Responses of Streams to Restoration of Intensively Grazed Riparian Areas: Spring Creek, West Branch Susquehanna River Watershed

Final Report

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Executive Summary

Sediment originating from intensively grazed pastures was linked to depressed reproduction of brown trout in Spring Creek, a limestone stream in the West Branch of the Susquehanna River basin. Public agencies and private organizations initiated a project in 1990 that was designed to restore degraded riparian areas and reduce sediment loading. Improvements included stabilizing eroding stream banks, installing rock-lined animal accesses and stream crossings, and constructing fences along the streams. Restoration efforts were concentrated in two tributaries to Spring Creek. There were 4.1 km of stream flowing through unfenced riparian pastures in the Slab Cabin Run basin, and 67% of this stream length was improved and well maintained through 2003. There were 2.5 km of stream flowing through unfenced pastures in the Cedar Run basin, and 98% of this stream length was improved and well maintained through 2003. Upper Spring Creek, which had no unfenced riparian pastures was used as a reference.

The objective of this study was to quantify the effects of streambank fencing and stabilization in Slab Cabin Run and Cedar Run. We measured channel morphology, substrate composition, stream temperatures, discharge, water quality, macroinvertebrate, and fish communities prior to restoration in 1991-1992. Restoration activities occurred during 1992-1998, and post-restoration assessments were completed in 2001-2003.

Stream bank fencing resulted in revegetation of eroded banks with primarily grasses and a few shrubs. No trees were planted, and none invaded the buffer zone. Stream channel morphology did not change after restoration. Total suspended solids (TSS) during baseflow in Cedar Run declined by 36 to 45% and in Slab Cabin Run TSS declined by 77 to 82% after restoration, though below average discharge contributed

somewhat to these reductions. During storm flow there were significant reductions in TSS in one of two years in Cedar Run and in both years in Slab Cabin Run. There were no significant changes in concentrations of nitrogen or phosphorus after restoration. The amount of fine sediments in the substrate of Cedar Run declined after restoration, but similar changes were not evident in Slab Cabin Run. There was no indication that stream temperatures changed as a result of stream bank restoration. Composition of the macroinvertebrate communities did not change, but there were significant increases in densities of macroinvertebrates after restoration. Composition of the fish communities and densities of wild brown trout were similar before and after restoration. In summary, stream bank fencing and bank stabilization led to revegetation of eroding stream banks, reductions in total suspended solids, and increases in densities of macroinvertebrates.

Introduction

Riparian zones play a critical role in maintaining healthy stream systems. Among their functions, riparian zones regulate water temperatures, shape channel morphology, and provide cover for fishes (Gregory et al. 1991; Naiman and Décamps 1997). Vegetation provides stability for streambanks, especially during high flows, and can mediate runoff from urban and agricultural sources by filtering overland and subsurface flows (Osborne and Kovacic 1993). In streams with livestock grazing in riparian zones, water quality is diminished, habitats for feeding and reproduction by stream organisms become degraded, and aquatic communities are less diverse and less productive than in streams without grazing (Armour et al. 1991; Fleischner 1994).

Without vegetated riparian zones in livestock pastures, pollutant-laden runoff drains directly into the adjacent waterways. The magnitude of nutrient pollution from agriculture land is highlighted by a Danish study, which reported that losses of total nitrogen and total phosphorus from agricultural watersheds were fourteen and four times greater, respectively, than from undisturbed watersheds (Kronvang et al. 1995). The amount of agricultural land and its land use determine how much pollution is exported from a watershed. In a study of agricultural basins in central Texas, nitrogen and phosphorus concentrations in stormwater increased with the proportion of land used in cattle farming (McFarland and Hauck 1999). Cattle pastures, as well as manure application fields, contribute to excessive nutrient loadings. Up to 20% of nutrients are lost from manure application through runoff, if rainfall immediately follows application

(Carpenter et al. 1998). Nutrient pollution not only contaminates waterways adjacent to pastures, but also may eventually reach lakes and coastal waterways. Excessive nutrients result in many ecological and nuisance problems. For instance, eutrophication, anoxia of bottom waters, algal blooms, and declines in marine organisms occur with increased nutrient inputs into estuaries and coastal systems (Fenn et al. 1998).

Riparian land use is linked both directly and indirectly to game fish populations and fish communities. Changes in fish populations are associated with reduced habitat quality in streams with riparian pastures. Increased stream widths, decreased depths, and decreased bank-full heights result from grazing (Keller and Burnham 1982; Knapp and Matthews 1996). The resulting shallower and wider channels do not provide the protection of deep and cool pools, which serve as thermal refugia for fish when streams and rivers warm during summer months (Elliott 2000; Heggenes 2002). Additionally, removal of riparian vegetation by livestock decreases overhead cover for fishes. In one study, the experimental removal of overhanging vegetation and undercut banks resulted in decreased rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*) densities (Boussu 1954). This pattern is evident in streams with grazing. Abundance, biomass, and total length of salmonids decreased in stream sections with riparian grazing, compared to areas with grazing exclosures (Keller and Burnham 1982; Knapp and Matthews 1996). Other fish respond similarly to removal of riparian grazing. Populations of several Nevada desert spring fishes increased after excluding cattle from streamside pastures (Taylor et al. 1989).

