Calibration of Paired Basins prior to Streambank Fencing of Pasture Land

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ABSTRACT

Streambank fencing is a best management practice that is targeted to reduce suspended sediment and nutrient inputs to streams by reducing direct inputs from animals, eliminating streambank trampling, and promoting streambank revegetation. A paired basin study is being conducted in two small adjacent basins to determine the water quality effects of streambank fencing. This article documents the 3-yr calibration period between control and treatment basins prior to fence installation. Approximately 70% of land adjacent to streambanks in the study area is used as pasture. Nutrient quantities applied as manure, benthic-macroinvertebrate communities, and the physical habitat of each stream were similar in both basins. Total N, P, and suspended sediment yields measured at the outlet of each basin averaged about 56, 2.8, and 2650 kg ha^{-1} on an annual basis. For both basins, about 90% of the total N yield was attributable to dissolved NO_3–N and about 90% of the total N yield occurred during nonstormflow; conversely, about 90% of the total P yield was attributable to stormflow and 60 to 65% of the total P yield was suspended. Regression equations developed between both basins for low flow and stormflow samples for nutrients, suspended sediment, and discharge indicated a significant relation for most constituents. Pretreatment relation between basins for low flow and stormflow samples would need to change by 6 and 14% for total N concentrations and 24 and 9% for total P concentrations in order for streambank fencing to significantly affect water quality in the treatment basin.

AGRICULTURE is the predominant land use in the Mill Creek Basin of Lancaster County, PA, and much of the area along streams is used to pasture dairy cattle. Pastured areas have been identified as nonpoint sources of suspended sediment and nutrients to streams (McLeod and Hegg, 1984; Edwards et al., 1996). Streambank fencing to exclude animal access is a best management practice (BMP) that is targeted to reduce suspended sediment and nutrient inputs to streams by reducing direct nutrient inputs from animals in the stream and stopping streambank trampling. Livestock trampling of streambanks increases bank erosion (Kauffman et al., 1983). Livestock also can change physical soil properties in grazed areas by increasing soil compaction (Alderfer and Robinson, 1949; Orr, 1960; Bryant et al., 1972), which causes decreases in soil infiltration rates (Rauzi and Hanson, 1966) and subsequent increases in overland flow. Development of a vegetative buffer along each side of the stream is used to stabilize streambanks, thereby reducing bank erosion (Rogers and Schumm, 1991) and potentially reducing the input of nutrients to the stream channel by filtration of overland flow (Cooper et al., 1987; Parsons et al., 1994; Pearce et al., 1997) and through the retention of nutrients in the subsurface of the riparian zone (Jacobs and Gilliam, 1985; Lowrance, 1992; Nelson et al., 1995).

The water quality effects of specific BMP implementation, such as streambank fencing, on a basin scale are not well documented because of the typical occurrence of mixed land uses within a basin (Mostaghimi et al., 1989) and the implementation of numerous BMPs on a basin wide scale. The water quality effects of multiple BMP implementation on a basin scale have been studied (Walker et al., 1995; Edwards et al., 1996), and some have quantified the effects of specific BMP implementation, such as pipe outlet terracing (Lietman et al., 1997) and nutrient management (Koerkle and Gustafson-Minnich, 1997). The quantification of BMP effects on water quality are critical to agencies or programs concerned with water resources. For example, the Chesapeake Bay Program has developed a basin model that requires data on the effectiveness of BMP implementation on nutrient reduction loads to receiving waters (Chesapeake Bay Program, 1992). This model has led to the development of a tributary nutrient reduction strategy in states that are within the Chesapeake Bay Basin. These tributary strategies need quantifiable results from BMP implementations to determine the percentage reduction that could be realized with the development of farm management plans within the basin.

Thus, it is apparent that more studies are needed that quantify the effects of specific BMPs at a basin scale to better calibrate basin scale models. However, studies quantifying the effectiveness of BMPs have typically not had ideal study designs, primarily because of the lack of experimental controls. The ability to control agricultural practices over an extended time period on a basin wide scale is very difficult. One of the more reliable methods of potentially documenting BMP effectiveness in improving water quality is the paired basin monitoring design (Clausen and Spooner, 1993; Clausen et al., 1996). This approach requires the use of two relatively similar basins with one basin for a control and the other where treatment is applied. A calibration period between the two basins is required so that water quality relations between the basins can be documented prior to any BMP implementation.

A 6- to 10-yr study is being conducted in two small paired basins to determine the water quality effects of streambank fencing in pastureland within the treatment

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Abbreviations: BMP, best management practice; C-1, outlet of the control basin; T-1, outlet of the treatment basin; EPT, Ephemeroptera, Plecoptera, and Trichoptera; HBI, Hilsenhoff Biotic Index; NMP, National Monitoring Program.

basin. The paired basins are located in areas where agriculture accounts for approximately 85% of land use. These study basins were chosen because of their similarities in hydrology, geology, and the presence of a stable agricultural community that has historically not deviated significantly from year-to-year farming practices. This relative constancy is critical to the study because other changes in agricultural activities could make it difficult to detect changes in water quality caused by streambank fencing.

This article presents the paired basin design and the pretreatment calibration developed over a 3-yr period (October 1993–September 1996) between a control basin and a treatment basin where streambank fencing was installed during 1997. Fencing was installed in the treatment basin to exclude dairy animals and provide vegetated buffers of 3 to 4 m width on either side of the streambank. Data on land use, hydrology, and water quality are presented. Water quality data collected in the basins include chemical, physical, and biological components. Biological water quality and land use data were qualitatively compared between basins. Chemical and physical water quality data were statistically compared between basins to determine if pretreatment relations were significant. Statistical techniques were also used to determine the percentage change required in the relations during the posttreatment period to detect significant effects of streambank fencing on water quality.

Site Description

The two adjacent study basins, similar in land use, are located within the Mill Creek Basin of Lancaster County (Fig. 1). The control basin is 460 ha with 4.3 total stream km and 3.1 km of stream running through open pasture. The treatment basin is 370 ha with 4.5 total stream km and 3.2 km of stream running through open pasture. Land uses in the basins are about 85% agriculture, 10% residential/commercial, and 5% forested. Agriculture in the two basins, consisting of about 14 major farming operations, primarily involves crop production (mostly corn [*Zea mays* L.] and alfalfa [*Medicago sativa* L.]) and dairy farming. According to data collected from farmers from 1993 through 1996, N and P applications to the basins were higher in the control basin. The average annual N and P applications to the control basin were about 180 kg N ha^{-1} and 39 kg P

