RESEARCH

Riparian zone impact on phosphorus movement to a Coastal Plain black water stream

J.M. Novak, P.G. Hunt, K.C. Stone, D.W. Watts, and M.H. Johnson

ABSTRACT: Riparian zones are an important conservation practice because they can decrease the entry of sediments and nutrients into sensitive aquatic ecosystems. We evaluated the effectiveness of a Coastal Plain riparian zone in decreasing the movement of phosphorus (P) into a black water stream from an overloaded swine manure spray field. Soil P concentrations (Mehlich 3 P, M3P; and total P, TP) were measured in a spray field, grass strip, mid-riparian, and stream edge continuum. Dissolved P (DP) was measured in ground water wells located in the spray field, grass strip, and stream edge and in in-stream grab samples. The spray field and grass strip areas had high soil M3P concentrations. Low M3P concentrations were detected in soils in the mid-riparian and stream edge areas, indicating effective retention of P by the grass strip area. Elevated DP concentrations were detected in the spray field and grass strip wells, while stream edge wells were low. The riparian zone contributed to decreased DP concentrations between the grass strip and stream edge wells. Furthermore, stream grab samples were consistently low in DP concentrations. We conclude that a riparian zone can effectively limit the movement of P-enriched sediments and prevent DP-enriched ground water from entering a local stream, even in a heavily loaded situation.

Keywords: Black water stream, grass buffer strip, phosphorus movement, riparian zone

Land application of animal manure in the southeastern Coastal Plain region has been associated with the buildup of excess P in topsoil and the deep leaching of P to subsoil depths (Kingery et al. 1994; Mozaffari and Sims 1994; Eghball et al. 1996). Water quality in nutrient-sensitive aquatic ecosystems can be at risk if P-enriched sediment or ground water enters these aquatic ecosystems. If P concentrations increase to sufficient levels, natural ecosystem structure and function can be disturbed (Likens and Bormann 1995) and surface water eutrophication can result (Sims 1993). Eutrophication of surface water reduces its potential use for recreational and industrial purposes. These conditions have even been linked to outbreaks of *Pfiesteria,* which are pathogens harmful to aquatic organisms and humans (Burkholder et al. 1997; Mallin 1999). Seaside communities in the southeastern Coastal Plain region are heavily dependent on maintaining high quality water for the tourist trade. It is financially important, as well as environmentally significant, that agricultural management strategies be developed that can reduce the amount of nonpoint source P entering surface water systems.

A simple agricultural management practice to minimize P entry into surface water sources is to reduce the movement of P-enriched runoff and eroded sediments and/or reduce nutrient concentrations in ground water recharge. Reduced sediment movement from agricultural fields using vegetative filter strips has been demonstrated by Dillaha et al. (1989) and Schellinger and Clausen (1992). Filter strips are bands of vegetation, usually grasses, located down slope from cropped fields or animal production facilities to slow sediment and filter chemicals/nutrients from surface runoff. Additionally, natural landscape features, such as forested riparian zones, have been reported to act as efficient P sinks and reduce levels of P in agricultural runoff to surface waters (Yates and Sheridan 1983; Brinson et al. 1984; Lowrance et al. 1984; Cooper and Gilliam 1987). Phosphorus retention is accomplished because riparian zones contain wetland soils, which have the ability to retain P by sorption to clays and Fe and Al oxides (Patrick and Khalid 1974; Mozaffari and Sims 1994). Additionally, associated wetland plant communities can assimilate P by vegetative uptake (Fail et al. 1986). Assimilative P capacities of wooded riparian zones have been reported between 30 and 50% in North Carolina (Cooper and Gilliam 1987) and Georgia (Lowrance et al. 1984). However, riparian zones have a finite ability to assimilate P (Reddy et al. 1999). Consequently, it is important to evaluate their assimilation performance when receiving high P input from heavily loaded areas.