Streambank erosion from riparian grazing and the subsequent deposition of sediment degrade spawning habitat for salmonids. Clean, coarse gravel is required for brown trout (*Salmo trutta*) redds (Ottaway et al. 1981). When substrate is covered by fine particles, survival of deposited eggs declines. Fine particles in redds reduce water flow that brings oxygen to developing embryos and carries away metabolic wastes. Several studies have shown that salmonid embryo survival and dissolved oxygen are lowered by high amounts of substrate fines (Turnpenny and Williams 1980; Tappel and Bjornn 1983; Witzel and MacCrimmon 1983).

The presence and quantity of streamside canopy has implications for stream temperatures, and in turn, for thermally sensitive stream organisms. The length and width of forested riparian buffers accounted for more than 90% of the variation in stream temperatures in southern Ontario trout streams (Barton et al. 1985). Because temperature can have a profound effect on the behavior and physiology of salmonids (Ojanguren et al. 2001), including their growth, feeding behavior, and activity levels, changes in water temperatures due to grazing may have population level effects. Studies on grazing have related salmonid biomass to temperature or to measures of solar input. Platts and Nelson (1989) found that unobstructed sun arc and thermal input were negatively associated with salmonid biomass. Similarly, in another study, increases in riparian canopy cover and decreases in daily maximum water temperature were associated with increases in rainbow trout biomass (Li et al. 1994).

The effects of riparian zone disturbance are seen in the fish community as a whole, owing to changes in food resources and reproductive habitats. Jones et al. (1999) found that removal of riparian forest resulted in a shift in reproductive guilds due to changes in the availability of suitable spawning habitat. Densities of benthic modifiers (those that modify substrate to cover eggs, i.e. some cyprinids and salmonids) decreased, while densities of pit spawners (those that excavate a depression on a soft substrate, i.e. some centrarchids) increased. Feeding requirements also influence fish species composition in streams with riparian disturbance. In warmwater Missouri streams, the abundance of benthic insectivores and herbivores declined in response to increased levels of fine sediments from agricultural activities (Berkman and Rabeni 1987). Schlosser (1982) found similar changes in feeding groups in streams where row crops replaced riparian vegetation; insectivore and insectivore-piscivore feeding guilds declined, while omnivores became more abundant. In both studies, shifts in the food base of these streams were believed to cause changes in feeding guilds.

The biological effects of riparian grazing are also evident in macroinvertebrate communities. The abundance of pollution intolerant taxa generally decreases with a higher proportion of agriculture land use (Dance and Hynes 1980; Lenat and Crawford 1994). In studies that specifically examined grazing, similar shifts in community composition occurred; Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness decrease in stream sections with grazing (Scott et al. 1994; Fritz et al. 1999; Weigel et al. 2000), while more tolerant taxa, like Mollusca and Oligochaeta, become abundant (Scott et al. 1994). Sedimentation produces responses in macroinvertebrate communities analogous to those associated with grazing. Overall densities, taxa richness, diversity, and biomass decrease as levels of fine sediments increase (Lenat et al. 1981; Lemly 1982; Angradi 1999; Fritz et al. 1999; Zweig and Rabeni 2001).

Macroinvertebrate community response to riparian grazing is determined, at least in part, by substrate preference. Taxa that require fast-moving water or need relatively clear water to protect filter-feeding or gill structures are absent from areas that have high amounts of fine sediment, while other taxa that can contend with sediment accumulation on their bodies occur in silty or sandy habitats (Chutter 1969). When increased sedimentation occurs, macroinvertebrates unsuited for silty habitats may be eliminated. One study documented the presence of fine organic particles and filamentous bacteria on the body surfaces of macroinvertebrates in stream sections that received sediment and nutrient pollution from pastures (Lemly 1982); the authors attributed the decline of some Trichoptera and Diptera to harmful particle accumulation on filter-feeding structures. Studies in macroinvertebrate behavior suggest that substrate particle size is an essential habitat feature. Cummins and Lauff (1969) studied the substrate preferences of a number of different aquatic insect species in the field and in the laboratory. In behavioral experiments, some insects, like the stonefly, *Perlesta placida*, the riffle beetle, *Stenelmis crenata*, and the caddisflies, *Pycnopsyche guttifer* and *P. lepida*, selected narrow ranges of particle sizes. Other researchers found that in laboratory experiments, the stonefly, *Skwala americana*, consistently chose unembedded cobble over embedded cobble even in the presence of predators (Haro and Brusven 1994). Deposited sediment that reduces the availability of coarse substrate in streams with riparian grazing contributes to the degradation of macroinvertebrate communities.