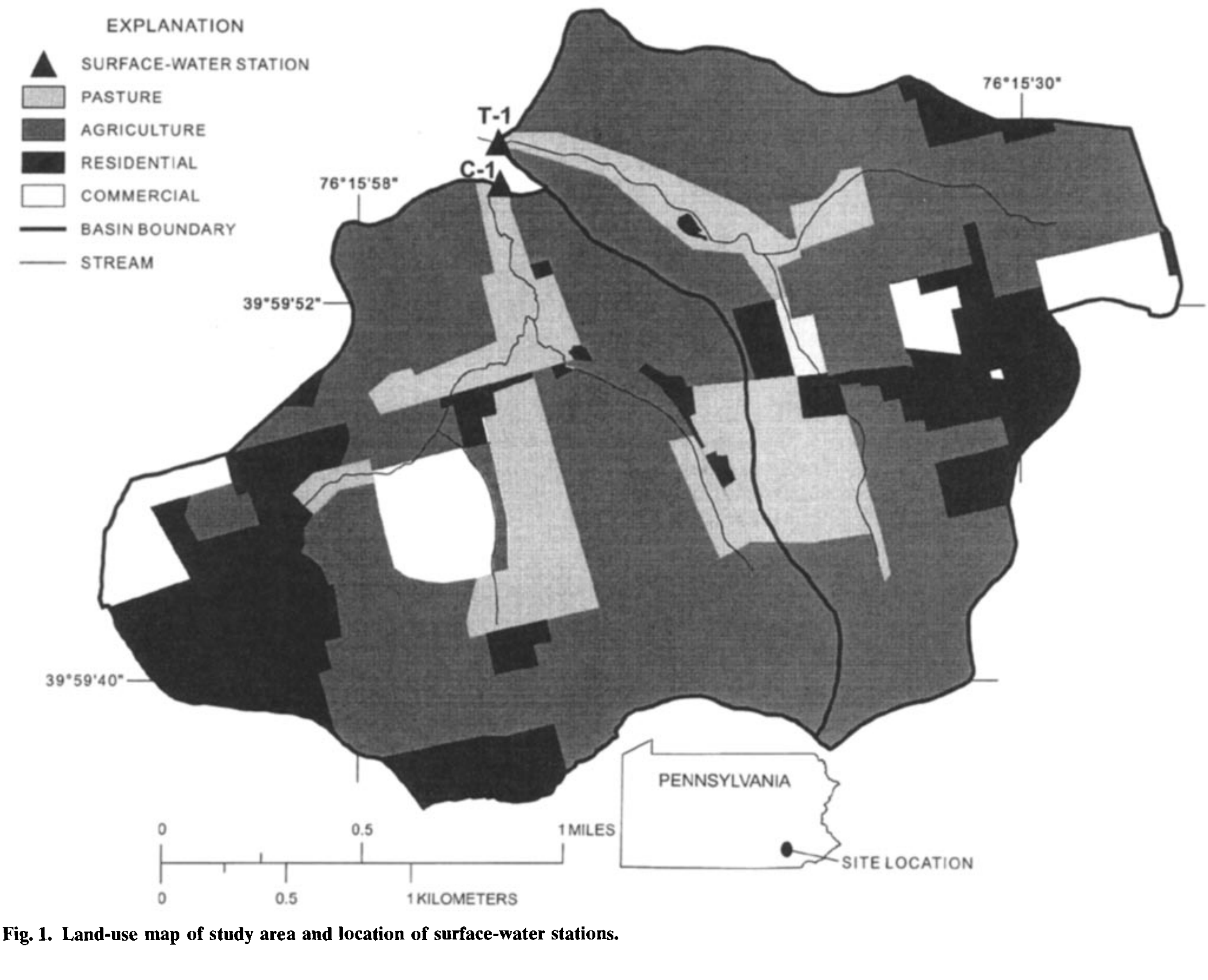

Fig. 1. Land-use map of study area and location of surface-water stations.

ha^{-1}. The average annual N and P applications to the treatment basin were about 140 kg N ha^{-1} and 36 kg P ha^{-1}. These nutrient loads to the basins include inorganic and organic fertilizer, in addition to manure deposited in pastures by dairy cows (*Bos taurus*). The amount of time that the cows were pastured was multiplied by the amount of manure (and nutrients) that a dairy cow typically releases on a daily basis (Pennsylvania Department of Environmental Resources, 1986).

During the pretreatment period, the number of dairy cows in the control and treatment basin were about 420 and 220, respectively, plus or minus 40 to 50 cows. The amount of pasture area in the control and treatment basin was approximately 24 and 34 ha, respectively. Data collected on a regular basis from the farm operators were the number of cows in each pasture and the number of hours the herd spent in the pasture per day. The average number of cows per day in pastured areas along streambanks was calculated by multiplying the number of cows in a pasture by the number of hours that the herd is in the pasture and then dividing this number by 24. The average numbers of cows per day for all pastures in the control and treatment basins were approximately 200 and 110, respectively. If cows were not pastured within the basin, for example during the winter months, then these days were not included in the average.

The basins are underlain by carbonate rock of the Conestoga Formation. This is an Ordovician-aged rock containing gray limestone with fine- to coarse-crystalline texture (Poth, 1977). The primary soils of the basins are of the Lehigh and Conestoga series (Custer, 1985). The Lehigh series is a fine-loamy, mixed, mesic Aquic Hapludalf and the Conestoga series is a fine-loamy, mixed, mesic Typic Hapludalf. Both soil series are well-drained and relatively deep and have slopes that range from 0 to 15%; most slopes in the basins range from 3 to 8% (Custer, 1985). Ground water well drilling logs report the depth to bedrock in the basins is anywhere from 2 to 7 m.

Annual precipitation averages 105 cm at a National Oceanic and Atmospheric Administration site about 3 km northeast of the basins (National Oceanic and Atmospheric Administration, 1994). One weighing bucket precipitation gauge was located at the outlet of the treatment basin to collect precipitation amounts continuously at 15-min intervals. For water years 1994 to 1996, annual precipitation amounts measured at this site were 131, 86.7, and 139 cm, respectively.

MATERIALS AND METHODS

Discharge at the outlets of control and treatment basins, hereinafter referred to as *C-1* and *T-1*, respectively, was monitored continuously by shaft encoders wired to data loggers recording data at 15-min intervals. Discharge was measured using either pygmy or double-A current meters at the outlets through a wide range of stages to develop a relation between stage and discharge. This relation was then used to convert the stage data to discharge.

Water samples for nutrient and suspended sediment analyses were collected at a fixed time interval and during storm events at C-1 and T-1. Samples were analyzed for dissolved forms of NH_3–N, NO_2–N, (NH_3 and organic)–N, (NO_2 and NO_3)–N, P, and ortho P. Analyses also included total forms of (NH_3 and organic)–N and P, and suspended sediment. Fixed time interval (grab) samples were collected every 10 d (regardless of flow conditions) from April through November and on a monthly basis during a low flow period from December through March. The more intensive sampling from April through November coincided with the typical period when cows are pastured in south-central Pennsylvania. Fixed time samples were collected by hand at the downstream side of the cement v-notch weir used to control flow at the outlets of both basins. At the time of sample collection, pH, temperature, dissolved oxygen, and specific conductance were recorded using a multiparameter probe placed at the upstream side of the weir after the grab samples were collected. Grab samples were filtered in the field through a 0.45-micron pore size filter that was prerinsed with deionized water. Samples to quantify the abundance of fecal streptococcus bacteria were collected once per month during low flow conditions, and enumeration analysis was performed according to techniques described by Ehlke et al. (1987) for the membrane filter method and immediate incubation test. Storm samples were collected with an automated sampler having a 72 bottle capacity. Sample collection during a storm event was initiated by a float switch that turned samplers on at a specific stage. This stage height was adjusted based on current flow conditions. After initialization, samples were collected every 15 min until either the 72 bottles were filled or the stage dropped below the point at which initialization occurred. Storm samples were retrieved within a day of event completion and chilled prior to sample processing. After defining the storm interval so that similar time intervals and parts of the hydrograph were used for both C-1 and T-1, samples from the storms were composited into one storm sample per event. Aliquots pipeted from bottles were flow weighted so the composite sample represented mean conditions for the storm event. Chemical and suspended sediment analyses were done on composited samples.

Chilled samples were shipped to the USGS National Water Quality Laboratory in Arvada, CO, for nutrient analysis. Analyses were performed according to techniques described in Fishman and Friedman (1989). Suspended sediment concentration analyses were conducted by the USGS Sediment Laboratory in Pennsylvania through water year 1995 and thereafter at the USGS Sediment Laboratory in Kentucky. Both sediment laboratories used procedures described by Guy (1969) to determine suspended sediment concentrations.

The benthic-macroinvertebrate stream community was sampled at the outlets of the treatment and control basins semi-annually in May and September. A pool and a riffle/run area were sampled at each site. The bottom substrate of a 1-m^2 area was vigorously agitated for 1 min by the foot activity of field personnel. Dislodged invertebrates were trapped with a 1-m^2 screen with a 600-μ mesh placed directly downstream of the agitated area. Samples were preserved in denatured alcohol and shipped to Lotic Environmental Consultants[1] for invertebrate identification to the genus level and calculation of six biological metrics to characterize health of the stream system.

