1996 were 0.24 mg nitrate-N L^{-1}, 0.16 mg ammonium-N L^{-1}, and 0.51 mg chloride L^{-1} (http://nadp.sws.uiuc.edu/nadpdata/; verified 22 June 2005). Discounting the effects of throughfall (the rainfall that comes through the forest canopy) and stemflow (the rainfall that flows down tree trunks), there should have been dilution by rainfall of nitrate-N, ammonium-N, and chloride entering the buffer in surface runoff. If average rainfall and average runoff were totally mixed, the concentrations would be about 0.83 mg nitrate-N L^{-1}, 0.77 mg ammo-

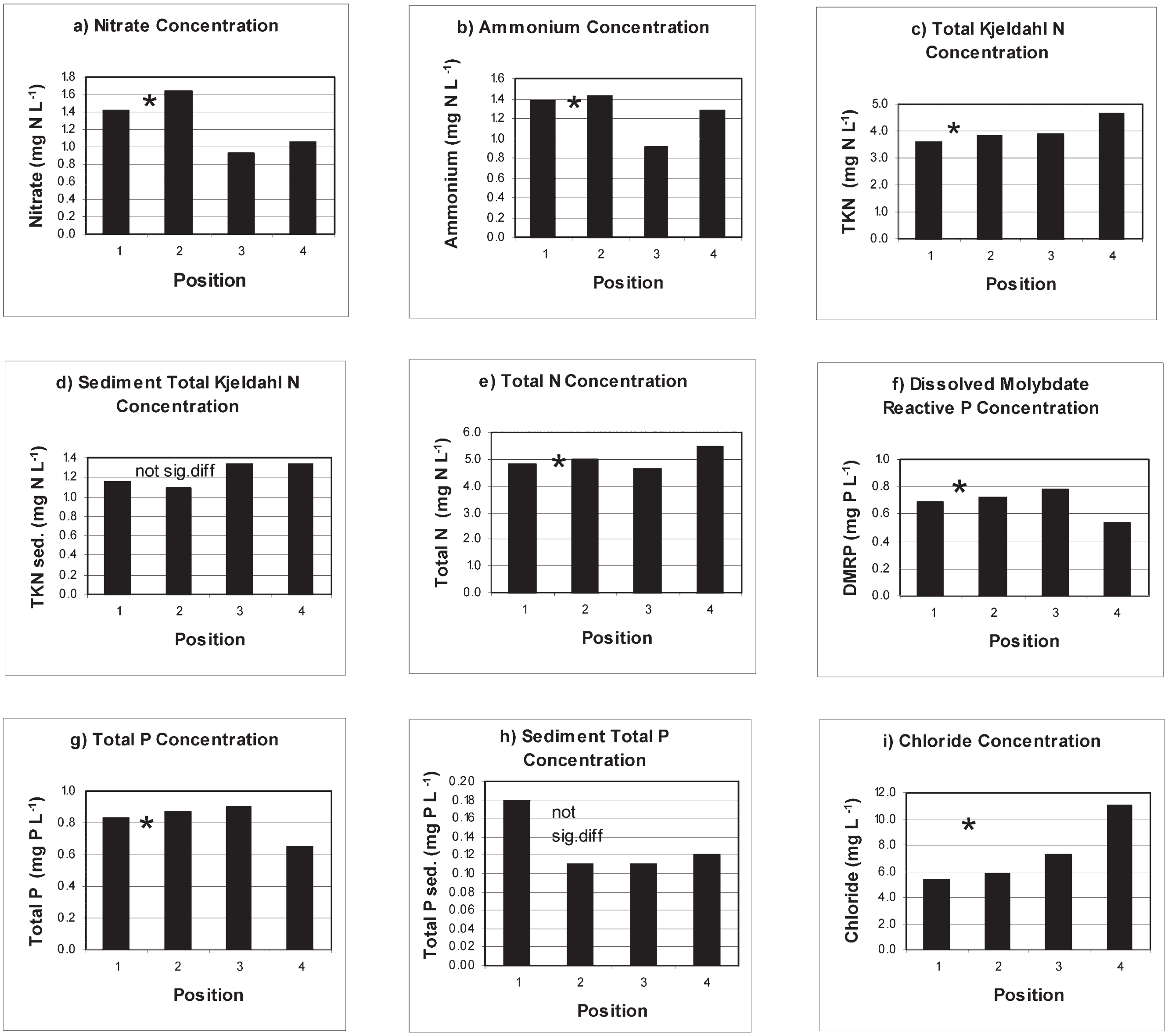

Fig. 2. Flow-weighted concentrations of nutrients in surface runoff by position in the Gibbs Farm riparian buffer. Position 1 is field edge. Position 2 is Zone 3 (grass buffer) downslope edge. Position 3 is middle of Zone 2 (managed forest). Position 4 is the downslope edge of Zone 2 (managed forest). Differences among positions based on rank sums using the Kruskal-Wallis test (0.05 level) are indicated by *.

nium-N L^{-1}, and 2.8 mg chloride L^{-1}. Concentrations in runoff higher than these indicate the mobilization of nutrients in runoff and throughfall/stemflow. Nitrate-N and ammonium-N were similar to the theoretical mixed concentration at Position 3 but increased at Position 4 as the water moved through the remainder of Zone 2 (Fig. 2a and 2b). Chloride presents a special case that will be discussed below because of the significant enrichment that occurs within the buffer (Fig. 2i).

Loadings at each position were controlled by the runoff volume for most nutrients (Fig. 3). All loadings were significantly different among positions (at least the 0.05 level for the Kruskal-Wallis test). Runoff volume decreased from Position 1 (field edge) to Position 2 (entering Zone 2) with a slight increase at Position 3 (middle of Zone 2) as it moved through the grass buffer and the first part of the forest buffer. Runoff increased at Position 4 (entering Zone 1). With the exception of chloride, all loads were lower at Position 4 than Position 1. All loads also increased from Position 3 to 4, showing the dominant influence of the amount of runoff on load calculations. The similarity of the patterns of load changes to the pattern of runoff volume changes across positions reflected the relatively minor concentration changes among positions. As with herbicides in surface runoff at this site (Lowrance et al., 1997), most of the load reduction takes place in the grass buffer, between Positions 1 and 2. Although all loads (except chloride) were reduced in the buffer compared with the edge of field load, the runoff volume increase within the buffer tended to increase the load at Position 4 as the water entered Zone 1.

Trends in chloride concentrations and loads provide insight into the hydrology of the system relative to the