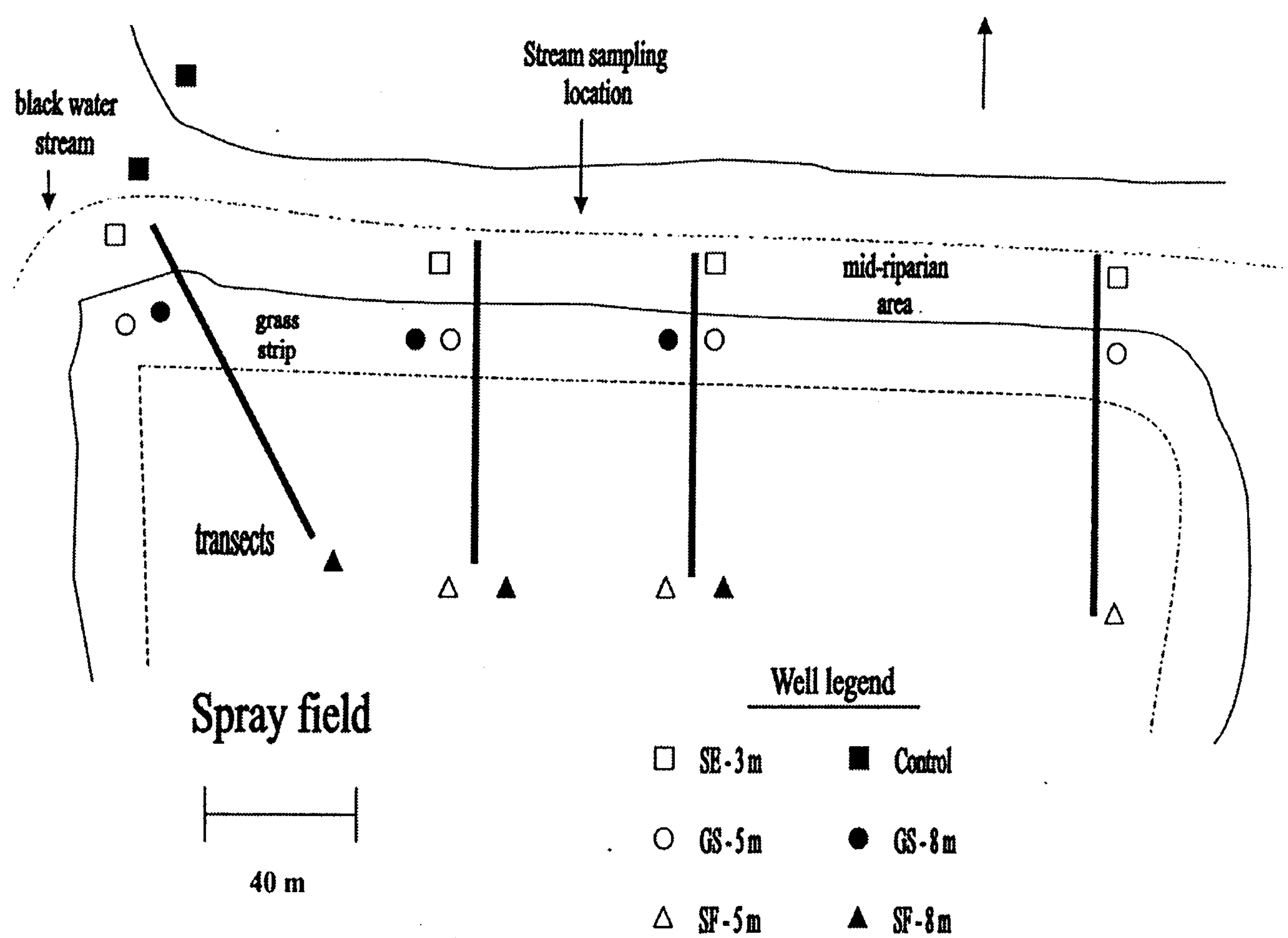
We investigated movement of P from a spray field that received intensive applications of swine manure (83 to 625 kg P ha^{-1} yr^{-1}; 74 to 558 lb P ac^{-1} yr^{-1}) for approximately 10 yr. This spray field was adjacent to a riparian zone and an adjoining stream. In a previous study of this spray field, Novak et al. (2000) reported that the intensive manure loading rates contributed to accumulation of excess P in the topsoil and enrichment of dissolved P (DP) in the ground water beneath the spray field. In the present study, our objective was to determine if a Coastal Plain riparian zone adjacent to the seriously overloaded swine manure spray field could effectively assimilate P and minimize the entry of P into a black water stream.

Methods and Materials

Description of the study area. The study area is located in the Cape Fear River basin on the Middle Coastal Plain physiographic region of North Carolina. Landscape features, agricultural activities, soil, aquifer description, and climatic conditions have been reported (Novak et al. 1998; Stone et al. 1998). The spray field is mapped as the Autryville soil series (loamy, siliceous, thermic Arenic Paleudult) and has slopes that range from 1 to 5%. Coastal Bermuda grass (*Conodon dactylon* L.) is grown in the spray field to assist in

Jeffrey M. Novak is a soil scientist, **Patrick G. Hunt** is a soil scientist and research leader, **Kenneth C. Stone** is an agricultural engineer, **Donald W. Watts** is a soil scientist, and **Melvin H. Johnson** is an agricultural engineer with the U.S. Department of Agriculture Agricultural Research Service (USDA-ARS) Coastal Plains Research Center (CPRC) in Florence, South Carolina.

Figure 1
Location of spray field, study transects, and ground water wells.



nutrient removal. This 5 ha (12.4 ac) field has received swine manure effluent for approximately 10 yr at a loading rate estimated (using model simulation and manure management plan) to be between 83 and 625 kg P ha^{-1} yr^{-1} (74 to 558 lb P ac^{-1} yr^{-1}).

Four parallel, 50 m (55 yd) long transects were established extending from the spray field, through the riparian area, and terminating at the stream edge (Figure 1). These transects were divided into sections of spray field (SF), grass strip (GS), mid-riparian, and stream edge (SE) areas according to Castelle et al. (1994) and Schultz et al. (1995). The transects represent a typical spray field-riparian zone-stream continuum of the southeastern Coastal Plain region. The grass strip area, which forms a border around the spray field, is approximately 5-10 m (5.5-11 yd) wide and has a permanent Bermuda grass cover. The mid-riparian area is adjacent to the grass strip area and varies in width from 12 to 18 m (13 to 20 yd). The stream edge area is located along a shallow black water stream that drains the study site. This stream segment is a primary ordered stream (Horton 1945) and is located in the headwaters of this subwatershed. Both the mid-riparian and stream edge areas are on a floodplain and are collectively 1 ha (2.5 ac) in size. Both areas contain the Bibb soil series. The Bibb soil (coarse-loamy, siliceous, acid, thermic Typic Fluvaquent) is poorly drained and has soil organic carbon contents ranging between 50 and 300 mg OC kg^{-1} (50 and 300 ppm OC). Bald Cyprus (*Taxodium distichum* L.), Swamp Chestnut Oak (*Quercus michauxii* Nutt.), Green Ash (*Fraxinus pennsylvanica* Marsh.), Red Maple (*Acer rubrum* L.), and Sycamore (*Platanus occidentalis* L.) were planted in the mid-riparian zone in 1994 to assist in sediment retention and nutrient removal. These trees currently range from 2 to 14 m (2 to 15 yd) in height and from 4 to 23 cm (1.5 to 9 in) in diameter. Two groups for each of the five tree species were identified and the population in each group was determined (n = 10-20). The above ground tree stem biomass (kg) for each tree population was estimated by measuring the diameter at breast height and the tree height. Only tree stem biomass was estimated, since total P (TP) uptake by this portion represents a long-term TP tie-up. Tree leaf TP uptake is a short-term tie-up since the leaves, after senescence, returns TP back into the P cycle. Total P was then estimated by measuring above ground woody biomass and multiplying by a sapwood TP concentration (99.9 µg TP g^{-1}; 99.9 ppm TP) from a Maryland Coastal Plain riparian forest containing similar tree species (Peterjohn and Correll 1984).