The six macroinvertebrates metrics calculated were:

1. Taxa richness, a measurement of the total number of taxa present, with higher numbers generally reflecting a more healthy macroinvertebrate community.

[1] The use of company names does not imply endorsement by the U.S. Geological Survey of the products named or criticisms of similar companies not mentioned.

2. Hilsenhoff biotic index, which summarizes the overall pollution tolerance of the benthic arthropod community. Range of values is from 0 to 10 with an increase in numbers indicating a decrease in water quality.

3. Ratio of scrapers to filter feeders, which provides an indication of the periphyton community composition. Scrapers increase with increased abundance of diatoms and decrease with increased abundance of filamentous algae and aquatic mosses. Because filamentous algae and aquatic mosses are utilized by filter feeders and are usually indicators of organic enrichment, a decrease in the ratio of scrapers to filter feeders is usually indicative of decreased water quality.

4. Ratio of Ephemeroptera, Plecoptera, and Trichoptera (EPT) to Chironomidae abundance; EPT are groups of invertebrates that are generally pollution sensitive. Chironomidae are tolerant of pollution. A ratio near or above 1.0 indicates a healthy community.

5. Percent dominant taxa, a measure of the percent of dominant taxa relative to the total number of organisms. As this number increases, community health typically decreases.

6. The EPT index, an increase in this index is generally indicative of increased water quality (Plafkin et al., 1989).

Stream habitat was qualitatively characterized during benthic-macroinvertebrate sampling using rapid bioassessment protocols that numerically rank such stream characteristics as bank stability, bottom scour, and pool/riffle ratios (Plafkin et al., 1989). Because of the subjective nature of this numerical system, the same field person performed the stream assessments throughout the study. Habitat was visually documented by photography over the course of the pretreatment period.

Annual loads of nutrients and suspended sediment for the two basins were estimated by least-squares regression equations. Regressions were developed separately for nonstorm periods and storm events. Regression equations for constituent concentrations were developed from discharge, seasonal components, and time. Models were selected based on the highest adjusted R^2, residuals plots to detect trends, and all F-values had to exceed the value for the F distribution for the appropriate degrees of freedom and $\alpha = 0.05$.

For nonstorm periods, constituent loads were estimated on a daily basis using the daily mean discharge data available from T-1 and C-1. Storm events were separated from the discharge record prior to nonstorm analysis. Constituent concentrations were estimated using the regressions that were developed. Estimates of constituent concentrations were based on approximately 75 low flow samples collected at both T-1 and C-1. Estimated concentrations were multiplied by the daily mean discharge to estimate daily loads. Loads were summed for all nonstorm periods.

Stormflow loads for nutrients and suspended sediment were estimated using mean discharge and mean constituent concentration for sampled storms. Because storms could not be sampled more than 18 h after the initial sample collection (the samplers hold 72 bottles and samples are collected every 15 min), storm intervals did not exceed 18 h for any one storm event. The mean discharge concentration relation developed for sampled storms ($n = 59$) using regression analysis was used to estimate concentrations for unsampled storms ($n = 35$). Mean discharge was calculated for unsampled storms using 15-min continuous stage data for the sites. This mean discharge was applied to the predicted concentration to estimate constituent loads for unsampled storms. For unsampled storms, any rise in stage equal to or greater than 8 cm was considered a storm event. The beginning of the storm was visually determined from the hydrograph. The end of the event was considered the stage on the recession part of the hydrograph when the initiation (beginning) stage was reached (unless this was greater than 18 h from the initiation time). Increases in stage caused by snowmelt events were analyzed separately by subsetting storm events sampled during snowmelts and using these regression relations to estimate loads for nonsampled snowmelt events.

Constituent loads on an annual basis for each basin were estimated by summing nonstorm and stormflow loads. The multiple regression technique used discharge and season to calculate loads and is similar to methods used by Andrews (1978) and Lystrom et al. (1978). However, it should be noted that these techniques tend to underestimate constituents loads in some cases (Ferguson, 1986; Koch and Smillie, 1986). The annual load data for the constituents were divided by the basin drainage areas to determine constituent yields. Percentage of total yield in stormflow was determined by summing sampled and unsampled storm yields and dividing this by total yield. The remaining yield was thus attributed to nonstormflow periods.

Seasonality in discharge, concentration and yield of N and P, and suspended sediment for both basins was quantitatively and qualitatively compared. Median values for each month were plotted to graphically present trends. Statistical differences were tested by separating data into dormant and growing seasons. Data were ranked and significant differences between seasons were detected using Tukey's multiple comparison tests (Helsel and Hirsch, 1992).

Statistical methods relating water quality characteristics between basins involved development of least-squares regression equations for paired observations between basins. These regression models were developed for all constituents concentrations and yields measured in fixed time and stormflow samples. Regression equations were developed from stormflow samples concurrently collected and fixed time samples collected on the same day at each basin. Transformations of the raw data were required with some models to normalize residuals. To reduce effects of high leverage observations (Helsel and Hirsch, 1992) when generating regression equations, high flow data were removed before development of regression equations for fixed time (10-d and monthly interval) data. Any samples that were collected above the 90th percentile of flow data were deleted from the data set. Thus, from this point onward, analysis of fixed time samples is referred to as low flow samples. Statistical analysis was performed using Statistical Analysis Systems software (SAS Institute, 1982).

The regression equations that were developed between basins for the calibration period were used to determine the percentage change (pc) required in the relations to detect a significant effect of streambank fencing on water quality. A technique derived from Clausen and Spooner (1993) was used to determine the pc required to detect a change in the posttreatment period relative to the calibration period. The method was originally developed to determine the number of samples required in the posttreatment period to detect some change in the calibration relation. The form of the equations are

$$\frac{S_{xy}^2}{d^2} = \frac{n_1 n_2}{n_1 + n_2}\left\{\frac{1}{F\left(1 + \dfrac{F}{n_1 + n_2 - 2}\right)}\right\} \qquad [1]$$

$$pc = \left(\frac{d}{\bar{x}}\right) \times 100 \qquad [2]$$

Equation [1] requires the input of the number of samples in

Table 1. Mean biological indices for T-1 and C-1 for water years 1994 through 1996. Data are means for six sampling events.

Level of water-quality impact	Taxa richness	Hilsenhoff biotic index	Ratio of scrapers to filter feeders	Ratio of EPT to Chironomidae abundances	Percent dominant taxa	EPT index
Nonimpacted†	≥20 C-1 = 23 T-1 = 26	0.00–6.50 C-1 = 5.4 T-1 = 6.4	0.8	2.0	≤20	6
Slightly impacted‡	11 to <20	6.51–8.50	0.4 to <0.8	0.6 to <2.0 C-1 = 0.61 T-1 = 0.67	>20 to 35 C-1 = 30 T-1 = 29	4 to <6
Moderately to severely impacted§	1 to <11	8.51–10.0	0.0 to <0.4 C-1 = 0.13 C-1 = 0.11	0.2 to 0.6	>35 to 50	1 to <4 C-1 = 3 T-1 = 3

† This range would indicate that, according to this metric, the water quality of the stream is excellent and the macroinvertebrate community is diverse (Bode et al., 1993).
‡ This range would indicate that, according to this metric, the water quality of the stream is good and the macroinvertebrate community is slightly but significantly altered from the pristine state (Bode et al., 1993).
§ This range would indicate that, according to this metric, the water quality is fair to poor, and the macroinvertebrate community is altered a large extent from the pristine state (Bode et al., 1993).

the calibration and posttreatment period (n_1 and n_2, respectively), an F-value (for the variance ratio at 1 and ($n_1 + n_2 - 3$) degrees of freedom) based on the confidence level desired, and the residual variance of the calibration regression equation (S^2xy). For this study, $\alpha = 0.05$ and it was assumed that the number of observations during the calibration period would equal the number during the posttreatment period. Solving Eq. [1] for d, this product was divided by the mean of the control watershed data ($\bar{x}$) to calculate pc. The equation was thus solved for the pc that would be needed to have 95% confidence the relation change from the calibration to the posttreatment period was significant.