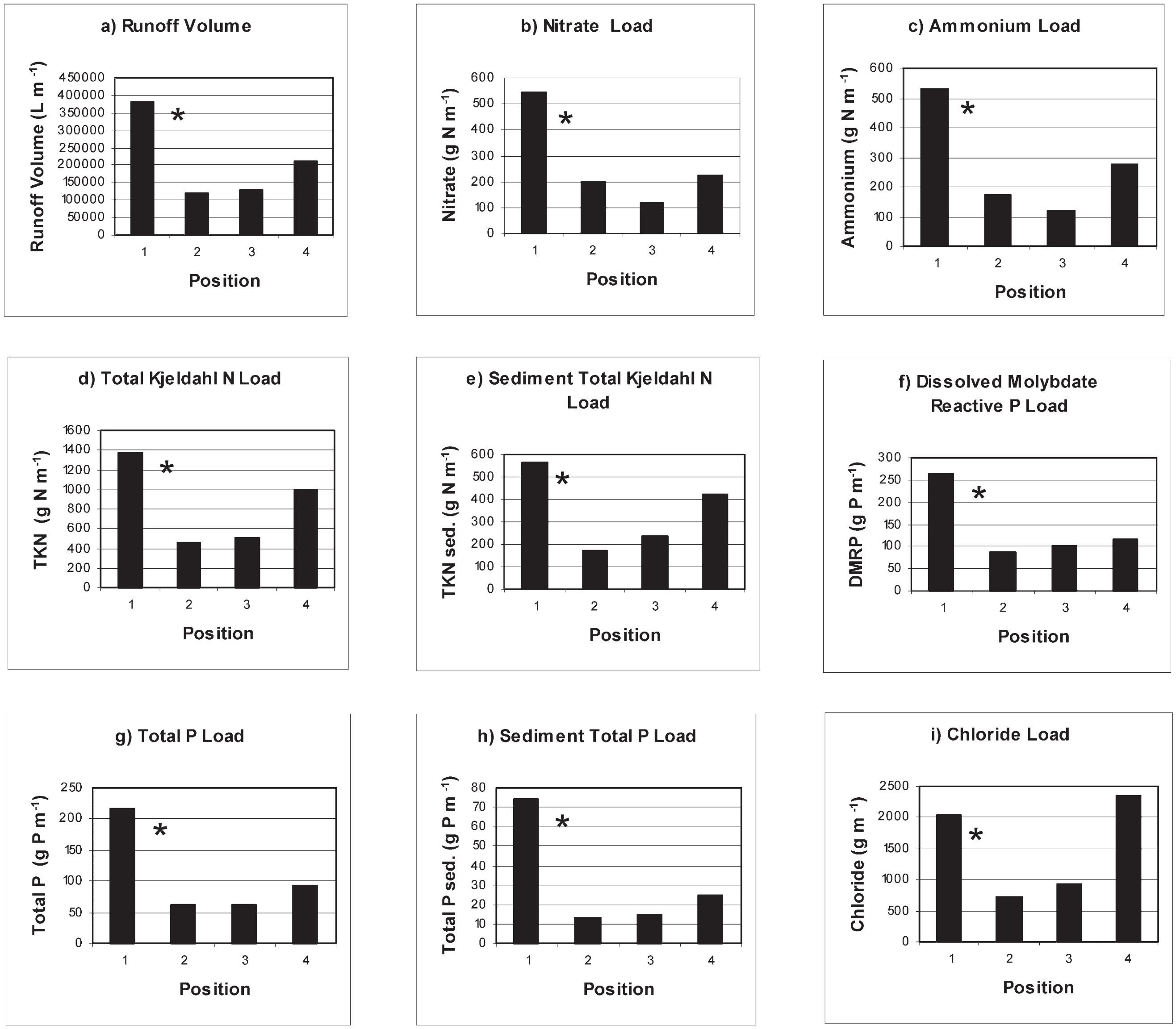

Fig. 3. Total nutrient loads and runoff volumes by position in the Gibbs Farm riparian buffer. Position 1 is field edge. Position 2 is Zone 3 downslope edge. Position 3 is middle of Zone 2. Position 4 is the downslope edge of Zone 2. Total N (nitrate-N + TKN) is not shown. Nutrient loads and water volumes are the cumulative totals for the entire time period of the experiment passing through a 1-m interface at the defined positions. All loads are different among positions based on rank sums using the Kruskal-Wallis test (0.05 level) as indicated by *.

surface runoff measured. Chloride concentrations in rainfall should be providing dilution of the chloride in runoff, but both concentrations and loadings increase. The increase in surface runoff concentration is consistent with an increase in subsurface concentrations in chloride in ground water (Lowrance et al., 2000b). Apparently the increase in surface runoff chloride concentration and load is due to exfiltrating ground water rather than direct surface runoff being generated by rainfall in the buffer. Water that exfiltrated closer to the stream and was caught in Position 4 collectors was depleted in nitrate because nitrate is reduced in shallow ground water moving through the buffer (Lowrance et al., 2000b). Thus, surface runoff near the stream was enriched with chloride more than nitrate because of differences in concentrations in exfiltrating ground water.

The total load changes within the overall buffer (Fig. 3) can be used to estimate the percentage load reduction by Zones 3 and 2 of the managed buffer system (Table 3). The overall buffer had load increases between Positions 2 or 3 and Position 4, largely due to flow increases nearer the stream. Thus, the percentage load reduction between Positions 1 and 4 was always less than the maximum percentage load reduction. The load reduction for the entire buffer was calculated as the difference between Positions 1 and 4. Table 3 shows load reductions between Position 1 and all downslope positions. Load reductions from Position 1 to 4 ranged from 27 to 63%. Maximum reduction generally occurred between Positions 1 and 2 in the grass buffer strip, except for nitrate-N and ammonium-N for which maximum reduction occurred between Positions 1 and 3. Maximum reductions ranged from 65 to 80%. These reductions represent the large amount of filtering through infiltration

Table 3. Percent change in runoff and nutrient load based on differences between incoming load (Position 1) and downslope load (Positions 2, 3, and 4). Reductions calculated as [(Position 1 load − Downslope load)/Position 1 load] × 100. Chloride load increased by 16% from Position 1 to 4.

