
Responses to Riparian Restoration in the Spring Creek Watershed, Central Pennsylvania

Robert F. Carline^{1,2} and Mary C. Walsh^{3,4}

Abstract

Riparian treatments, consisting of 3- to 4-m buffer strips, stream bank stabilization, and rock-lined stream crossings, were installed in two streams with livestock grazing to reduce sediment loading and stream bank erosion. Cedar Run and Slab Cabin Run, the treatment streams, and Spring Creek, an adjacent reference stream without riparian grazing, were monitored prior to (1991–1992) and 3–5 years after (2001–2003) riparian buffer installation to assess channel morphology, stream substrate composition, suspended sediments, and macroinvertebrate communities. Few changes were found in channel widths and depths, but channel-structuring flow events were rare in the drought period after restoration. Stream bank vegetation increased from 50% or less to 100% in nearly all formerly grazed riparian buffers. The proportion of fine sediments in stream substrates decreased in Cedar Run

but not in Slab Cabin Run. After riparian treatments, suspended sediments during base flow and storm flow decreased 47–87% in both streams. Macroinvertebrate diversity did not improve after restoration in either treated stream. Relative to Spring Creek, macroinvertebrate densities increased in both treated streams by the end of the posttreatment sampling period. Despite drought conditions that may have altered physical and biological effects of riparian treatments, goals of the riparian restoration to minimize erosion and sedimentation were met. A relatively narrow grass buffer along 2.4 km of each stream was effective in improving water quality, stream substrates, and some biological metrics.

Key words: grazing, macroinvertebrate communities, restoration, riparian, water quality.

Introduction

This study was motivated, in part, by findings of Beard and Carline (1991), who showed that poor reproductive success of brown trout (*Salmo trutta*) in a 16-km section of Spring Creek, central Pennsylvania, was related to a high proportion of fine sediment in substrates used for spawning. Fine sediment reduced the amount of suitable spawning habitat, and where brown trout spawned, embryo survival was low. Surveys of headwater tributaries to Spring Creek revealed that two subbasins with substantial proportions of unfenced riparian pastures were the likely sources of fine sediment.

In an effort to reduce sediment loading to Spring Creek, several local private organizations and public agencies collaborated on a riparian restoration project. Their goal was to substantially reduce livestock access to tributary streams and to stabilize eroding stream banks. Initial contacts with landowners revealed that they might be willing to participate in the project if buffer strips were kept

narrow, that is, 3–4 m wide, and woody vegetation was not planted. Narrow buffer strips were desired because most riparian pastures were small (<5 ha), and landowners were reluctant to give up forage production. Landowners were not in favor of planting trees in the buffer strip because of eventual accumulation of woody debris in pastures and shading that would further reduce forage production. Dutcher (1999) surveyed riparian landowners in three nearby watersheds and found similar attitudes about management of riparian vegetation.

Organizers of the project were concerned about the effectiveness of narrow, grass buffers in reducing sediment transport to streams. At that time, resource management agencies in the Chesapeake Bay watershed were actively promoting establishment of relatively wide forested riparian buffers to reduce sediment and nutrient loading to streams largely because of the demonstrated effectiveness of such buffers (Lowrance et al. 1984, 1986). Experimental plots with grass filter strips showed that more than 50% of sediment eroding from row crops or bare soil could be retained (Dillaha et al. 1989; Magette et al. 1989). More recently, Line (2003) measured effectiveness of grass buffers at the pasture scale, and Galeone (2000) studied responses to grass buffers in a 3.7-km² catchment, but there have been no tests of the effectiveness of narrow grass buffers at larger scales. These questions about the utility of narrow, grass riparian buffers prompted initiation of this study. Specifically, our objectives were to

¹ U.S. Geological Survey, Pennsylvania Cooperative Fish and Wildlife Research Unit, 402 Forest Resources Building, University Park, PA 16802, U.S.A.

² Address correspondence to R. F. Carline, email rcarline@psu.edu

³ School of Forest Resources, Pennsylvania State University, University Park, PA 16802, U.S.A.

⁴ Present address: Western Pennsylvania Conservancy, 208 Airport Drive, Middletown, PA 17057, U.S.A.

quantify the effects of stream bank fencing with narrow grass buffer strips on channel morphology, stream substrate composition, and macroinvertebrate communities at the reach scale and sediment loads at the catchment scale in two second-order streams that had been subjected to unrestricted grazing by cattle and horses. A preconstruction assessment of the treatment subbasins and a reference subbasin was completed during 1991 and 1992 (Wohl & Carline 1996). Riparian restoration projects were done from fall 1992 to summer 1998. The postconstruction study, which extended from January 2001 to January 2003, was delayed for more than two years after the last restoration project to allow riparian vegetation to become well established.

Study Area

The study was conducted in the headwaters of the Spring Creek watershed (378 km²), Centre County, central Pennsylvania, in a limestone valley within the Valley and Ridge physiographic province. Spring Creek flows into Bald Eagle Creek, a tributary to the West Branch of the Susquehanna River. The three study basins, Spring Creek, Cedar Run, and Slab Cabin Run, are adjacent (Fig. 1), sharing similar geology and soils (Braker 1981). The headwaters of the upper Spring Creek and Slab Cabin Run basins originate on a sandstone ridge that yields water low in pH (<7.0) and alkalinity. When these low-pH streams reach the valley floor, groundwater inputs from limestone aquifers alters their chemistry. Headwaters of Cedar Run originate in the valley floor. All three study streams have relatively high pH (7.7–8.4), high total alkalinity (196–208

mg/L as CaCO₃), and high concentrations of calcium (47–68 mg/L).

The reference station on Spring Creek was located 31 km upstream from the mouth of Spring Creek. More than one-half of this subbasin was forested and 22% of the land was in agriculture, but there were no unfenced riparian pastures (Table 1). Cedar Run and its main tributary, Mackey Run, flowed through 2.5 km of unfenced pasture prior to riparian restoration; agricultural lands comprise 69% of these subbasins. The surface area of the riparian pastures was 44 ha, and livestock density ranged from 3.9 to 4.0 animals/ha between 1991 and 2003. The upper basin of Slab Cabin Run was 40% agricultural lands. About 4.1 km of the stream flowed through unfenced riparian pastures prior to restoration, and stream bank erosion was evident in all pastures. Surface area of the riparian pastures was 76 ha, and livestock density ranged from 5.3 to 5.6 animals/ha.