RESULTS AND DISCUSSION

Benthic Macroinvertebrates and Physical Habitat

For each of the six metrics, the impact level in the three ranges classified was the same for the treatment and control outlets (Table 1). Similar structure in benthic communities was also evident from similarity indices. Calculated values for the Jacard Coefficient of Community Similarity (Plafkin et al., 1989) and the Similarity Index (Odum, 1983) between sites were 0.55 and 0.70, respectively. These indices are based on the presence or absence of taxa, and calculations were based on all six samples collected at each site. The overall similarities between the two sites allows for documentation of any effects of streambank fencing on the benthic-macroinvertebrate community at T-1. The most common benthic macroinvertebrates identified at C-1 and T-1 were *Gammarus* spp. (amphipods) and *Baetis* spp. (mayflies), respectively. The most common genus found in both basins was *Orthocladius* spp. (midges).

The six metrics used to characterize the benthic-macroinvertebrate communities indicated different levels of community health. For taxa richness, the Hilsenhoff biotic index (HBI), and the EPT index, the level of water quality impact was based on ranges modified from Bode et al. (1993). For the other three indices, levels of impact were based on best professional judgment (Mike Bilger, U.S. Geological Survey, oral communication, 1993) because there was a lack of a reference site for comparisons. Both taxa richness and the HBI indicated nonimpacted sites. The taxa richness values are indicative of sites with either good water quality and/or diverse habitat (Plafkin et al., 1989). The two metrics that indicated moderately to severely impacted sites were the ratio of scrapers to filter feeders and the EPT index. The scrapers/filter feeders ratio showed that scrapers were uncommon, which indicated that diatoms were not prevalent in the system. The EPT index was low, and relative to Chironomids, the EPT levels indicated slightly impacted streams. The low values for the EPT index signified either a pollution problem in the system or a lack of habitat.

Qualitative assessment of physical habitat at time of benthic-macroinvertebrate sampling indicated many similarities at the outlets of control and treatment basins; the only difference was related to bank stability. Streambanks appeared to be more stable at C-1 because of a higher percentage area covered with vegetation, thus reducing erosion probability during flood events. Bottom substrate at both sites was rubble and gravel; these materials were embedded in finer particles of silt and sand. Bottom scouring and deposition in pools was

Table 2. Mean annual yields for T-1 and C-1 estimated using data collected from 1 Oct. 1993 through 30 Sept. 1996, and the percentage of the annual yield attributable to nonstormflow and stormflow.

	T-1			C-1		
Constituent	Annual yield	Percent nonstormflow	Percent stormflow	Annual yield	Percent nonstormflow	Percent stormflow
	kg ha^{-1}			kg ha^{-1}		
Dissolved NH_3–N	0.92	21	79	0.93	20	80
Dissolved NO_3–N	48	91	9	52	89	11
Total N	54	83	17	58	82	18
Total P	2.9	10	90	2.7	9	91
Dissolved P	0.99	17	83	1.1	17	83
Suspended sediment	2600	5	95	2700	10	90

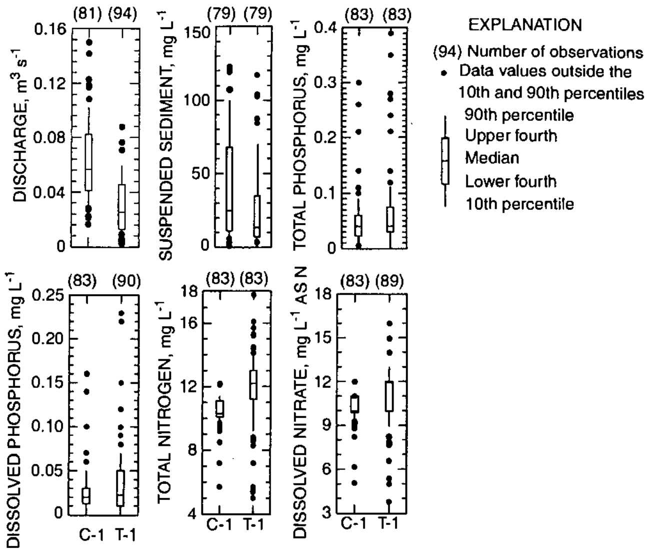

Fig. 2. Distribution of discharge and constituent concentrations for low flow samples from T-1 and C-1 for water years 1994–1996.

apparent, but not significant since depth in pools and riffles was adequate. Streamside vegetation was grasses where vegetation existed.

Nutrients and Sediment

Estimated annual yields for nutrients and sediment for water years 1994 through 1996 were similar for the two basins (Table 2). Nutrient and suspended sediment yields were comparable to those reported by Lietman et al. (1983) and Unangst (1992) for other small agricultural drainage basins located in Lancaster County, Pennsylvania. The annual yields reported by Lietman et al. (1983) for two basins predominantly used for either pasture or corn crops were 42 and 56 kg ha^{-1} for total N and 16 and 1.9 kg ha^{-1} for total P. Total N and P yields were 2 to 3 and 18 to 150 times higher, respectively, than the yields for a forested site upstream of the agricultural

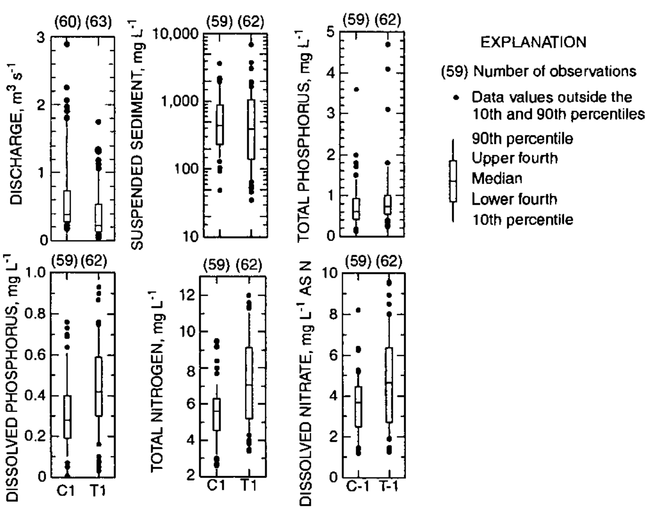

Fig. 3. Distribution of mean discharge and mean water weighted concentrations for stormflow samples from T-1 and C-1 for water years 1994–1996.

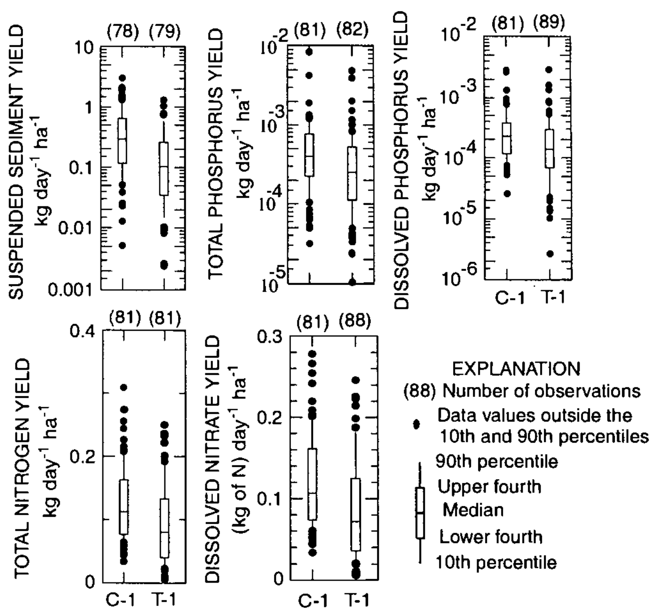

Fig. 4. Distribution of constituent yields for low-flow samples at T-1 and C-1 for water years 1994–1996. Yields on a per day basis were calculated by multiplying concentration by flow, assuming that the instantaneous flow was the mean flow for the day.

sites reported by Lietman et al. (1983). Percentage of total N yield as NO_3–N for T-1 and C-1 was about 90%. Studies by Lietman et al. (1983) and Unangst (1992) reported NO_3–N to total N percentages of 73 to 94%. For T-1 and C-1, approximately 80 to 85% of the total N yield was attributable to nonstormflow conditions. Conversely, about 90% of the total P yield was attributable to stormflow. Approximately 60 to 65% of the total P yield for T-1 and C-1 was in the suspended form; about 95% of the suspended P yield was transported during stormflow. Ninety to 95% of suspended sediment yield was attributable to stormflow for T-1 and C-1. Lietman et al. (1983) reported 85 to 90% of the suspended sediment yield was attributable to stormflow. The higher percentage of total P yield and suspended sediment transported during stormflow should increase the probability of detecting significant changes in yields at T-1 after the streambank fencing is installed because the vegetative buffer strip would help trap sediment and associated P before reaching the stream channel. This would be more likely during low intensity storm events.