Well installation, water and soil collection. Our well sampling locations and installation depths were part of a prior research project concerned with an assessment of nitrate distribution trends in shallow ground water around this watershed (Stone et al. 1995). During that project, 19 ground water wells were installed in and around the spray field and riparian areas adjacent to the spray field according to local topographic features, interaction with the landowner, and anticipated ground water flow paths (Stone et al. 1995). We used these same 19 wells for our study. Six wells are located in the spray field, seven are located in the grass strip area, and four at the stream edge area (Figure 1). Stone et al. (1995) installed the wells at different soil depths (8, 5, and 3 -m [26, 16, and 10 ft] deep) in the spray field, grass strip, and stream edge areas because well depth is known to influence the susceptibility of a well to nutrient contamination (Bruggeman et al. 1995; Richards et al. 1996). Two control wells are located across the black water stream in an area that has not received swine manure. One control well is located at a stream edge area (3 m [10 ft] deep), while the other well is installed in a grass strip area (5 m [16 ft] deep).

Water table depths in each well were measured to determine ground water flow patterns and the volume of water necessary to purge the well of stagnant water before sample collection. Flow potentials determined using Surfer (Golden Software, Inc., Golden, CO) along the transects showed that ground water flows normal to the stream (K.C. Stone, personal communication, 2000). Ground water in all the wells was sampled monthly from March 1992 to July 1998 and then twice in 1999 (June and December) using methods described previously (Novak et al. 1998). Stream grab samples were collected in mid-stream (Figure 1), following the same collection schedule as the wells. Sampling frequency was reduced to two samples in 1999 because of low variation in DP detection trends.

Soil sampling sites (four per transect) were selected in the spray field, grass strip, mid-riparian, and stream edge locations. Soil samples were collected during June 1999, using an 8 cm (26 in) diameter bucket auger at depths of 0-15, 15-45, and 45-90 cm (0-6, 6-18, and 18-35 in). Background soil Mehlich 3 P (M3P) and TP concentrations for an Autryville soil collected 100-150 m (328-492 ft) away from the spray field have been published previously (Novak et al. 2000).

Phosphorus measurements. All ground and stream water samples were filtered 0.45 μm (2 x 10^{-5} in) to remove sediment and were analyzed for DP on a TRACCS Auto analyzer (Bran Lubber, Elmsford, NY) using U.S. Environmental Protection Agency method 354.1 (Kopp and McKee 1983). Total P was measured using the ascorbic acid method according to Standard Methods (American Public Health Association 1992) with a Technicon Auto analyzer (Tarrytown, NY). The minimum detection limit for both of these methods was 40 μg P L^{-1} (40 ppb). All soils were extracted for M3P (Mehlich 1984) and quantified using a colorimetric method (Murphy and Riley 1962) with a minimum detection limit of 0.5 mg P kg^{-1} (0.5 ppm).

Statistics. All soil and water P concentrations were tested for normality by the Kolmogorova-Smirnova test. Results in most cases indicated that the data could not be assumed normally distributed. Thus, movement of soil P from the spray field into the riparian areas was statistically evaluated by comparing the M3P and TP concentrations between locations using a nonparametric Kruskal-Wallis test. When significant differences were found, the nonparametric Dunn's test was used to make multiple comparisons among the spray field, grass strip, mid-riparian, and stream edge areas (control). The stream edge location was chosen as the control because its soil M3P and TP contents were similar to contents measured at three other stream edge locations upstream from the spray field. Significant differences in the yearly well mean DP concentration between the spray field 5 m (16 ft) versus spray field 8 m (26 ft) wells and the grass strip 8 m (26 ft) versus grass strip 5 m (16 ft) versus stream edge 3 m (10 ft) wells were also tested using a Kruskal-Wallis multiple comparison test. For both of these tests, a zero value was arbitrarily used when a DP concentration was below the level of DP detection (< 40 μg P L^{-1}; 40 ppb). The use of a zero value resulted in some mean ground water and stream water DP concentrations being < 40 μg P L^{-1}(40 ppb). Ground water mean DP concentrations were also calculated using a concentration of 20 μg P L^{-1} (20 ppb) when the DP concentration was below the detection limit (40 μg P L^{-1}; 40 ppb). However, our statistical results have been determined using the 0 μg P L^{-1} approach. This allowed consistency with our previously reported results on P dynamics in this spray field (Novak et al. 2000). All statistical tests were preformed using SigmaStat software (version 2.03; SPSS, Inc., Chicago, IL).