Methods

Restoration Techniques

Stream bank fencing, construction of animal crossings and animal accesses to one side of the stream, and bank stabilization were the primary restoration techniques. Electrified fences, which were usually high-tensile wire, were installed about 3 m from the stream bank. Where livestock had to cross streams, rock-lined ramps were installed on both sides of the stream and wires were stretched across the stream on either side of the crossing. Animal accesses consisted of rock-lined ramps on one side of the stream.

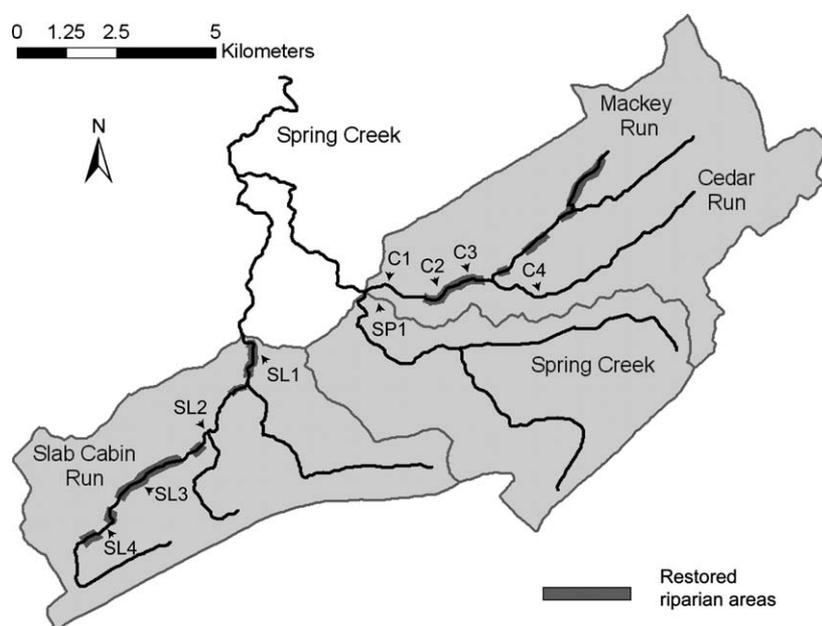


Figure 1. Locations of sampling sites and restored riparian areas in headwater subbasins of the Spring Creek watershed, Pennsylvania. Gauging stations were located at the intersections of the streams and the subbasin boundaries.

Table 1. Physical characteristics and land use of study stream basins within the Spring Creek watershed, Pennsylvania, and lengths of permanent streams flowing through unfenced pastures prior to restoration in 1991 and lengths of stream with added grass buffers are shown for 2003.

	Spring Creek	Cedar Run	Slab Cabin Run
Basin area (km ²)	34	46	44
Land use (% area)			
Agriculture	22	69	40
Forest	56	19	48
Urban	22	12	12
Stream length (km)			
Unfenced pastures (1991)	0	2.5	4.1
Grass buffers (2003)	—	2.4	2.5

Vertical stream banks that were likely to continue eroding in the absence of disturbance by livestock were lined with 15-cm limestone rocks. No plantings were made in the buffer strip; all vegetation colonized naturally.

Construction began in the Slab Cabin Run basin in late 1992 and in the Cedar Run basin in 1994. The last construction project was completed in 1998. Restoration activities were completed on seven properties in the Cedar Run basin; 2,000 m of stream were fenced, 14 animal accesses were installed, and 245 m of bank were stabilized with rock. Improvements were made on 10 properties in the Slab Cabin Run basin; 2,740 m of stream were fenced, 26 animal crossings and accesses were installed, and 1,875 m of bank were stabilized. Construction was done at no cost to the landowners, who were required to sign agreements ensuring that they maintained all improvements for 10 years. All properties were inspected in 2003; two landowners in the Slab Cabin Run basin had not maintained stream bank fences. Thus, along Cedar Run and Slab Cabin Run, 98 and 61%, respectively, of previously unfenced stream pastures had intact streamside fences during the posttreatment assessment (Table 1).

Pre- and Posttreatment Assessments

In each of the treatment watersheds, we established four sampling sections—two that were grazed and two that were ungrazed but downstream from a grazed stream reach (Table 2). Stream bank vegetation, channel morphology, substrate, and macroinvertebrate communities were evaluated at the four sampling stations for potential localized changes in grazed stream reaches and in reaches immediately downstream after treatment (Table 2). Gauging stations located at the outflow of the study catchments monitored sediment and stream flow discharged from the upstream watersheds during the pre- and posttreatment studies.

We established only one sampling station immediately upstream from the gauging site in Spring Creek, the reference stream. We were not able to establish other sampling

Table 2. Sampling sites (shown in Fig. 1) and gauging station locations in study streams and types of data collected by location in the pre- and posttreatment assessments.

Sampling Sites and Gauging Stations	Distance from Gauging Stations (km)	Riparian Grazing	Type of Data Collected
Spring Creek			
SP1 and gauging station	0	Ungrazed	SD, W, C, S, M
Cedar Run			
C1 and gauging station	0	Ungrazed	SD, W, C, S, M
C2	2.2	Grazed	C, S, M
C3	4.4	Grazed	C, S, M
C4	5.1	Ungrazed	C, S, M
Slab Cabin Run			
Gauging station	0		SD, W
SL1	0.8	Grazed	C, S
SL2	3.2	Ungrazed	C, S, M
SL3	5.7	Grazed	C, S, M
SL4	7.9	Ungrazed	C, S, M

Types of data are represented by the following codes: SD, stage and discharge; W, water samples of suspended sediment; C, channel morphology; S, substrate composition; M, macroinvertebrate communities.

stations upstream because Spring Creek flows underground during dry periods of the year and reemerges about 400 m upstream of the gauging site, which was insufficient distance to create another sampling section.