The concentrations of nutrients and suspended sediment for low flow and stormflow were somewhat different for T-1 and C-1 (Fig. 2 and 3). Median concentrations of total N were higher for T-1 for both low flow and stormflow samples. For low flow, the 1.9 mg L^{-1} difference in median values of total N was attributable to higher NO_3–N concentrations for T-1. The dilution of NO_3–N during storm events helped to decrease the difference (1.4 mg L^{-1}) in median values of total N for stormflow between T-1 and C-1. Other forms of N, such as NH_3 and organic N increased in concentration from low flow to stormflow for both basins. Median concentrations of total and dissolved P were identical for both basins during low flow; however, median concentrations of total and dissolved P for stormflow were about 0.1

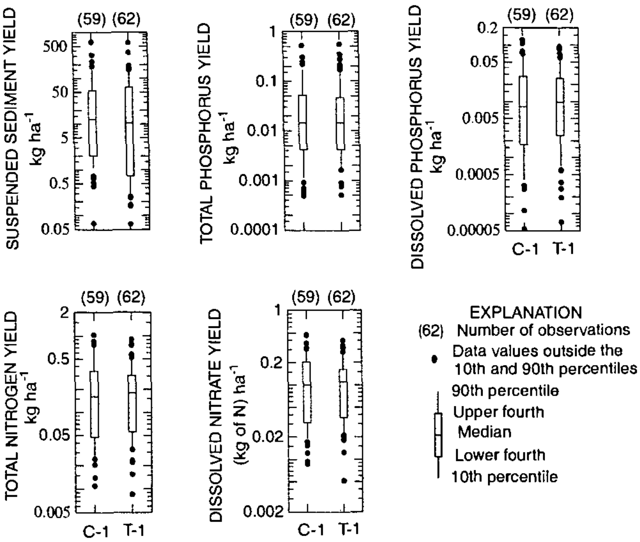

Fig. 5. Distribution of constituent yields for flow-weighted composite stormflow samples at T-1 and C-1 for water years 1994–1996.

mg L^{-1} higher at T-1. The suspended portion of P during storms was similar. The suspended sediment concentration was higher for C-1 for both low flow and stormflow. The higher concentrations of suspended sediment during low flow were in samples collected when cows were in the stream channel upstream of the sampling point.

Discharge measured at C-1 during low flow and stormflow was about twice as high as that measured at T-1 (Fig. 2 and 3). The higher discharge caused yields for total P and N and suspended sediment at C-1 to exceed those measured at T-1 for low flow periods (Fig. 4). Stormflow yields were very similar between basins as is evident by the medians and the distribution overlap in quartile data (Fig. 5). The yields for low flow and stormflow samples are not comparable because stormflow yields were calculated per storm event, and low flow yields were estimated on a daily basis. Median time interval for storms for both basins was 8 h.

Seasonality in constituents for both basins was most evident for total N concentrations and yields. Yields of total N for low flow samples for T-1 and stormflow samples for C-1 and T-1 were significantly higher during the dormant season (Fig. 6). Total N concentrations were significantly higher during the dormant season at T-1 for low flow samples and at C-1 for stormflow samples. Variation in total N was primarily because of seasonal variations in NO_3–N. Significantly higher flows during the dormant season also contributed to the significant difference in total N yields. The significantly lower total N concentrations for T-1 during the growing season were consistent with other studies for Lancaster County (Koerkle et al., 1996). Concentration of total N at C-1 for low flow did not vary with discharge. The difference between the 25th and 75th percentile for total N at C-1and T-1 was about 1 and 4 mg L^{-1}, respectively. This indicates that N source for C-1 at low flow was relatively constant and not seasonally dependent. During a storm event for any one basin, percentage of base flow (or flow derived from ground water discharge) generally decreases with an increase in total flow, with a subsequent increase in the percentage of overland flow and interflow (Bachman et al., 1998). Thus, for stormflow, the proportion of N derived from soil zones increases relative to the amount of N from ground water. Plant uptake during the growing season decreases available N storage in soil and potentially could cause decreased transport of N to stream systems.

Suspended sediment yield was significantly higher during the dormant season at T-1 for low flow (Fig. 6). This is partially due to increased vegetative cover and reduced potential for soil erosion, but it was also flow related since the suspended sediment concentrations did not show significant seasonal differences. Concentra-

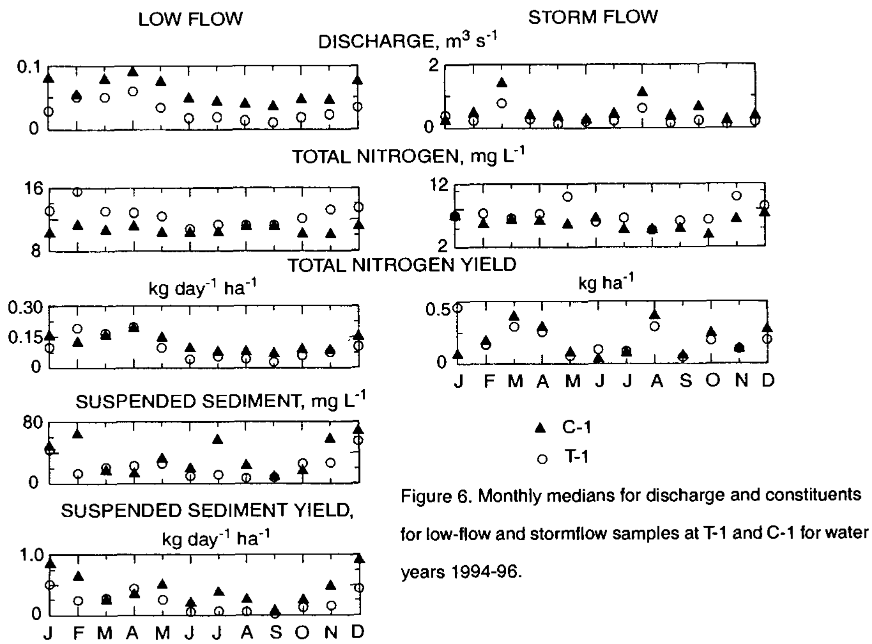

Fig. 6. Monthly medians for discharge and constituents for low flow and stormflow samples at T-1 and C-1 for water years 1994–1996.

Table 3. Equations generated for T-1 data regressed against C-1 data for water years 1994 through 1996 for low-flow samples and the percentage change required in the untransformed parameter mean that would be necessary to detect a significant effect of fencing during the postfencing period relative to the prefencing period. Shaded rows indicate insignificant models.