Results and Discussion

Soil phosphorus concentrations. Ten years of intensive swine manure application to this spray field has contributed to a high mean M3P concentration throughout the Autryville soil profile within the spray field (Table 1). High M3P concentrations were also measured in soil profiles in the grass strip area, which was installed around the perimeter of the spray field as part of a best management practice strategy to minimize offsite P movement. Although it is not part of the actual spray field, both management information and the high soil M3P concentrations in the grass buffer strip indicate that it has received large but undefined amounts of both liquid and solid manure over the life of the study. Nonetheless, the ability of the grass buffer strip to retain/restrict movement of P is evident by the dramatic decline in soil M3P concentrations in the mid-riparian and

Table 1. Mean and standard deviations for Mehlich 3 P (M3P) and total P (TP) concentrations at specific landscape locations† in the spray field-riparian zone continuum (standard deviations are in parentheses)‡

Soil depth (cm)	Spray field	Grass strip	Mid-riparian	Stream edge
	M3P (mg kg^{-1})			
0-15	251 (73)[a]	247 (115)[a]	21 (31)	12 (12)
15-45	195 (57)[a]	176 (150)	12 (9)	4 (4)
45-90	112 (72)[a]	71 (91)	7 (5)	3 (1)
	TP (mg kg^{-1})			
0-15	382 (101)	343 (109)	118 (60)[a]	359 (202)
15-45	278 (100)	223 (123)	110 (63)	246 (162)
45-90	269 (122)	135 (74)	135 (59)	191 (101)

† n = 4 per location

‡ means followed by the letter "a" are significantly different than the stream edge location (P <0.1)

stream edge locations ($P < 0.1$).

As with the M3P, TP concentrations in soil profiles were high in the spray field and grass strip areas due to the application of swine manure (Table 1). The stream edge area also had high TP concentrations. However, this area of the riparian zone probably accumulated P by stream bank deposition of P-enriched sediments. Only the surface soil of the mid-riparian zone had a significantly lower mean TP concentration. Low TP concentrations in this area of the mid-riparian zone were probably due to a lack of P input from either the stream system or swine manure spray field.

The buffering effect of the riparian zone is evident when the M3P to TP ratios are plotted along the transects (Figure 2). In the two soil layers [0-15 and 15-45 cm (0-6 and 6-18 in) deep] of the spray field and grass strip areas, the majority of P in the total P pool was plant available (Figure 2, > 65%). In contrast, the M3P to TP ratio decreased substantially in the mid-riparian and grass strip areas (< 20%), indicating that more P occurred in forms unavailable to plants. Stream edge ratios (< 0.05) were similar to background control soil values in a nearby Autryville soil that had not received animal manure (Novak et al. 2000). The large relative difference in the M3P to TP ratio between the manure and non-manure treated soils documents that the intensive rates of manure application to this spray field have contributed to a preferential accumulation of P forms that are plant available (Mehlich 3 reagent).

The high M3P to TP ratios in the spray field soil profiles are consistent with results of Novak et al. (2000), and indicate leaching of P forms. Reddy et al. (1980) suggested that organic acids from the decomposition of manure would form stable complexes with Fe and Al and, consequently, block P retention. Eghball et al. (1996) also suggested that organic P forms might move preferentially in a Nebraska sandy loam soil treated with cattle manure. Our data further suggest that the soil P sorption sites are probably over-saturated and that organic P forms are leaching through the Autryville soil in this spray field. Since the organic P forms may not bind to Fe and Al functional groups on solid phases as well as inorganic P forms, they could move preferentially to the ground water.

Dissolved P concentrations in ground and surface water. We detected DP-enriched ground water beneath this spray field in the

Figure 2

Relationship between Mehlich 3 P (M3P) and total P (TP) ratio measured in soil by depth and distance from the stream.

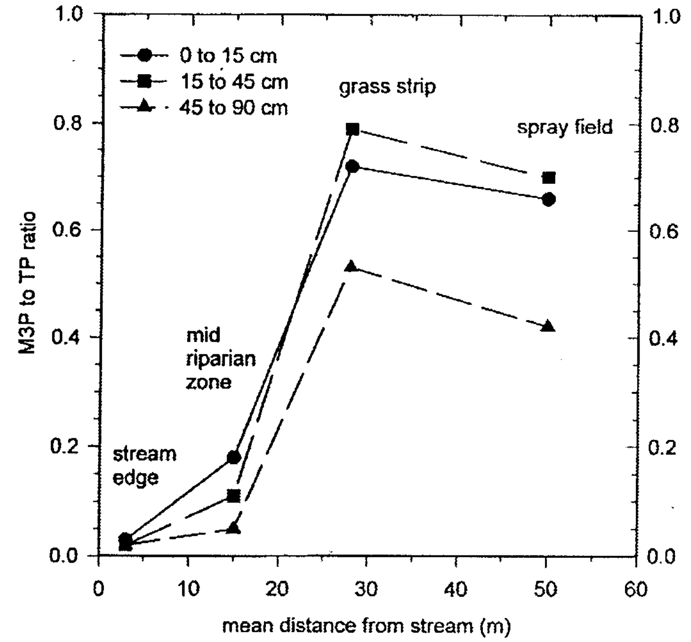


Figure 3

Yearly mean dissolved P (DP) concentrations for the spray field (SF) 5 and 8 m deep wells (yearly mean well DP concentration compared, and symbols followed by different letter are significantly different $P < 0.1$).