Sampling sections ranged from 150 to 250 m in length and typically had three riffle–pool sequences. We measured channel morphology at each of the nine sample sections in July 1992 and 2001. Width was measured at eight or more transects per section, and depth and velocity measurements were taken at 10–15 points along each transect. Velocity was measured at 0.6 of total water depth from the water surface using a portable flow meter (Marsh-McBirney Flow-mate 2000). At both ends of the cross-section, the percentage of vegetated stream banks was estimated as 0, 25, 50, 75, or 100%; vegetation was characterized as herbs, shrubs, or trees.

Substrate similar to brown trout spawning habitat was sampled in May 1992 and 2001 at each of the nine sampling sections. Beard (1990) described Brown trout redds in Spring Creek as having the following characteristics: velocity ranged from 0.25 to 0.57 m/second, depth ranged from 0.2 to 0.5 m, and substrate diameters ranged from 2 to 50 mm. A stovepipe sampler (McNeil & Ahnell 1964) with 10 cm diameter was used to collect four samples at each site. Samples were dried at 105°C and sifted through a series of 12 sieves with pore sizes ranging from 0.25 to 12.7 mm; the portion retained by each sieve was weighed. We used the percentage by weight of fines (<1 mm) as an index of substrate composition.

To monitor hydrologic and water quality characteristics, gauging stations were established during the prerestoration phase of the study. Water levels were recorded during the prerestoration period from September 1991 through

August 1992 (Wohl & Carline 1996) and during the post-restoration phase from 1 January 2001 to 15 January 2003.

Water level recorders (Design Analysis DH-21 Submersible Waterlogger), installed in stilling wells, measured stream stage at 0.5-hour intervals. Stream discharge was measured 10–15 times over a wide range of stage values at each gauging station to develop rating curves. Ice in the channel and in the stilling well in Slab Cabin Run from January 2001 to March 2001 precluded monitoring of water levels; water levels were estimated from a gauging station located farther downstream on Slab Cabin Run. We used a regression equation to estimate discharge at the upstream gauging station on Slab Cabin Run from discharge at the downstream gauging station ($R^2 > 0.95$).

Water Quality

Suspended sediment was characterized from September 1991 to August 1992 (Wohl & Carline 1996) and from January 2001 to January 2003 at each gauging station. Water samples were collected weekly during base flow. We sampled 12 storms during the preconstruction assessment and 43 storms during the postconstruction phases. During storms, which were defined as periods when discharge increased by 20% or more from base flow, water was collected hourly with automatic samplers (Hach American Sigma 900). Six samples per storm, including two each from the rising limb, peak, and falling limb of the hydrograph, were analyzed for total suspended sediment (TSS). Owing to below-normal precipitation during the postconstruction period, base flow in the upper portion of Slab Cabin Run was much reduced. No water samples were collected in Slab Cabin Run from 25 July 2001 to 2 April 2002 and from 26 August 2002 to 21 November 2002, when stream flow was intermittent upstream of the gauging station.

Turbidity was measured on all samples with an Orbeco-Hellige Digital Turbiditymeter (model 965). A subset of samples was analyzed for TSS using method 2540-D (APHA, AWWA, & Water Environment Federation 1995). After more than 20 samples were analyzed for TSS and turbidity from each stream, relationships between turbidity and TSS were developed using regression analyses. Thereafter, regressions were used to estimate TSS for samples when only turbidity was measured. Turbidity was well correlated with TSS in all streams ($R^2 > 0.9$).

Macroinvertebrate Communities

Triplicate benthic samples were taken from riffles using a Surber sampler at each of the nine sites in May and August 1992, August 2000, May and August 2001, and May 2002. Samples were fixed in 10% formalin and transferred to 90% ethanol. Insect taxa were identified to genus, except the dipteran family Chironomidae; all other invertebrates were identified to class or the lowest taxonomic level possible. The Shannon diversity index (Pielou

1975), density, taxa richness, and Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness were determined for all samples. The Shannon diversity index (H') is calculated as follows:

$$H' = - \sum p_i \log_e p_i,$$

where p_i = the proportion of the community belonging to the i th taxa.

The Shannon diversity index is a widely used measure of community structure, combining the measures of taxa richness and evenness. Its maximum score has been debated among ecologists, but 5.0 is generally accepted as the highest score (Washington 1984).

Data Analyses

We tested null hypotheses that postrestoration data were equal to pre-restoration data. Because of nonnormal distribution of data, nonparametric tests were used for most variables. To determine if channel morphology characteristics were different after restoration, median percentage of vegetated stream bank, width, depth, and percentage fines were compared within sites using a Mann–Whitney test. Similarly, we compared median discharge during days when water samples were collected and TSS during base flow and storm flow.

Any responses to restoration in macroinvertebrate community composition were qualitatively assessed by comparing relative abundances of taxa from prerestoration to postrestoration sampling periods. Further analysis examined changes in community metrics. To test whether macroinvertebrate community metrics had changed within sampling sites and within streams from prerestoration to postrestoration periods relative to reference location on Spring Creek, differences in mean sample metrics between restored sample sites and Spring Creek were analyzed for each sampling season. Mean differences in community metrics were considered a measure of changes in macroinvertebrate communities in restored streams that accounts for natural variation in the reference stream. Nested general linear models were applied to differences in sample macroinvertebrate community metrics (diversity, total taxa richness, and EPT taxa richness) for each seasonal sampling period. Dunnnett multiple comparisons were used to estimate significant differences detected by general linear models.

General linear models for each season were constructed as follows:

$$Y = \text{Year} + \text{Stream} + \text{Site}(\text{Stream}) + \text{Error},$$

where Y = mean restored site community metric – mean Spring Creek community metric.

If restoration benefited macroinvertebrate communities, we would expect this derived statistic to change from a large negative value to a small negative or a positive value and that there would be a year effect. A significant stream effect would indicate that benthic community responses

were different between the two treated streams, and a significant site effect would indicate that ungrazed and restored sites responded differently to riparian treatments.

Results

Stream Bank and Channel Morphology

Stream banks in grazed sections of Cedar Run were 50% vegetated, whereas those in grazed sections of Slab Cabin Run were completely unvegetated owing to intense live-

stock use (Table 3). After livestock were excluded from the riparian zones, stream banks in the restored sections were largely vegetated, and there was no evidence of bank erosion. Grasses constituted most of the colonizing vegetation along with some herbaceous dicots. Woody shrubs accounted for the rest of the vegetation (14% of total), but young trees were rare in the buffer strips.