Decreased riparian vegetation from grazing and increased solar input may result in increased macroinvertebrate production. Production for some dipterans and some ephemeropterans increases with a higher proportion of open canopy higher food quality due to more algal growth may be responsible for increased macroinvertebrate production (Behmer and Hawkins 1986). In streams with grazing, solar input was positively correlated with algal biomass and total herbivorous invertebrate biomass (Li et al. 1994).

However, light may not be limiting stream primary production in streams. Algal primary production is maximized at levels as low as 20% sunlight (Gregory et al. 1987). Therefore, further increases in sunlight above the 20% saturation from removal of riparian canopy would not have any effects on algal primary production and would not promote additional macroinvertebrate production. Additionally, benefits to macroinvertebrate production from higher solar input may be outweighed by other unfavorable conditions due to riparian grazing.

Alteration of stream temperatures due to riparian disturbance may influence macroinvertebrate communities. Increased temperatures in deforested streams greatly change life history traits of macroinvertebrates, such as growth rate, survivorship, adult size and fecundity, and timing of reproduction (Sweeney 1993). Another study found that temperature differences between California streams were related to taxa composition, densities, and biomass of macroinvertebrate assemblages (Hawkins et al. 1997); community structure was thought to be dissimilar between streams because of altered developmental rates due to temperature or because of different thermal tolerances of insect taxa.

To ameliorate the effects of riparian grazing, streambank fencing is one technique widely used by natural resource managers. It excludes animals from the streambanks, reduces streambank erosion, and prevents direct inputs of animal waste to the stream. Large-scale watershed nutrient reduction plans often include streambank fencing. For instance, Pennsylvania’s Chesapeake Bay Nutrient Reduction Strategy allocated $2 million for fencing streambanks (Pennsylvania Department of Environmental Protection 1996). Despite the extensive implementation of this method and large monetary investment, relatively few scientifically sound studies of streambank fencing have been conducted.

Riparian management plans are not often conducive to well-designed studies. Restoration techniques are commonly applied without measurement of baseline conditions. One of the major criticisms of restoration is that inferences about its effectiveness are made without any pre-restoration data (Platts 1982). Using a paired watershed approach would be a better method for measuring the effects of restoration; the approach compares the relative change in a manipulated watershed to another similar watershed that serves as a reference. But, this approach is often impossible owing to the practical constraints of controlling land use practices on a large scale. Instead, reference and fenced reaches are commonly studied within the same stream. This flawed approach assumes that that upstream reaches do not influence downstream reaches (Rinne 1988). Use of replication and adequate controls would result in more meaningful research, but have rarely been accomplished in current riparian studies (Rinne 1999).

The objective of this study was to quantify the effects of streambank fencing and stabilization in two small, central Pennsylvania streams. We measured channel morphology, substrate composition, stream temperatures, discharge, water quality, macroinvertebrate, and fish communities prior to restoration in 1991-1992. Restoration activities occurred during 1992-1998, and post-restoration assessments were completed in 2001-2003.

Study Area

The study area is located within the Spring Creek watershed (381 km^2) in Centre County, Pennsylvania (Figure 1). The watershed receives approximately 97 cm of precipitation annually (based on records from 1931-2001 at the State College Weather Station); mean air temperatures range from –4˚C in January to 22˚C in July. During this project, 1991-2003, annual precipitation ranged from 77 to 151 cm and stream flow varied accordingly (Figure 2).

The Spring Creek watershed lies within the Valley and Ridge physiographic province, which is characterized by sandstone ridges and limestone valleys. The three study basins, Spring Creek, Cedar Run, and Slab Cabin Run are adjacent (Figure 1), sharing similar geology and soils. Headwaters of the upper Spring Creek basin and of Slab Cabin Run originate on Tussey Mountain. Streams flowing from this sandstone ridge are typically low in pH (<7.0) and alkalinity. When these streams reach the valley floor, groundwater from limestone aquifers accrues to the streams and chemistry changes. Headwaters of Cedar Run originate in the valley floor. Valley streams have relatively high pH (7.7-8.4), high total alkalinity (196-208 mg/L as $CaCO_3$), and high concentrations of calcium (47-68 mg/L). Additional details about valley streams are provided by Carline et al. (1991).