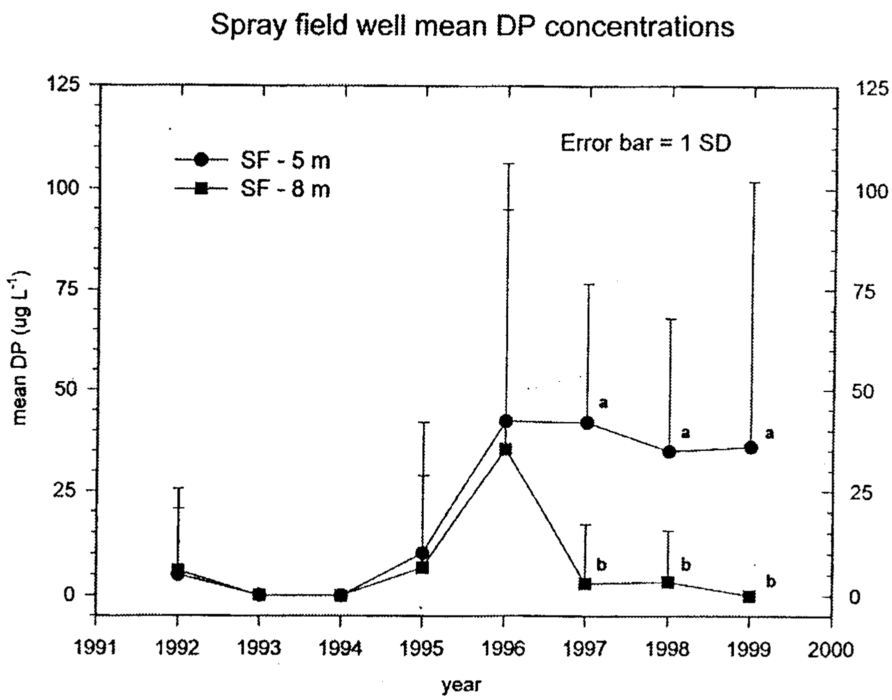
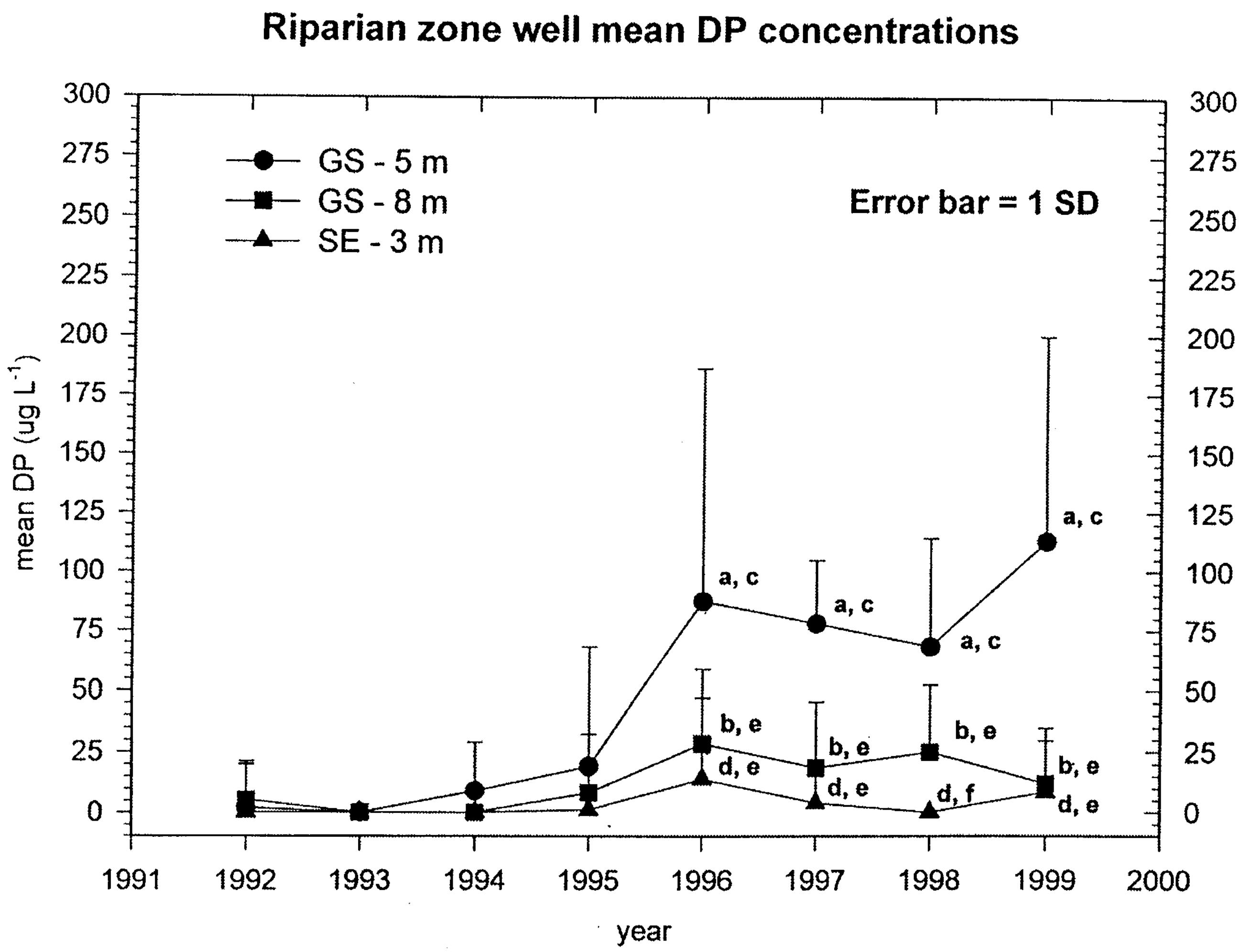


Figure 4

Yearly mean dissolved P (DP) concentration for the grass strip (GS) 5 and 8 m deep wells and for the grass strip (GS) 3 m deep wells (yearly mean well DP concentration compared, and symbols followed by different letter are significantly different $P < 0.1$).



spray field 5 m (16 ft) and 8 m (26 ft) deep wells (Figure 3). Initially, all SF wells were low in mean DP concentrations (1992 to 1995). However, in 1996 a large increase in the mean ground water DP concentration occurred, which implies a breakthrough of P. From 1997 to 1999, the spray field 5 m (16 ft) deep wells had mean DP concentrations that remained between 35 and 40 μg P L^{-1} (35 and 40 ppb). Mean DP concentrations in the spray field 8 m (26 ft) deep wells returned to near 0 μg P L^{-1} shortly after P breakthrough.