Stream channel morphology changed little after riparian restoration. There were no significant changes in channel width except for a restored site on Slab Cabin Run (Table 3). Median depth did not change significantly in

Table 3. Median values and interquartile ranges for percentage of vegetated stream bank, channel width and depth, and percentage of fines (<1 mm) in substrates in Spring Creek, Cedar Run, and Slab Cabin Run.

	Vegetated Stream Bank (%)		Width (m)		Depth (m)		Fines (%)	
	1992	2001	1992	2001	1992	2001	1992	2001
Spring Creek—ungrazed								
SP1								
Median	100	100	4.6	5.2	0.20	0.22	5.6	8.2
Quartile	100, 100	100, 100	4.2, 5.8	4.1, 7.0	0.15, 0.28	0.15, 0.25	4.3, 6.9	5.4, 14.0
<i>n</i>	20	20	10	10	10	10	4	4
Cedar Run—ungrazed								
C1								
Median	100	75*	8.0	8.2	0.14	0.19	18.2	11.7*
Quartile	100, 100	50, 100	6.6, 9.2	6.7, 8.5	0.11, 0.20	0.15, 0.24	14.0, 23.5	10.0, 13.7
<i>n</i>	18	18	9	9	9	9	4	4
C4								
Median	100	100	5.8	6.1	0.23	0.17*	20.3	15.6*
Quartile	100, 100	100, 100	5.0, 7.6	4.9, 6.2	0.20, 0.24	0.14, 0.20	18.9, 25.9	10.0, 16.9
<i>n</i>	20	20	9	10	9	9	4	4
Grazed/restored								
C2								
Median	50	100*	6.3	6.6	0.33	0.21*	28.9	6.9*
Quartile	50, 72	100, 100	6.0, 7.0	6.3, 7.6	0.19, 0.40	0.13, 0.27	22.9, 37.0	5.5, 17.7
<i>n</i>	20	20	10	10	10	10	4	4
C3								
Median	50	100*	6.4	5.6	0.18	0.17	33.5	23.3*
Quartile	25, 50	100, 100	5.6, 6.6	5.3, 6.2	0.16, 0.24	0.14, 0.18	28.8, 37.8	18.6, 26.4
<i>n</i>	20	20	10	10	10	20	4	4
Slab Cabin Run—ungrazed								
SL2								
Median	100	100	4.4	4.3	0.20	0.21	34.6	35.4
Quartile	85, 100	75, 100	3.3, 5.5	3.9, 4.7	0.18, 0.24	0.14, 0.22	31.4, 37.5	26.8, 48.4
<i>n</i>	24	20	12	10	12	10	4	4
SL4								
Median	100	100	2.4	3.0	0.10	0.10	18.6	16
Quartile	100, 100	100, 100	2.0, 2.6	2.3, 3.3	0.09, 0.13	0.09, 0.12	13.0, 26.4	8.5, 28.3
<i>n</i>	16	16	8	8	8	8	4	4
Grazed/restored								
SL1								
Median	0	100*	7.5	5.8	0.11	0.13	18.6	a
Quartile	0, 0	100, 100	5.8, 8.7	3.4, 6.8	0.08, 0.15	0.05, 0.15	13.0, 26.4	
<i>n</i>	54	18	27	20	27	20	4	
SL3								
Median	0	88*	1.7	3.4*	0.18	0.12*	35.7	51.9*
Quartile	0, 0	50, 100	1.5, 2.5	2.2, 4.3	0.14, 0.23	0.08, 0.17	24.8, 38.5	43.0, 65.0
<i>n</i>	40	20	20	10.0	20	10.00	4	4

At sites C2, C3, SL1, and SL3, values for 1992 represent grazed conditions and 2001 represent restored conditions. Asterisks indicate that values in 2001 were significantly different ($p < 0.05$) than those in 1992 (Mann-Whitney test). The lowercase letter "a" denotes that the channel was dry during survey.

most sections; at three sites, depth decreased, which was mostly the result of lower discharge during the 2001 survey compared to that in 1992. The amount of fines in stream substrates decreased significantly at all stations in Cedar Run, which suggests a reduction in sediment loading. But in Slab Cabin Run, the trend of decreasing fines was not evident. Low discharge may have contributed to this lack of response.

Stream Flow and TSS

Discharge in Spring Creek was below average during the pre- and most of the posttreatment assessment periods (Fig. 2). The posttreatment period spanned the latter part of a four-year drought, which ended during the last few months of posttreatment measurements. The effects of the drought were most evident in Slab Cabin Run, which has a lower water yield than the other subbasins. In the lower portion of the subbasin, Slab Cabin Run is perched above

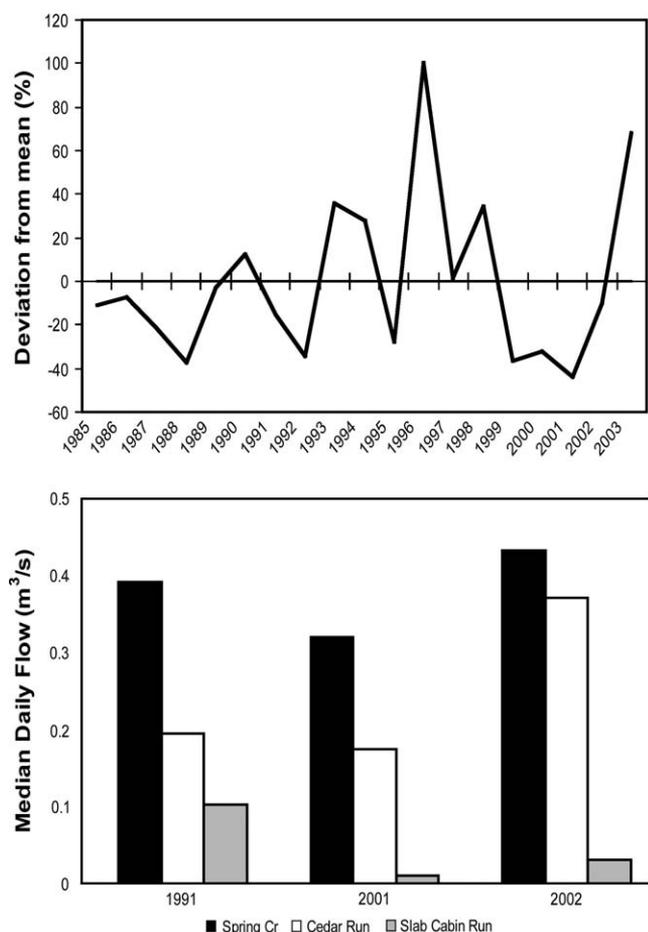


Figure 2. (a) Deviation from the mean annual flow at the Houseville gauge on Spring Creek, 6.5 km downstream of the gauge monitoring the reference subbasin of Spring Creek. (b) Median daily discharge at the upper Spring Creek, Cedar Run, and Slab Cabin Run gauges. All discharge values represent the 12-month periods from 1 September to 31 August 1985–2003.