Characteristic or constituent	No. of samples	Regression model	Adj. R^2	Percentage change required
$\log_{10}$(instantaneous discharge, $m^3\ s^{-1}$)	75	$-0.514 + 1.61\log_{10}X$	0.84	2.14/2.19
Temperature, degrees Celsius	74	$-6.54 + 1.62X$	0.95	3.28
Specific conductance, $\mu S\ cm^{-1}$	76	$217.2 + 0.633X$	0.17	2.44
$\log_{10}$[Fecal Streptococcus, col (100 mL)$^{-1}$]	33	$2.25 + 0.359\log_{10}X$	0.05	59.1/146
Concentration, mg L^{-1}				
Dissolved oxygen	71	$-1.314 + 1.128X$	0.75	3.32
$\log_{10}$(dissolved NH_3–N)	76	$-0.438 + 0.665\log_{10}X$	0.40	15.4/18.3
$\log_{10}$[dissolved (NH_3 + organic N) − N]	76	$-0.352 + 0.184\log_{10}X$	0.01	13.6/15.8
$\log_{10}$[total (NH_3 + organic N) − N]	76	$-0.103 + 0.402\log_{10}X$	0.14	10.8/12.1
$\log_{10}$(dissolved NO_2–N)	75	$-0.361 + 0.553\log_{10}X$	0.45	9.5/10.4
Dissolved NO_3–N	75	$0.706 + 1.03X$	0.26	5.87
Total N	76	$2.24 + 0.912X$	0.16	6.16
$\log_{10}$(dissolved P)	76	$-0.512 + 0.632\log_{10}X$	0.29	21.3/27.1
$\log_{10}$(suspended P)	43	$-0.797 + 0.476\log_{10}X$	0.16	26.6/36.3
$\log_{10}$(total P)	76	$-0.574 + 0.546\log_{10}X$	0.18	23.8/31.2
(Suspended sediment)$^{0.5}$	73	$2.62 + 0.348X^{0.5}$	0.15	23.9/27.1
Yield, kg $d^{-1}\ ha^{-1}$				
$\log_{10}$(dissolved NH_3–N)	75	$-0.276 + 0.973\log_{10}X$	0.51	26.5/36.1
$\log_{10}$[dissolved (NH_3 + organic N) − N]	75	$-1.07 + 0.599\log_{10}X$	0.20	36.9/58.5
$\log_{10}$[total (NH_3 + organic N) − N]	75	$-0.838 + 0.662\log_{10}X$	0.35	47.8/91.6
$\log_{10}$(dissolved NO_2–N)	74	$-0.682 + 0.780\log_{10}X$	0.57	14.7/17.2
Dissolved NO_3–N	74	$-0.0374 + 1.089X$	0.83	7.30
Total N	75	$-0.0375 + 1.098X$	0.83	7.41
$\log_{10}$(dissolved P)	75	$-1.29 + 0.685\log_{10}X$	0.31	28.2/39.3
$\log_{10}$(suspended P)	75	$-1.08 + 0.764\log_{10}X$	0.37	39.7/65.9
$\log_{10}$(total P)	75	$-1.35 + 0.655\log_{10}X$	0.26	32.6/48.5
(Suspended sediment)$^{0.5}$	72	$0.136 + 0.417X^{0.5}$	0.27	22.7/25.7

tions and yields of P during low flow and stormflow conditions do not show any significant seasonal differences for either basin.

Least-squares regression equations of chemical and physical constituents were generated for low flow and stormflow samples to determine the relation between T-1 and C-1 before fence installation (Tables 3 and 4) and the percentage change required during the posttreatment period to detect a significant effect of fencing on water quality. Regression equations were generated for concentration and yield data. To determine yields for instantaneous low flow samples, it was assumed that discharge at time of sample collection was constant over a 24-h period and this was used to determine a daily load for that sample. For sampled storms, the actual mean discharge over the storm period was used along with mean concentration for the storm period to calculate a storm load. All the models were significant at a 95% confidence interval except for two models developed for low flow samples. The constituent that will

Table 4. Equations generated for T-1 data regressed against C-1 data for water years 1994 through 1996 for stormflow samples and the percentage change required in the untransformed parameter mean that would be necessary to detect a significant effect of fencing during the postfencing period relative to the prefencing period.

Characteristic or constituent	No. of samples	Regression model	Adj. R^2	Percentage change required
Mean discharge, $m^3\ s^{-1}$	60	$-0.0216 + 0.0158X$	0.89	6.41
Concentration, mg L^{-1}				
Dissolved NH_3–N	59	$0.123 + 0.829X$	0.65	21.6
$\log_{10}$[dissolved (NH_3 + organic N) − N]	59	$0.119 + 0.397\log_{10}X$	0.16	14.7/17.1
$\log_{10}$[total (NH_3 + organic N) − N]	59	$0.252 + 0.414\log_{10}X$	0.23	17.7/21.5
(Dissolved NO_2–N)$^{-1}$	59	$1.51 + 0.660X^{-1}$	0.47	10.2/12.9
$\log_{10}$(dissolved NO_3–N)	59	$0.127 + 0.973\log_{10}X$	0.64	13.8/16.0
Total N	59	$3.020 + 0.767X$	0.23	14.1
Dissolved P	59	$0.134 + 0.972X$	0.60	17.0
$\log_{10}$(suspended P)	59	$-0.147 + 0.619\log_{10}X$	0.51	17.8/21.6
$\log_{10}$(total P)	59	$0.0184 + 0.626\log_{10}X$	0.47	9.24/16.2
(Suspended sediment)$^{0.5}$	58	$-6.58 + 1.21X^{0.5}$	0.86	16.6/18.2
Yield, kg ha^{-1}				
$\log_{10}$(dissolved NH_3–N)	59	$-0.498 + 0.774\log_{10}X$	0.74	26.5/36.1
$\log_{10}$(dissolved [NH_3 + organic N) − N]	59	$-0.271 + 0.8231\log_{10}X$	0.90	14.8/17.4
$\log_{10}$(total (NH_3 + organic N) − N]	59	$-0.208 + 0.866\log_{10}X$	0.88	16.9/20.3
$\log_{10}$(dissolved NO_2–N)	59	$-0.780 + 0.717\log_{10}X$	0.80	15.7/18.7
$\log_{10}$(dissolved NO_3–N)	59	$-0.199 + 0.841\log_{10}X$	0.76	17.6/18.7
Total N	59	$0.0275 + 0.774X$	0.86	11.7
(Dissolved P)$^{0.5}$	59	$0.0152 + 0.829X^{0.5}$	0.94	12.3/13.1
$\log_{10}$(suspended P)	59	$-0.355 + 0.876\log_{10}X$	0.84	24.6/32.6
$\log_{10}$(total P)	59	$-0.215 + 0.904\log_{10}X$	0.89	19.5/32.6
$\log_{10}$(suspended sediment)	58	$-0.373 + 1.10\log_{10}X$	0.87	21.2/26.9

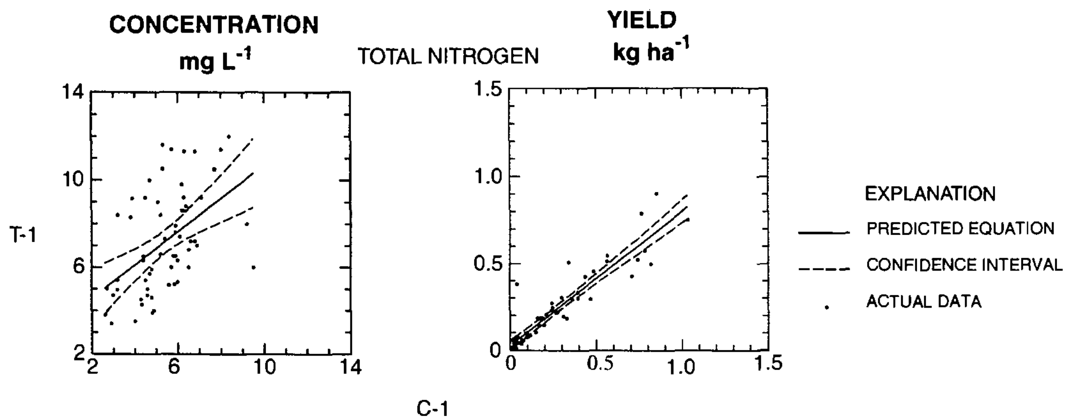

Fig. 7. Relation between total N concentrations and yields of flow weighted composite stormflow samples collected concurrently during water years 1994–1996 at the outlet of the treatment (T-1) and control (C-1) basins. The line for the predicted regression equations and the upper and lower 95% confidence intervals are given.