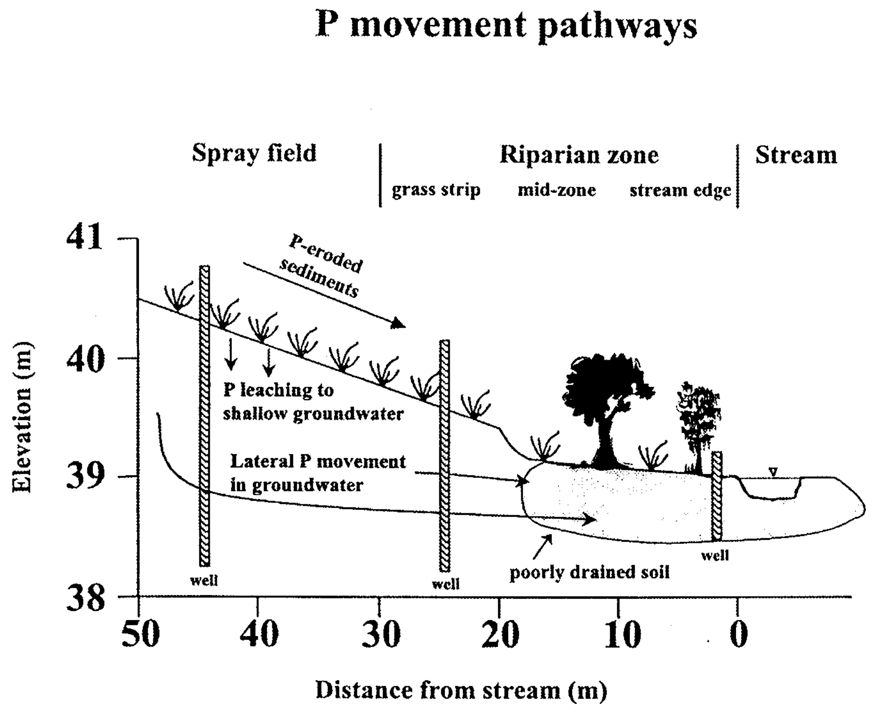
It was hypothesized that ground water beneath the grass strip area would be impacted the same as the spray field, since they both had comparable soil M3P concentrations. We found such an impact; wells located in the grass strip area had high DP concentrations (Figure 4). In fact, the ground water DP concentrations were higher than in spray field wells (compare Figure 3 with Figure 4). Additionally, the impact was greatest in the grass strip 5 m (16 ft) deep wells. There was relatively little impact on the grass strip 8 m (26 ft) wells. Nonetheless, DP concentrations in the grass strip 8 m wells were higher than the 0 μg P L^{-1} in the grass strip 5 m (16 ft) control well located across the stream.

We did not find DP-enriched ground water at the stream edge area of the riparian zone (Figure 4). In fact, the mean DP concentration in these wells was similar to the stream edge control well located across the stream. Low ground water DP concentrations at the stream edge location suggest that the grass strip area was capable of buffering ground water DP concentrations. This idea is supported by high soil M3P concentrations in the grass strip area compared to low soil M3P concentrations in the mid-riparian and stream edge areas (Table 1). Soil in the mid-riparian and stream edge areas would probably have high soil M3P concentrations if DP-enriched ground water had totally traversed through these areas of the riparian zone.

The P buffer effectiveness of the grass strip area may only be a temporary condition. It is conceivable that the grass strip may eventually become P saturated and allow for P forms to enter into the mid-riparian and stream edge areas. However, in these areas P will probably be assimilated by various biotic and abiotic mechanisms, thus maintaining very low stream water DP concentrations.

Comparison of the pooled (1992-1999) DP mean concentrations in the stream edge 3 m (10 ft) wells (3.4 μg L^{-1}; 3.4 ppb) with the pooled mean DP concentration in the grass

Figure 5
Phosphorus movement pathways along the spray field-riparian zone-stream continuum (note scale differences between the x and y axes).



strip 5 m (16 ft) wells (25.4 μg L^{-1}; 25.4 ppb) showed a large relative difference of 87%. This is a simple comparison of pooled mean DP values; nonetheless, the large difference in mean ground water DP concentration between the stream edge and grass strip wells verifies that substantial retention of DP has occurred.

Although DP concentrations in ground water of the riparian zone can be influenced by rainfall and flooding by stream water (R. Lowrance, personal communication, 2000), nutrient removal mechanisms associated with the forested wetland riparian zone can also influence ground water DP concentrations. The tree species planted in the riparian zone were estimated to collectively assimilate 7.7 kg TP ha^{-1} yr^{-1} (6.8 lb TP ac^{-1} yr^{-1}). Our TP assimilation estimation is similar to TP storage by riparian vegetation in Maryland (Correll and Weller 1989, 3-5 kg TP ha^{-1} yr^{-1}; 2.7-4.5 lb TP ac^{-1} yr^{-1}) and by a riparian forest in Georgia (Fail et al. 1986, 6.9 kg TP ha^{-1} yr^{-1}; 6.2 lb TP ac^{-1} yr^{-1}). Compared to the total mass of TP annually applied to the spray field (415 kg TP; 913 lb TP), these riparian tree species have collectively taken up approximately 2% of the annual TP mass. These data suggest that P assimilation by woody structures of these five tree species is minimal when compared to the annual P mass load applied to this heavily loaded site. The major mechanism of DP removal from ground water by the wooded riparian zone is probably via sorption reactions by soil components (Brinson et al. 1984; Mulholland 1992; Mozaffari and Sims 1994).