the groundwater table, resulting in prolonged periods of dry channel immediately upstream of the gauging site, when groundwater levels are below normal (O'Driscoll 2004).

Owing to below-normal discharge, we collected fewer water samples from Slab Cabin Run during the posttreatment period than during the pretreatment assessment period (Table 4). Despite the relatively dry posttreatment period, on days when we collected water samples, there were no significant differences ($p > 0.05$, Mann–Whitney test) in base flow or storm flow discharge in the treatment streams before and after riparian restoration. Therefore, a comparison of TSS before and after restoration should not be unduly influenced by discharge at the time of sample collection.

At base flow, Spring Creek had the lowest TSS levels during both pre- and postrestoration periods (Table 4). During the posttreatment period, median values of TSS ranged from 1.8 to 2.7 mg/L, which were significantly different ($p < 0.05$) than pretreatment levels. Cedar Run had intermediate levels of TSS during the prerestoration period; during the postrestoration years, TSS declined significantly from 47 to 57% relative to prerestoration levels. Prior to restoration, median TSS at base flow in Slab Cabin Run was quite high, 43.0 mg/L, but after restoration median values had declined by 85–87%.

During storms, Spring Creek had the lowest median TSS among the three streams during the pretreatment period (Table 4). During the posttreatment period, TSS in Spring Creek increased above pretreatment levels even though discharge was lower during the posttreatment period relative to the pretreatment period. This increase in sediment load corresponded with several construction projects in the basin. Cedar Run and Slab Cabin Run had high TSS levels before restoration, and these levels decreased by 75–79% in Cedar Run and by 83% in Slab Cabin Run after restoration. These decreases in TSS were not related to discharge during storms because there were no significant differences ($p > 0.05$) in discharge between pre- and postrestoration in either stream. These large reductions in TSS during storms provide strong evidence that riparian restoration had beneficial effects at the catchment scale.

Macroinvertebrate Communities

Two to three taxa dominated the study streams, making up more than one-half of the individuals collected (Table 5). In Spring Creek, Amphipoda and Diptera comprised a large proportion of taxa in pre- and postrestoration periods. In Cedar Run, the most common taxa were Amphipoda, Isopoda, and Diptera during all sampling periods. The macroinvertebrate community in Slab Cabin Run mostly consisted of Isopoda, Coleoptera, and Diptera. Amphipods were present in Slab Cabin Run but occurred in fewer numbers than the other streams. Although Diptera comprised a large proportion of the macroinvertebrates at all sampling periods, the relative

Table 4. Median and interquartile range of daily discharge when water samples were collected and concentrations of TSS during base flow and storm flow in study streams before (1991–1992) and after riparian restoration.

Stream, Year	Daily Discharge (m ³ /second)						TSS (mg/L)					
	Base Flow			Storm Flow			Base Flow			Storm Flow		
	Median	Quartile	n	Median	Quartile	n	Median	Quartile	n	Median	Quartile	n
Spring Creek												
1991–1992	0.39	0.29, 0.54	77	0.98	0.44, 1.33	9	3.3	2.4, 5.6	77	14.3	3.3, 16.7	9
2001	0.32*	0.22, 0.38	49	0.39*	0.26, 0.80	42	1.8*	1.0, 2.9	49	20.4	8.0, 42.3	20
2002	0.43	0.24, 0.61	51	0.36*	0.28, 0.54	37	2.7*	1.7, 3.7	51	26.0*	17.4, 68.8	19
Cedar Run												
1991–1992	0.21	0.16, 0.37	79	0.25	0.08, 0.57	7	21.9	17.3, 31.7	79	97.0	71.2, 158.2	7
2001	0.17*	0.14, 0.25	49	0.24	0.16, 0.46	39	9.5*	6.7, 14.1	49	20.6*	15.1, 34.3	19
2002	0.33*	0.24, 0.54	50	0.44	0.27, 0.57	36	11.7*	9.4, 15.5	50	24.7	20.5, 161.0	20
Slab Cabin Run												
1991–1992	0.07	0.05, 0.09	76	0.11	0.08, 0.57	12	43.0	21.8, 68.3	76	106.0	43.8, 143.7	12
2001	0.05	0.01, 0.16	28	0.23	0.09, 0.44	24	6.6*	5.4, 9.0	28	18.2*	10.4, 46.6	12
2002	0.14*	0.08, 0.31	27	0.14	0.01, 0.50	30	5.4*	4.0, 6.6	27	17.6*	11.8, 99.5	17

Asterisks indicate significant differences (Mann–Whitney test, $p < 0.05$) between prerestoration and each of the postrestoration years within streams.

abundance of Dipterans decreased during both seasons after restoration. Declines in the relative abundance of Oligochaeta also occurred since restoration. Similar patterns in relative abundance were observed in August samples.

Prior to restoration, macroinvertebrate densities in Slab Cabin Run and Cedar Run were depressed relative to Spring Creek, but densities exceeded those in Spring Creek by the end of the posttreatment sampling period (Fig. 3). For each sample date, we computed the difference in macroinvertebrate density between a treatment stream and Spring Creek and used this statistic to model the effects of year, stream, and sites within streams on densities. These difference statistics ranged from negative values in 1992 prior to restoration to positive values by August 2001 and May 2002 (Fig. 3). We analyzed data for

May and August collections separately to avoid possible seasonal influences.