Parameter	Load or runoff change: Position 1–2	Position 1–3	Position 1–4
		%	
Runoff volume	68	66	44
Nitrate-N	64	78	59
Ammonium-N	67	78	48
Total Kjeldahl N	66	63	27
Sediment total Kjeldahl N	70	61	37
Total N	67	68	37
Dissolved molybdate-reactive P	67	62	56
Total P	67	64	56
Sediment total P	80	79	63
Chloride	65	54	−16

that occurred in Zone 3. Sheridan et al. (1999) found similar sediment load reductions in Zone 3 with about 80% of the entering load deposited in the Zone 3 grass buffer. This was similar to the 80 and 70% reduction in Zone 3 (Positions 1 and 2, Table 3) observed for sediment N and sediment P in this study.

During dry periods, the load reductions between Positions 1 and 4 do not represent the entire retention capacity of the buffer system. The placement of the Position 4 samplers at the beginning of Zone 1 meant that with surface runoff generated under drier conditions due to intense rainfall events, there would be less channel expansion and more possibility for filtering in Zone 1. Conversely, during times of high stream flow, water leaving Zone 2 would be entering an expanded channel that would cover much of Zone 1. Under higher flow conditions, the observed filtering capacity between Positions 1 and 4 would represent the entire function of the buffer.

SUMMARY AND CONCLUSIONS

Although it is possible to understand some of the internal dynamics in buffer system surface runoff water quality with an experimental design of this sort, the inherent variability of the system makes it more reasonable to evaluate overall function of the buffer by comparing edge of field water quality to the quality of water in surface runoff near the stream. Although all loads decreased within the buffer, the increase in runoff volume at Position 4 (entering Zone 1) lead to load increases at this position. The load increase at Position 4 was particularly pronounced for chloride, which also had a concentration increase at Position 4, possibly due to exfiltrating ground water. There were significant differences in the nutrient concentrations and loading entering the buffer system and within the buffer system based on both treatment and position. Some of the apparent treatment effects were due to inherent differences in water moving into the buffer from the field and within the buffer itself. The mature Zone 2 section had significantly lower concentrations at Position 3 (middle of Zone 2) but these differences were not observed at Position 4 (entering Zone 1). Overall, this study showed that complex, managed, three zone buffers can reduce most nutrient loads entering a stream from an upslope field.

REFERENCES

Bosch, D.D., R.K. Hubbard, L.T. West, and R. Lowrance. 1994. Subsurface flow patterns in a riparian buffer system. Trans. ASAE 37:1783–1790.

Bosch, D.D., J.M. Sheridan, and R. Lowrance. 1996. Hydraulic gradients and flow rates of a shallow Coastal Plain aquifer in a forested riparian buffer. Trans. ASAE 39:865–871.

Clausen, J.C., K. Guillard, C.M. Sigmund, and K.M. Dors. 2000. Water quality changes from riparian buffer restoration in Connecticut. J. Environ. Qual. 29:1751–1761.

Clesceri, L.S., A.E. Greenburg, and D.A. Eaton. (ed.). 1998. Standard methods for the examination of water and wastewater. 20th ed. Am. Public Health Assoc., Washington, DC.

Daniels, R.B., and J.W. Gilliam. 1996. Sediment and chemical load reduction by grass and riparian filters. Soil Sci. Soc. Am. J. 60: 246–251.

Dillaha, T.A., R.B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. Trans. ASAE 32:513–519.

Ettema, C.H., R. Lowrance, and D.C. Coleman. 1999a. Riparian soil response to surface nitrogen input: Temporal changes in denitrification, labile and microbial C and N pools, and bacterial and fungal respiration. Soil Biol. Biochem. 31:1609–1624.

Ettema, C.H., R. Lowrance, and D.C. Coleman. 1999b. Riparian soil response to surface nitrogen input: The indicator potential of free-living soil nematode populations. Soil Biol. Biochem. 31:1625–1638.

Hubbard, R.K., and R. Lowrance. 1997. Assessment of forest management effects on nitrate removed by riparian buffer systems. Trans. ASAE 40:383–391.