require the largest percent change to detect a significant effect of streambank fencing is the fecal streptococcus colonies in 100 mL of sample water. This is primarily because the variability in the fecal streptococcus relation between basins before fencing was relatively high. For nutrients, there was a a clear difference between the percentage change required in low flow and stormflow samples for any one constituent. Generally, the percent changes required for N forms were less while the percent changes required for P forms were greater for low flow relative to stormflow samples. For low flow samples, NO_3–N and total N concentrations and yields would require only a 6 to 7% change in the difference between means for T-1 and C-1 to detect a significant change; whereas for stormflow samples, a 12 to 18% change would be required. Conversely, for low flow samples, total P concentration and yield would require a 24 and 33% change, respectively, in the difference between the means for T-1 and C-1 to detect a significant change; whereas, for stormflow, a 9 and 20% change would be required. Regression equations developed between the two outlets can be used in a general sense to visually identify the magnitude of the effect required to cause a significant change if confidence intervals are plotted along with the predictive equation (Fig. 7). Any deviation of regression relation beyond limits of the upper and lower 95% confidence intervals would generally indicate a significant change from pretreatment to the posttreatment period.

Given results from other studies, for low flow samples it is possible that significant changes will occur for total N and nitrate concentrations and yields from the pretreatment to posttreatment period. However, NO_3–N annual yields for this study may overwhelm the nitrate withholding capacity of the streamside buffer and/or the denitrification effects that other studies have documented. Many riparian zone studies in agricultural areas have detected significant reductions in either shallow ground water or base flow NO_3–N concentrations because of riparian zone effects (Lowrance et al., 1984; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Haycock and Pinay, 1993; Jordan et al., 1993; Osborne and Kovacic, 1993). Results from these studies generally indicate about a 90% reduction in nitrate concentrations in subsurface flow draining through riparian zones. For C-1 and T-1, approximately 90% of the nitrate yield is attributable to low flow periods, and 89% of total N yield is attributable to nitrate (Table 2). It should be noted that some of these studies had substantially lower yields of total N to reduce. The annual yield of NO_3–N for Peterjohn and Correll (1984) and Lowrance et al. (1984) were about 2.4 and 13 kg ha^{-1}, respectively. This is well below the 48 to 52 kg ha^{-1} annual yield of NO_3–N estimated for C-1 and T-1. A 7% reduction in the annual yield of NO_3–N for T-1 would be equal to approximately 2.4 kg ha^{-1}. Assuming that 3 km of stream channel are fenced with a 2- to 3-m buffer on each side of the channel during the posttreatment period of this study, about 2 ha will be revegetated and inaccessible to pastured cows. The total drainage area of T-1 is 360 ha, so about 1200 kg of NO_3–N would have to be retained by the vegetation within the fence or be denitrified on an annual basis within these 2.4 ha for a 7% reduction in the present annual yield of NO_3–N.

Peterjohn and Correll (1984) also detected a 37% increase in ammonium, a 13% decrease in organic N, and a 33% decrease in total P in shallow ground water within a 19 m wide riparian zone. For this study, the fenced area is 2 to 3 m on either side of the stream. Reductions of the magnitude found by Peterjohn and Correll would indicate a significant difference in low flow samples for the constituents (NH_3–N, organic N, and total P) in question for this study (Table 3), but the buffer strip size for the Peterjohn and Correll study was 6 to 9 times wider than this study. It is possible that the buffer strip size for this study is not sufficient to reduce nutrient loads to the stream system during low flow conditions. However, instream processes, such as the seasonal vegetative uptake of dissolved nutrients, which does not occur in ground water systems, could affect the low flow component for nutrient constituents. Nitrogen can be removed by aquatic vegetation as nitrate and ammonium, whereas P is removed as a monovalent or divalent phosphate ion (Salisbury and Ross, 1985). Plant decay also can change instream nutrient concentrations.

For stormflow, past studies indicate constituents associated with suspended materials should be trapped within a vegetated buffer, but the effects of a buffer on dissolved constituents are more questionable. Surface runoff studies have documented that 67 to 94% of sediment transported from upland agricultural areas was trapped by vegetative strips of varying width and plant community structure (Young et al., 1980; Peterjohn and Correll, 1984; Cooper et al., 1987; Dillaha et al., 1989). It follows that constituents associated with suspended materials, such as total P and organic N, would also be trapped. Percentage reductions of the magnitude identified by these surface runoff studies would result in a significant reduction for this study. However, trapping efficiency may decrease over time as sediment is accumulated, and the 2- to 3-m wide buffer is narrower than any of the studies cited. Also, for high intensity storms, gully erosion and the resulting sediment movement would limit trapping efficiency of a vegetative strip. For dissolved constituents, surface runoff studies have identified reductions in ammonium and phosphate ions from upland agricultural areas of about 60 to 75% (Young et al., 1980; Peterjohn and Correll, 1984). Reductions of this magnitude would indicate a significant effect of streambank fencing in the treatment basin relative to the control basin. However, for these surface runoff studies, it must be noted that the effects of vegetative strips is isolated to only runoff concentrations. Instream concentrations will reflect the effect on surface runoff, but surface runoff during a storm event is only a partial component of total stream discharge. Thus, effects of buffers on surface runoff will be diluted by other contributions to stormflow such as interflow and base flow.

CONCLUSIONS

Results from the calibration period indicated that control and treatment basins responded comparably to environmental factors. Benthic-macroinvertebrate communities were similar in composition and community health. Concentrations and yields of nutrients and suspended sediment were similar at the outlet of the basins. Linear regression models generated for low flow and stormflow samples for nutrients and suspended sediment indicated the relations between basins were not significantly different. The calibration of paired basins prior to BMP implementation is critical to identifying effects of implementation of land management practices, in this case, streambank fencing and subsequent development of a vegetative buffer strip. The relations developed for chemical and physical water quality constituents can be used to quantify effects that may occur during the posttreatment period.

The detection of significant effects of streambank fencing in the treatment basin will depend on the media in question. Significant deviations from the calibration to posttreatment period are unlikely for nutrient species during low flow conditions; whereas, for stormflow, suspended sediment and constituents associated with the suspended phase, such as P and organic N, are more likely to show significant deviations from the calibration relation. Changes in the benthic-macroinvertebrate community are likely due to changes in the physical habitat associated with vegetation growth near and within the stream channel.

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REFERENCES

Alderfer, R.B., and R.R. Robinson. 1949. Runoff from pastures in relation to grazing intensity and soil compaction. J. Am. Soc. Agron. 39:948–958.

Andrews, E.D. 1978. Present and potential sediment yields in the Yampa River Basin, Colorado and Wyoming. U.S. Geol. Surv. Water-Resour. Invest. Rep. 78-105.

Bachman, L.J., B. Lindsey, J. Brakebill, and D.S. Powars. 1998. Ground-water discharge and Base-flow nitrate loads of nontidal streams, and their relation to a hydrogeomorphic classification of the Chesapeake Bay watershed, middle Atlantic coast. U.S. Geol. Surv. Water-Resour. Invest. Rep. 98-4059.

Bode, R.W., M.A. Novak, and L.E. Abele. 1993. 20 year trends in water quality of rivers and streams in New York state based on macroinvertebrate data 1972–1992. New York Dep. of Environmental Conservation.

Bryant, F.T., R.E. Blaser, and J.R. Peterson. 1972. Effect of trampling by cattle on bluegrass yield and soil compaction of a Meadowville Loam. Agron. J. 64:331–334.

Chesapeake Bay Program. 1992. Progress report of the baywide nutrient reduction reevaluation. USEPA, Annapolis, MD.