For August samples, general linear models of the log-transformed difference in macroinvertebrate density (density metric) between restored streams and Spring Creek demonstrated that treatment year ($p < 0.001$) and stream were significant factors ($p = 0.014$), but there was no significant effect of site ($p = 0.063$) nested within streams. Individually, treatment sites did not respond differently to restoration. However, a significant stream effect suggests that Cedar Run and Slab Cabin Run differed, on the whole, in their responses to restoration. Density metrics for Cedar Run were greater than those for Slab Cabin Run (Dunnnett comparisons, $p = 0.0139$). Dunnnett multiple comparisons revealed that postrestoration

Table 5. Mean macroinvertebrate relative abundance (percent of total collections) and total number of class and order taxa for Spring Creek, Cedar Run, and Slab Cabin Run in May during prerestoration (1992) and postrestoration (2001–2002) periods.

	Mean Relative Abundance (% of Total)								
	Spring Creek			Cedar Run			Slab Cabin Run		
	1992	2001	2002	1992	2001	2002	1992	2001	2002
Amphipoda	61	9	23	17	20	14	2	5	5
Isopoda	0	<1	<1	30	32	45	1	20	27
Coleoptera	3	2	10	2	9	9	<1	19	1
Diptera	21	28	31	21	19	10	70	33	40
Ephemeroptera	3	2	3	1	2	1	1	3	<1
Trichoptera	7	2	17	14	6	17	2	2	4
Plecoptera	0	<1	0	0	<1	<1	<1	0	0
Oligochaeta	2	45	2	6	8	<1	21	2	14
Turbellaria	3	6	5	1	1	2	1	1	3
Other	0	6	10	8	3	3	1	15	6
Number of taxa	7	10	9	9	10	10	10	9	9

Other taxa include Gastropoda, Decapoda, Hirudinea, and Hydracarina. In Spring Creek, Cedar Run, and Slab Cabin Run, 3, 12, and 9 samples, respectively, were collected during each sampling period.

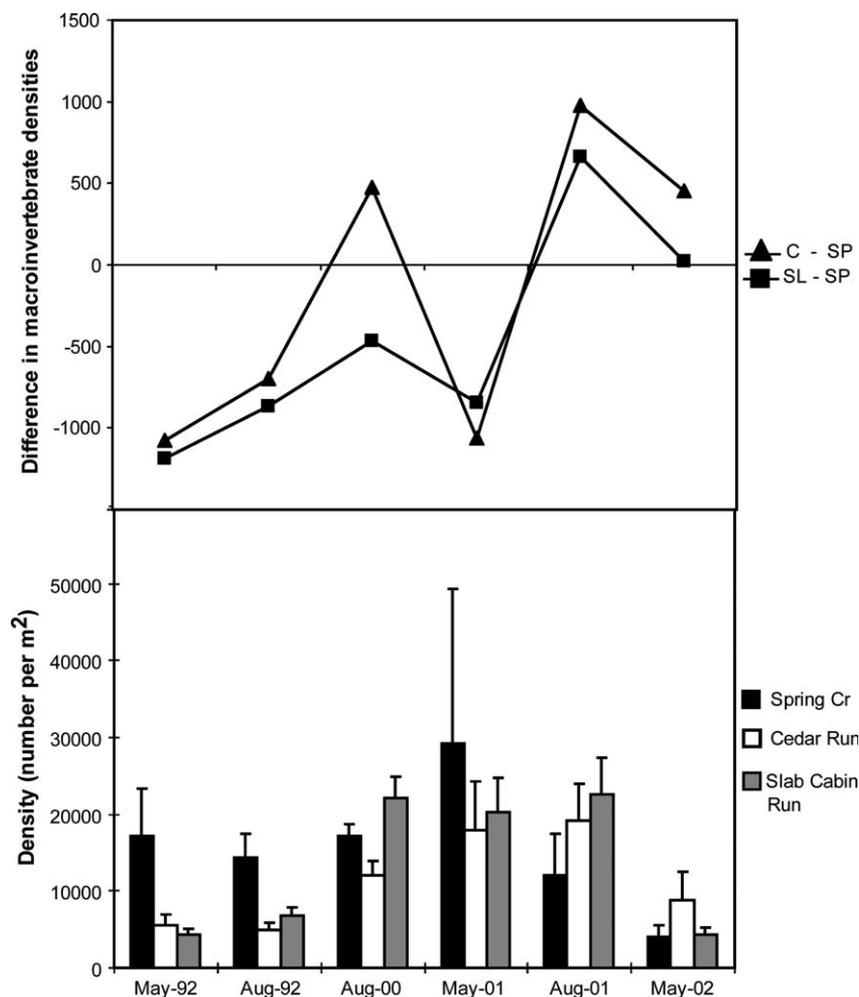


Figure 3. (a) Mean differences between macroinvertebrate densities in Cedar Run (C) and Spring Creek (SP) and between those in Slab Cabin Run (SL) and Spring Creek during prerestoration (1992) and postrestoration periods (2000–2002). (b) Mean macroinvertebrate density (number per square meter) and SE for Spring Creek, Cedar Run, and Slab Cabin Run during prerestoration (1992) and postrestoration periods (2000–2002). Three samples from Spring Creek, 12 from Cedar Run, and 9 from Slab Cabin Run were collected during each sampling event.

densities in 2000 ($p = 0.005$) and 2001 ($p < 0.001$) were significantly different from the pre-restoration densities in treatment streams. Thus, relative to changes in densities in Spring Creek, macroinvertebrate densities in August in Cedar Run and Slab Cabin Run were higher after restoration than in 1992.

For May samples, general linear models of the log-transformed difference between macroinvertebrate densities in restored streams and in Spring Creek demonstrated that treatment year was significant ($p = 0.003$), but there were no significant stream ($p = 0.760$) or site ($p = 0.700$) effects. Prerestoration macroinvertebrate densities in treatment streams were significantly different from 2002 macroinvertebrate densities ($p = 0.002$) but not from 2001 densities ($p = 0.410$) based on Dunnett multiple comparisons. Thus, relative to Spring Creek, macroinvertebrate densities in Cedar Run and Slab Cabin Run increased in May 2002 but not in May 2001.