Our approach of calculating mean well DP concentrations using a zero value could be low, since a substantial number of water samples had DP concentrations that were below our 40 μg L^{-1} (40 ppb) detection limit. Nonetheless, calculation of our mean DP concentrations using a value that was equal to one-half of our detection limit (20 μg L^{-1}; 20 ppb) rather than zero resulted in the pooled grass strip 5 m (16 ft) and stream edge 3 m (10 ft) wells having a mean ground water DP concentration of 39.8 versus 22.0 μg L^{-1} (39.8 versus 22.0 ppb). Results from either calculation scheme show that a Coastal Plain riparian zone can substantially reduce ground DP concentrations even when it is heavily impacted from a field receiving intensive rates of animal waste applications.

The studied stream system occurs in the headwater area of this subwatershed, so stream water characteristics provide a good indication of the function of the riparian zone. Low stream concentrations were the primary demonstration of the high potential of the riparian zone to minimize the impact on the stream. Very few DP detections (8%, total sample basis, n = 90) were above the detection limit, and the mean concentration was < 5 μg L^{-1} (5 ppb) (1992-1999). The occurrence of a low stream water mean DP concentration during 8 years of study exemplifies the beneficial effect of a Coastal Plain riparian zone in reducing the potential transport of P to a surface water source.

Summary and Conclusion

Decreasing the offsite movement of P from manure-treated fields into ground and surface water systems is an important water quality goal. Best management practices to reduce offsite nutrient movement include the creation and management of a riparian zone around the field (Figure 5). The riparian zone in our study was composed of a grass strip section, a mid-riparian zone planted with trees, and a stream edge area. Phosphorus applied as manure to fields can be physically transported via runoff or with eroded sediments from the site of application into the riparian zone. The grass buffer strip, designed to reduce the surface movement of runoff and eroded sediments, was confirmed to be an effective first line of defense against stream water P contamination. The grass buffer strip contributed to a dramatic decline in the soil M3P/TP ratio between soil in the grass strip (0.55-0.80) and the mid-riparian zone (0.05-0.2).

Phosphorus can leach through Coastal Plain soils and move to ground water from swine manure treated fields (Figure 5). Our results showed that over-application of swine manure for a long period of time contributed to the leaching of DP through the soil profile and enrichment of shallow ground water beneath the field. Ground water monitoring wells and soil sampling in transects through the spray field-riparian zone-stream continuum allowed us to determine the magnitude of DP leaching and transport. Our data showed that soil and ground water in the spray field and grass strip areas were high in P concentrations, probably caused by leaching of P and transport by laterally flowing groundwater (Figure 5). Stream edge wells and stream grab samples, on the other hand, were low in DP concentrations, suggesting that the grass strip area was effective at buffering DP in ground water. Removal of DP in ground water was probably due to sorption by solid phases, since our results showed that P mass assimilated by the planted tree species was low when compared to the annual P mass load applied to the field.

It is conceivable that some DP-enriched ground water may eventually move from the grass strip into the mid-riparian area (Figure 5). Very low ground water DP concentrations at the stream edge and low soil M3P concentrations measured in the mid-riparian and stream edge areas, however, suggest that it did not traverse across the mid-riparian area. Phosphorus removal by these areas will probably continue to be an effective buffer of ground water DP concentrations for a period of time, although the effectiveness may be temporary. These areas may eventually become P-saturated and then act a source area of P to the stream. Nonetheless, the grass filter strip and mid-riparian areas were effective at buffering P entry from the spray field into the adjacent stream. Our results confirm the beneficial effects of riparian zones in the Coastal Plain landscape and highlight their positive management effect on water quality.

Disclaimer

Mention of a specific product or vendor does not constitute a guarantee or warranty of the product by the U.S. Department of Agriculture or imply its approval to the exclusion of other products that may be suitable for use.

Acknowledgements

Sincere gratitude is expressed to employees of the North Carolina Cooperative Extension Service, and North Carolina State University for assistance with this project. The research was partially funded by CSREES Grant no. 58-6657-9-011 entitled "Management practices to reduce nonpoint source pollution on a watershed basis."

References Cited

American Public Health Association. 1992. Standard Methods for the Examination of Water and Waste Water. 18th ed. American Public Health Association, Water and Environment Federation, Washington, DC.

Brinson, M.M., H.D. Bradshaw, and E.S. Kane. 1984. Nutrient assimilative capacity of an alluvial floodplain swamp. Journal of Applied Ecology 21:1041-1057.

Bruggeman, A.S., S. Mostaghimi, G.I. Holtzman, V.O. Shanholts, S. Shukla, and B.B. Ross. 1995. Monitoring pesticide and nitrate in Virginia's groundwater-A pilot study. Transaction of the American Society of Agricultural Engineers 38:797-807.

Burkholder, J.M, M.A. Mallin, H.B. Glasgow, jr., L.M. Larsen, M.R. McIver, G.C. Shank, D.D. Melia, D.S. Briley, J. Springer, B.W. Touchette, and E.K. Hannon. 1997. Impacts to a coastal river and estuary from rupture of a large swine waste holding lagoon. Journal of Environmental Quality 26:1451-1466.