Hubbard, R.K., G.L. Newton, J.G. Davis, R. Lowrance, G. Vellidis, and C.R. Dove. 1998. Nitrogen assimilation by riparian buffer systems receiving swine lagoon wastewater. Trans. ASAE 41:1295–1304.

Inamdar, S.P., R. Lowrance, L.S. Altier, R.G. Williams, and R.K. Hubbard. 1999a. Riparian Ecosystem Management Model (REMM): II. Testing of the water quality and nutrient cycling component for a coastal plain riparian system. Trans. ASAE 42:1691–1707.

Inamdar, S.P., J.M. Sheridan, R.G. Williams, D.D. Bosch, R. Lowrance, L.S. Altier, and D.L. Thomas. 1999b. Riparian Ecosystem Management Model (REMM): I. Testing of the hydrologic component for a coastal plain riparian system. Trans. ASAE 42:1679–1689.

Jacobs, T.C., and J.W. Gilliam. 1985. Riparian losses of nitrate from agricultural drainage waters. J. Environ. Qual. 14:472–478.

Lee, K.H., T.M. Isenhart, and R.C. Schultz. 2003. Sediment and nutrient removal in an established multi-species riparian buffer. J. Soil Water Conserv. 58:1–8.

Lee, K.H., T.M. Isenhart, R.C. Schultz, and S.K. Mickelson. 2000. Multispecies riparian buffers trap sediment and nutrients during rainfall simulations. J. Environ. Qual. 29:1200–1205.

Lowrance, R. 1991. Effects of buffer systems on the movement of N and P from agriculture to streams. p. 87–96. *In* V. Holter (ed.) Proc. of the Int. Conf. on Nitrogen, Phosphorus, and Organic Matter. Danish Ministry of the Environment, Copenhagen, Denmark.

Lowrance, R. 1992. Groundwater nitrate and denitrification in a Coastal Plain riparian forest. J. Environ. Qual. 21:401–405.

Lowrance, R., L.S. Altier, R.G. Williams, S.P. Inamdar, D.D. Bosch, R.K. Hubbard, and D.L. Thomas. 2000a. The Riparian Ecosystem Management Model. J. Soil Water Conserv. 55:27–36.

Lowrance, R., R.K. Hubbard, and R.G. Williams. 2000b. Effects of a managed three-zone riparian buffer system on shallow groundwater quality in the southeastern coastal plain. J. Soil Water Conserv. 55:212–220.

Lowrance, R., R.L. Todd, and L.E. Asmussen. 1983. Waterborne nutrient budgets for the riparian zone of an agricultural watershed. Agric. Ecosyst. Environ. 10:371–384.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34:374–377.

Lowrance, R., G. Vellidis, R.D. Wauchope, P. Gay, and D.D. Bosch.

1997. Herbicide transport in a managed riparian forest buffer system. Trans. ASAE 40:1047–1057.
Magette, W.L., R.B. Brinsfield, R.E. Palmer, and J.D. Wood. 1989. Nutrient and sediment removal by vegetated filter strips. Trans. ASAE 32:663–667.
Peterjohn, W.T., and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology 65:1466–1475.
SAS Institute. 1999. SAS system for Windows. Version 8. SAS Inst., Cary, NC.
Sheridan, J.M., R. Lowrance, and D.D. Bosch. 1999. Management effects on runoff and sediment transport in riparian forest buffers. Trans. ASAE 42:55–64.
Sheridan, J.M., R. Lowrance, and H.H. Henry. 1996. Surface flow sampler for riparian studies. Appl. Eng. Agric. 12:183–188.
Sokal, R.R., and F.J. Rohlf. 1981. Biometry. W.H. Freeman, San Francisco.
Vellidis, G., R. Lowrance, P. Gay, and R.K. Hubbard. 2003. Nutrient transport in a restored riparian wetland. J. Environ. Qual. 32:711–726.
Verchot, L.V., E.C. Franklin, and J.W. Gilliam. 1997. Nitrogen cycling in Piedmont vegetated filter zones: I. Surface soil processes. J. Environ. Qual. 26:327–336.
Welsch, D.J. 1991. Riparian forest buffers: Function and design for protection and enhancement of water resources. NA-PR-07-91. USDA Forest Service, Radnor, PA.