Some macroinvertebrate community metrics in Cedar Run and Slab Cabin Run improved qualitatively after restoration, but responses were not consistent among streams or over time. August macroinvertebrate communities in Cedar Run had higher total taxa richness and EPT taxa richness (Table 6) in 2000 and 2001 relative to prerestoration values. During August in Slab Cabin Run, total taxa richness was higher in 2000 but metric values declined by 2001. Spring Creek had similar increases during the summer in total and EPT taxa richness in 2000, but elevated EPT richness diminished by 2001. In general linear models for summer samples of the difference between Cedar Run and Spring Creek total richness and the difference between Slab Cabin Run and Spring Creek total richness, year ($p = 0.277$), stream ($p = 0.906$), and site ($p = 0.580$) were not significant factors. Similarly, year ($p = 0.123$), site ($p = 0.218$), and stream ($p = 0.219$) were not significant factors in general linear models of EPT richness

Table 6. Mean macroinvertebrate taxa richness, EPT richness, Shannon diversity index (H'), SE, and number of samples for Spring Creek, Cedar Run, and Slab Cabin Run in May and August during prerestoration (1992) and postrestoration (2000–2002) periods.

Stream	Macroinvertebrate Metric	1992		2000 August	2001		2002 May
		May	August		May	August	
Spring Creek	Total taxa richness	16.33	14.33	17.67	22.33	15.00	15.33
	SE	0.67	0.88	1.76	2.91	0.00	2.19
	<i>n</i>	3	3	3	3	3	3
	EPT taxa richness	3.33	2.00	2.33	3.33	1.67	2.33
	SE	0.33	0.58	0.67	0.67	0.33	0.67
	<i>n</i>	3	3	3	3	3	3
	H'	1.29	1.37	1.36	1.76	1.92	1.95
	SE	0.10	0.07	0.11	0.09	0.11	0.15
	<i>n</i>	3	3	3	3	3	3
Cedar Run	Total taxa richness	13.08	11.92	15.83	18.83	13.92	14.58
	SE	1.01	0.47	1.28	1.28	0.83	1.04
	<i>n</i>	12	12	12	12	12	12
	EPT taxa richness	1.67	0.92	1.42	2.33	1.17	1.50
	SE	0.40	0.15	0.34	0.33	0.17	0.26
	<i>n</i>	12	12	12	12	12	12
	H'	1.77	1.60	1.69	1.98	1.54	1.81
	SE	0.08	0.06	0.11	0.04	0.13	0.11
	<i>n</i>	12	12	12	12	12	12
Slab Cabin Run	Total taxa richness	9.50	11.50	13.33	14.33	11.00	9.82
	SE	0.56	0.57	0.99	1.56	1.10	1.35
	<i>n</i>	9	9	9	9	9	9
	EPT taxa richness	1.67	1.17	0.83	1.00	0.58	0.82
	SE	0.36	0.21	0.17	0.25	0.15	0.26
	<i>n</i>	9	9	9	9	9	9
	H'	1.32	1.62	1.34	1.14	1.18	1.19
	SE	0.09	0.04	0.11	0.12	0.12	0.15
	<i>n</i>	9	9	9	9	9	9

differences and diversity differences for August. For community metrics, no year effect, indicating a response to restoration, was found in August samples. Sampling sites and both treatment streams had statistically similar community metrics.

In May 2001 samples, mean total taxa richness also increased in restored streams relative to pretreatment levels. By May 2002, in Cedar Run total taxa richness continued to be higher than prerestoration values but decreased in Slab Cabin Run to near pretreatment levels. Increases in EPT richness and diversity were not consistent among streams. Diversity values increased in spring samples of both postrestoration years in Cedar Run but not in Slab Cabin Run in May samples. EPT richness did not generally improve in treatment streams during May with the exception of elevated values in Cedar Run relative to pretreatment EPT richness during May 2001. Despite some qualitative changes in metrics, general linear models of differences between Cedar Run and Spring Creek and between Slab Cabin Run and Spring Creek for total richness, EPT richness, and diversity did not demonstrate that year, site, and stream were consistently significant factors for all metric variables. Because no year effect was found, restoration did not significantly improve any community metrics in treatment streams. Both Cedar

Run and Slab Cabin Run had statistically similar communities because stream or site did not have significant effects. Although some improvements in macroinvertebrate densities occurred, no other community metrics were statistically significant after restoration.

Discussion

The primary question addressed by this project was whether a narrow, grass riparian buffer would result in a significant reduction in sediment load at the stream reach and catchment scales, providing that a large proportion of the degraded riparian habitats were restored. Excluding livestock from the riparian zone allowed grasses to quickly colonize and stabilize stream banks. In a similar physiographic setting, Galeone (2000) showed extensive colonization of stream bank grasses one year after fence installation on southeast Pennsylvania dairy farms. Although grasses quickly colonized eroding stream banks in our study, channel morphology was unchanged.

We measured stream channel attributes 3–9 years after fence construction and found no significant changes in channel width or depth. A common response to livestock exclusion is narrowing and deepening of the stream channel (Knapp & Matthews 1996; Goodwin et al. 1997). Lyons

et al. (2000) noted that grasses are particularly effective in trapping sediment in the riparian zone, which leads to a buildup of the banks and a narrowing of the channel. As the channel narrows, water velocity should increase and degradation of the stream bottom should deepen the channel. Kondolf (1995) suggests that 10 or more years may be needed to assess channel responses to riparian restoration in part because numerous high flow events may be required to move stream substrates, presumably to different stable conditions. Below-average stream flow during the last few years of the study may have contributed to the lack of change in channel morphology.

We also used the percentage of fines in brown trout spawning habitat to assess channel response to riparian restoration because survival of salmonid embryos during incubation is directly related to substrate composition (Chapman 1988). The percentage of fines in treated pasture stream reaches in Cedar Run was reduced after restoration, but no significant response was noted in Slab Cabin Run. Two factors may have contributed to the lack of response in Slab Cabin Run: 1.6 km of stream still flow through unfenced riparian pastures and low stream flow after riparian restoration had not yet flushed out fine sediments. Because we suspected that stream flow may have influenced substrate composition, we resampled sediments in May 2005, when stream flow had been above average for three years. The median percentage of fines in Cedar Run did not change significantly, but among three sites in Slab Cabin Run, median percentage of fines declined from 36.2 to 14.4% from prerestoration values ($p = 0.014$, Mann-Whitney test). These results seem to support the notion that below-average stream flow in Slab Cabin Run during the postconstruction period contributed in part to the lack of response in substrate composition.