Castelle, A.J., A.W. Johnson, and C. Conolly. 1994. Wetland and stream buffer size requirements — A review. Journal of Environmental Quality 23:878-882.

Cooper, J.R. and J.W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Science Society of America Journal 51:1600-1604.

Correll, D.L. and D.E. Weller. 1989. Factors limiting processes in freshwater wetlands: An agricultural primary stream riparian forest. Pp. 9-23. In: R. Sharitz and J. Gibbons (eds). Freshwater Wetlands and Wildlife. U.S. Department of Energy, Oak Ridge, TN.

Dillaha, T.A., R.B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. Transaction of the American Society of Agricultural Engineers 32:513-519.

Eghball, B., G.D. Binford, and D.D. Baltensperger. 1996. Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. Journal of Environmental Quality 25:1339-1343.

Fail, J.L., M.N. Hamzah, B.L. Haines, and R.L. Todd. 1986. Above and below ground biomass, production, and element accumulation in riparian forests of an agricultural watershed. Pp. 193-224. In: D. L. Correl (ed). Watershed Research Perspectives. Smithsonian Institution, Washington, DC.

Horton, R.E. 1945. Erosional development of streams and their drainage basin. Geological Society of America Bulletin 56:275-370.

Kingery, W.L., C.W. Wood, D.P. Delaney, J.C. Williams, and G.L. Mullins. 1994. Impact of long-term land application of broiler litter on environmentally related soil properties. Journal of Environmental Quality 23:139-147.

Kopp, J.F. and G.D. McKee (eds). 1983. Methods for chemical analysis of water and wastes. USEPA-600/4-79-020. Environmental Monitoring and Support Lab, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, OH.

Likens, C.E. and F.H. Bormann. 1995. Biochemistry of a Forested Ecosystem. 2nd ed. New York: Springer-Verlag.

Lowrance, R.R., R.L. Todd, and J. Fail, Jr. 1984. Riparian forests as nutrient filters in agricultural watersheds. Biological Science 34:374-377.

Mallin, M.A. 1999. Impacts of industrial animal production on rivers and estuaries. American Scientist 88:26-37.

Mehlich, A. 1984. Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. Communication in Soil Science and Plant Analysis 15:1409-1416.

Mozaffari, M. and J.T. Sims. 1994. Phosphorus availability and sorption in an Atlantic Coastal Plain watershed dominated by animal based agriculture. Soil Science 157:97-107.

Mulholland, P.J. 1992. Regulation of nutrient concentrations in a temperate forest stream: Roles of upland, riparian, and instream processes. Limnology and Oceanography 37:1512-1526.

Murphy, J. and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Analytical Chimica Acta 27:31-36.

Novak, J.M., D.W. Watts, K.C. Stone, M.H. Johnson, and P.G. Hunt. 1998. Pesticide and metabolites in the shallow ground water of an eastern Coastal Plain watershed. Transaction of the American Society of Agricultural Engineers 41:1383-1390.

Novak, J.M. D.W. Watts, K.C. Stone, M.H. Johnson, and P.G. Hunt. 2000. Phosphorus movement to ground water from intensive swine manure application. Journal of Environmental Quality 29:1310-1315.

Patrick, W.H. and R.A. Khalid. 1974. Phosphate release and sorption by soils and sediments: Effect of aerobic and anaerobic conditions. Science 186:53-55.

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.

Reddy, K.R., M.R. Overcash, R. Kahled, and P.W. Westerman. 1980. Phosphorus absorption-desorption characteristics of two soils utilized for disposal of animal manures. Journal of Environmental Quality 9:86-92.

Reddy, K.R., R.H. Kadlec, E. Flaig, and P.M. Gale. 1999. Phosphorus retention in streams and wetlands: A review. Critical Reviews in Environmental Science and Technology 29:83-146.

Richards, R.P. D.P. Baker, N.L. Creamer, J.W. Kramer, D.E. Ewing, B.J. Merryfield, and L.R. Wallrabenstein. 1996. Well water quality, well vulnerability, and agricultural contamination in the Midwestern United States. Journal of Environmental Quality 25:389-402.

Schellinger, G.R. and J.C. Clausen. 1992. Vegetative filter treatment of dairy barnyard runoff in cold regions. Journal of Environmental Quality 21:40-45.

Schultz, R.C., J.P. Colletti, T.M. Isenhart, W.W. Simkins, C.A. Rodrigues, P. Wray, M.L. Thompson, and J. Pease. 1995. Riparian buffer strip systems that improve water quality. Pp. 235-238in the proceedings of Clean Water-Clean Environment-21st Century, 5-8 Mar, Kansas City, MO.

Sims, J.T. 1993. Environmental soil testing for phosphorus. Journal of Production Agriculture 6:501-507.

Stone, K.C., P.G. Hunt, S.W. Coffey, and T.A. Matheny. 1995. Water quality status of a USDA water quality demonstration project in the eastern Coastal Plain. Journal of Soil and Water Conservation 50:567-571.

Stone, K.C., P.G. Hunt, M.H. Johnson, and T.A. Matheny. 1998. Impact of swine waste application on ground and stream water quality in an eastern Coastal Plain watershed. Transactions of the American Society of Agricultural Engineers 41:1665-1670.

Yates, P. and J.M. Sheridan. 1983. Estimating the effectiveness of vegetated flood-plains/wetlands as nitrate-nitrite and orthophosphorus filters. Agricultural Ecosystems and Environment 9:303-314.