Clausen, J.C., W.E. Jokela, F.I. Potter III, and J.W. Williams. 1996. Paired watershed comparison of tillage effects on runoff, sediment, and pesticide losses. J. Environ. Qual. 25:1000–1007.

Clausen, J.C., and J. Spooner. 1993. Paired watershed study design. USEPA Rep. 841-F-93-009. USEPA, Office of Water, Washington, DC.

Cooper, J.R., J.W. Gilliam, R.B. Daniels, and W.P. Robarge. 1987. Riparian areas as filters for agricultural sediment. Soil Sci. Soc. Am. J. 51:416–420.

Custer, B.H. 1985. Soil survey of Lancaster county, Pennsylvania. USDA-SCS, Washington, DC.

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.

Edwards, D.R., T.C. Daniel, H.D. Scott, J.F. Murdoch, M.J. Habiger, and H.M. Burks. 1996. Stream quality impacts of best management practices in a northwestern Arkansas basin. Water Resour. Bull. 32:499–509.

Ehlke, T.A., G.G. Ehrlich, P.E. Greeson, R.T. Kirkland, and G.E. Mallard. 1987. Description of methods. Part 1. Bacteria. p. 3–98. *In* L.J. Britton and P.E. Greeson (ed.) Methods for collection and analysis of aquatic biological and microbiological samples. U.S. Geol. Surv. Techniques of Water-Resour. Invest.

Ferguson, R.I. 1986. River loads underestimated by rating curves. Water Resour. Res. 22:74–76.

Fishman, M.J., and L.C. Friedman. 1989. Methods for determination of inorganic substances in water and fluvial sediments. p. 311–389. U.S. Geol. Surv. Techniques of Water-Resources Invest.

Guy, H.P. 1969. Laboratory theory and methods for sediment analysis. p. 11–17. U.S. Geol. Surv. Techniques of Water-Resour. Invest.

Haycock, N.E., and G. Pinay. 1993. Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. J. Environ. Qual. 22:273–278.

Helsel, D.R., and R.M. Hirsch. 1992. Statistical methods in water resources. Elsevier Science Publ. Co., Amsterdam.

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

Jordan, T.E., D.L. Correll, and D.E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. J. Environ. Qual. 22:467–473.

Kauffman, J.B., W.C. Krueger, and M. Vavra. 1983. Impacts of cattle on streambanks in northeastern Oregon. J. Range Manage. 36: 683–685.

Koch, R.W., and G.M. Smillie. 1986. Bias in hydrologic prediction using log-transformed regression models. Water Resour. Bull. 22:717–723.

Koerkle, E.H., D.K. Fishel, M.J. Brown, and K.M. Kostelnik. 1996. Evaluation of agricultural best-management practices in the Conestoga River headwaters, Pennsylvania: Effects of nutrient management on water quality in the Little Conestoga Creek headwaters, 1983–89. U.S. Geol. Surv. Water-Resour. Invest. Rep. 95-4046.

Koerkle, E.H., and L.C. Gustafson-Minnich. 1997. Surface-water quality changes after 5 years of nutrient management in the Little Conestoga Creek headwaters, Pennsylvania, 1989–91. U.S. Geol. Surv. Water-Resour. Invest. Rep. 97-4048.

Lietman, P.L., L.C. Gustafson-Minnich, and D.W. Hall. 1997. Evaluation of agricultural best-management practices in the Conestoga River headwaters, Pennsylvania: Effects of pipe-outlet terracing on quantity and quality of surface runoff and ground water in a small carbonate-rock basin near Churchtown, Pennsylvania, 1983–89. U.S. Geol. Surv. Water-Resour. Invest. Rep. 94-4206.

Lietman, P.L., J.R. Ward, and T.E. Behrendt. 1983. Effects of specific land uses on nonpoint sources of suspended sediment, nutrients, and herbicides—Pequea Creek Basin, Pennsylvania, 1979–80. U.S. Geol. Surv. Water-Resour. Invest. Rep. 83-4113.

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

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.

Lystrom, D.J., F.A. Rinella, D.A. Rickert, and L. Zimmerman. 1978. Multiple regression modeling approach for regional water quality management. USEPA Rep. 600/7-78-198. USEPA, Washington, DC.

McLeod, R.V., and R.O. Hegg. 1984. Pasture runoff quality from application of inorganic and organic nitrogen sources. J. Environ. Qual. 13:122–126.

Mostaghimi, S., P.W. McClellan, U.S. Tim, J.C. Carr, R.K. Byler, T.A. Dillaha, V.O. Shanholtz, and J.R. Pratt. 1989. Watershed/water quality monitoring for evaluating animal waste BMP effectiveness—Owl Run watershed. Pre-BMP final report. Rep. O-P1-8906. Virginia Dep. of Conservation and Historic Resour., Div. of Soil and Water Conservation, Richmond, VA.

National Oceanic and Atmospheric Administration. 1994. Climatological data—annual summary—Pennsylvania 1994. National Climate Data Center, Asheville, NC.

Nelson, W.M., A.J. Gold, and P.M. Groffman. 1995. Spatial and temporal variation in groundwater nitrate removal in a riparian forest. J. Environ. Qual. 24:691–699.

Odum, E.P. 1983. Basic ecology. Saunders College Publ., Philadelphia, PA.

Orr, H.K. 1960. Soil porosity and bulk density on grazed and protected Kentucky bluegrass range in the Black Hills. J. Range Manage. 13:80–86.

Osborne, L.L., and D.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biol. 29:243–258.

Osmond, D.L., D.E. Line, and J. Spooner. 1995. Section 319 National Monitoring Program—an overview. NCSU Water Quality Group, Biological and Agricultural Engineering Dep., North Carolina State Univ., Raleigh, NC.

Parsons, J.E., R.B. Daniels, J.W. Gilliam, and T.A. Dillaha. 1994. Reduction in sediment and chemical load agricultural field runoff by vegetative filter strips. Rep. 286. Water Resour. Res. Inst. of the Univ. of North Carolina, Raleigh, NC.

Pearce, R.A., M.J. Trlica, W.C. Leininger, J.L. Smith, and G.W. Frasier. 1997. Efficiency of grass buffer strips and vegetation height on sediment filtration in laboratory rainfall simulations. J. Environ. Qual. 26:139–144.

Pennsylvania Department of Environmental Resources. 1986. Field application of manure. Commonwealth of Pennsylvania, Harrisburg, PA.

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.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers. USEPA Rep. 444/4-89-001. USEPA, Washington, DC.

Poth, C.W. 1977. Summary ground-water resources of Lancaster County, Pennsylvania. Pennsylvania Geol. Surv. Water Resour. Rep. 43.

Rauzi, F., and C.L. Hanson. 1966. Water intake and runoff as affected by intensity of grazing. J. Range Manage. 19:351–356.

Rogers, R.D., and S.A. Schumm. 1991. The effect of sparse vegetative cover on erosion and sediment yield. J. Hydrol. 123:19–24.

Salisbury, F.B., and C.W. Ross. 1985. Plant physiology. 3rd ed. Wadsworth Publ. Co., Belmont, CA.

SAS Institute. 1982. SAS user's guide: Statistics. SAS Inst., Cary, NC.

Unangst, D. 1992. The Conestoga Headwaters Project 10-Year Report of the Rural Clean Water Program. Pennsylvania State Rural Clean Water Program Coordinating Committee. USDA, Agricultural Stabilization and Conservation Service.

Walker, J.F., D.J. Grackles, S.R. Cores, D.W. Owens, and J.A. Whirl. 1995. Evaluation of nonpoint-source contamination, Wisconsin: Land-use and best-management-practices inventory, selected stormwater-quality data, urban-watershed quality assurance and quality control, constituent loads in rural streams, and snowmelt-runoff analysis, water year 1994. U.S. Geol. Surv. Open-File Rep. 95-320.

Young, R.A., T. Huntrods, and W. Anderson. 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. J. Environ. Qual. 9:483–487.