Our results are consistent with other studies that have shown that grass buffers can effectively trap sediment and contribute to reduced sediment loads in streams. Our study differs from other published works in that we measured responses at the small watershed scale (area of circa 45 km²). When grass buffers ranging in width from 4.0 to 9.2 m were tested on experimental field plots, sediment transport was reduced from about 50 to 84% (Dillaha et al. 1989; Magette et al. 1989; Parsons et al. 1994). Line (2003) found a 60% reduction in TSS after a 10- to 16-m-wide grass buffer had been established for 4–5 years in a 57-ha dairy cow pasture. Galeone (2000) examined effects of 2- to 4-m-wide grass buffers on dairy cow pastures in a 3.7-km² catchment and found a 17–54% reduction in TSS during the first year after fencing. Our buffer widths (averaged 3 m) were similar to those of Galeone (2000), and three or more years after fencing, TSS during base flow was reduced from 47 to 87% and during storm flow from 75 to 83% at the catchment outflow. These results provide strong evidence that relatively narrow grass buffers combined with fencing can effectively reduce sediment loading and that these reductions can be measured at the small watershed scale.

Macroinvertebrate communities, our biological assessment variable, demonstrated mixed responses to riparian restoration. Among treated streams, composition, diversity, and total richness had weak improvements, but changes were variable between treatment streams and over postrestoration sampling periods. Community metrics indicated that macroinvertebrate community composition and structure changed little after restoration in Slab Cabin Run and Cedar Run. In other studies, effects of improvements or additions of riparian buffers range from small to large shifts in community metrics but may depend on buffer characteristics. Streams fenced from riparian grazing may respond with significantly improved macroinvertebrate richness or increases in pollution-intolerant taxa (Rinne 1988; Galeone 2000). In a Pennsylvania study, Galeone (2000) found substantial increases in macroinvertebrate richness 3–4 years after stream bank fencing. But other studies have reported relatively small changes in macroinvertebrate abundance, evenness, and diversity after riparian vegetation plantings (Suren & McMurtrie 2005).

Macroinvertebrate responses to riparian treatments may vary depending on riparian buffer widths and lengths (Parkyn et al. 2003) and vegetation type (Sovell et al. 2000). Buffer dimensions and vegetation characteristics influence macroinvertebrates in adjoining stream reaches because of their differing abilities to filter run-off, to contribute woody debris and food resources, and to influence stream habitat characteristics (Sweeney 1992; Vought et al. 1995; Lyons et al. 2000; Schultz et al. 2004).

Despite variable community metrics, densities of macroinvertebrates improved in restored streams. Relative to Spring Creek, macroinvertebrate numbers increased from prerestoration values in both Cedar Run and Slab Cabin Run. Densities most likely responded to decreased fine sediments from stabilized banks in pasture reaches. Cedar Run's decline in fine particles at all sampling stations was accompanied by macroinvertebrate density increases after riparian treatments. Reduction in fine particles at two of the four sampling stations also coincided with higher densities in Slab Cabin Run after restoration. On the other hand, declines in macroinvertebrate densities in Spring Creek in August 2001 and May 2002 occurred with higher proportions of substrate fines. In contrast to our study, Rinne (1988) observed higher densities and biomass in grazed rather than buffered stream sections; however, high numbers of pollution-tolerant taxa dominated areas disturbed by grazing. Implementation of stream channel and riparian restoration resulted in increased densities of macroinvertebrates in an Indiana study (Moerke et al. 2004). In other field studies, increases in abundance and biomass occur with decreasing sedimentation (Lenat et al. 1981; Lemly 1982; Angradi 1999). Macroinvertebrates with preferences for larger particle sizes may become more abundant in habitats with reduced amounts of fine particles.

Low water levels throughout the posttreatment study period may have dampened the effects of riparian

treatments on macroinvertebrate communities. In Slab Cabin Run, macroinvertebrate metrics declined by August 2001 and May 2002 as low flows became more severe. Spring Creek and Cedar Run metrics also dropped off by the second year of postrestoration sampling. Studies of intermittent streams and drought-stressed streams demonstrate that community composition, evenness, and diversity can be altered in streams during low flows (Pires et al. 2000; McManaman 2001). Drought conditions may also confound changes in macroinvertebrate abundance, perceived as restoration responses. However, studies have found that abundance may be reduced rather than increased by drought conditions (Wood & Petts 1999; Fritz & Dodds 2004). Streams might have demonstrated stronger community and even greater abundance responses if drought conditions were not present. Longer-term monitoring over varying hydrologic conditions would further elucidate biological responses to riparian treatments.

Despite water levels and study limitations, changes in physical and biological variables were detected at multiple scales within the treatment catchments. Localized improvements in bank erosion and revegetation resulted in overall reduction in sediment detected in stream substrates in treated pasture reaches and in reaches downstream of treatments. Biological monitoring of macroinvertebrates also found that stream reaches with riparian buffers may have sufficiently improved habitat conditions and water quality for communities to have limited changes.

Conclusions

A 3- to 4-m grass buffer established after streamside fencing, installation of rock-lined animal crossings, and bank stabilization with rock resulted in stream responses that were measurable at the reach and catchment scales. At the reach scale, responses were mixed: stream bank stability was enhanced, channel morphology was unchanged, fine substrates decreased in one stream, and benthic macroinvertebrate diversity was unchanged, but density increased. At the catchment scale, TSS during base flow and storm flow decreased substantially in both streams after riparian treatments.

Implications for Practice

- After livestock were excluded from riparian areas with streamside fences, vegetation, consisting primarily of grasses, colonized quickly.
- Narrow grass buffers when combined with rock-lined animal crossings and eroding stream banks stabilized with rock proved effective in substantially reducing sediment loads transported out of the catchment.
- Reach scale benefits of establishing grass buffers included some reductions in fine substrates and increases in macroinvertebrate density.

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