The upper portion of Spring Creek serves as the reference stream. Its headwaters drain a forested area on Tussey Mountain; upon reaching the valley, Spring Creek flows through agricultural land. Even though about one-third of the upper basin drains agricultural areas, Spring Creek has no riparian grazing. It has the highest annual discharge per unit basin area, which is more than two times that of Cedar Run and Slab Cabin Run (Table 1). Cedar Run and its tributary Mackey Run previously flowed through unfenced pasture for about 60% of their lengths; Wohl and Carline (1996) found that 50% of the streambanks in riparian pastures in the Cedar Run basin were eroding. The upper basin of Slab Cabin Run has predominantly agricultural land use. About 75% of stream length flowed through riparian pastures prior to restoration, where streambank erosion was evident in all pastures. A wellfield adjacent to the stream channel withdrew 10.3 million liters per day in 2001 for local drinking water supply (D. Nevel, State College Water Authority, personal communication).

Figure 1. Map of Spring Creek, Cedar Run, and Slab Cabin Run basins.

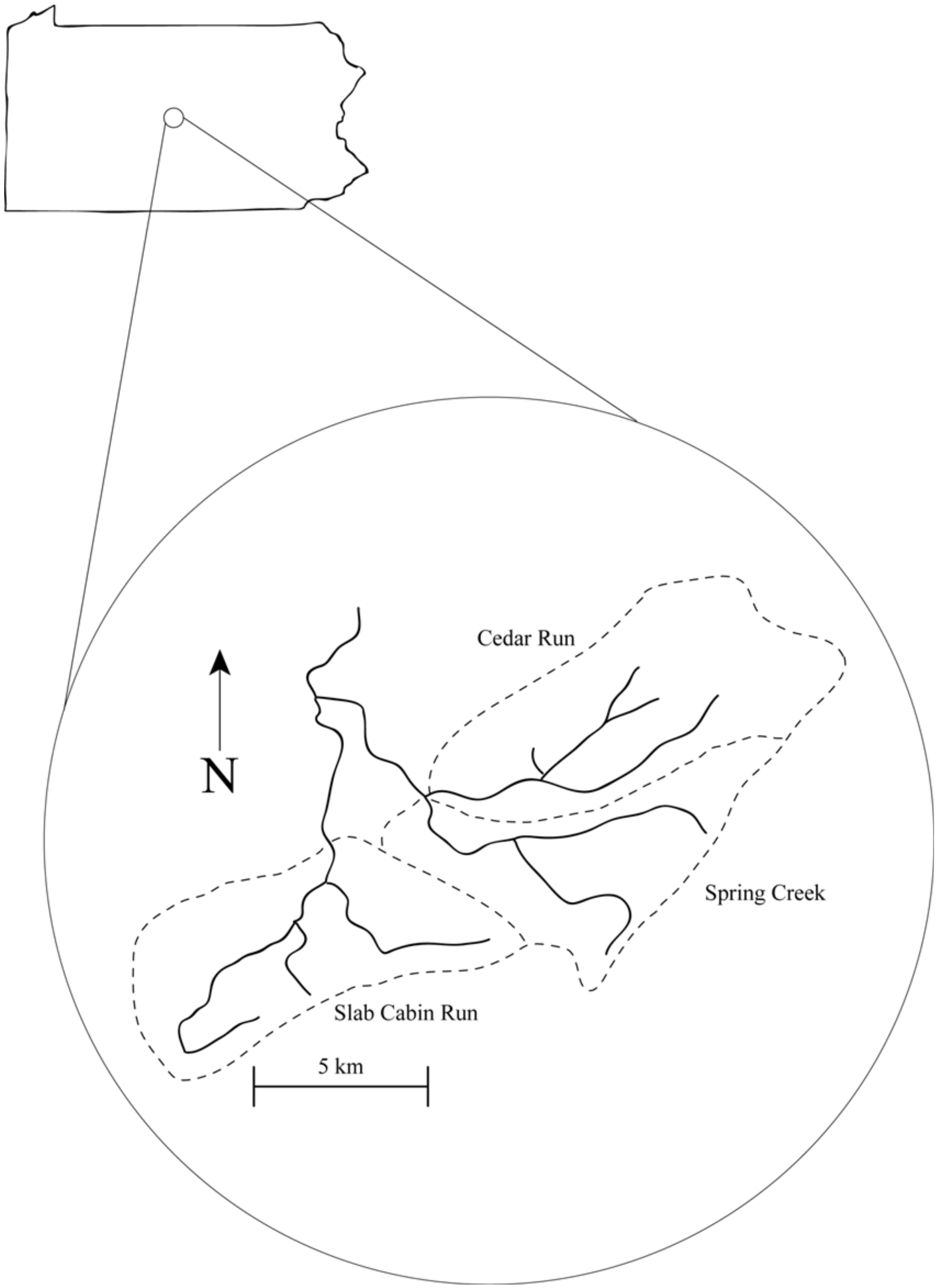

Figure 2. Mean annual stream flow (line with diamonds) at the Houserville gauging station on Spring Creek and total annual precipitation at State College, 1985-2002. The gauging station is 2.1 km downstream of the confluence of Slab Cabin Run and Spring Creek.

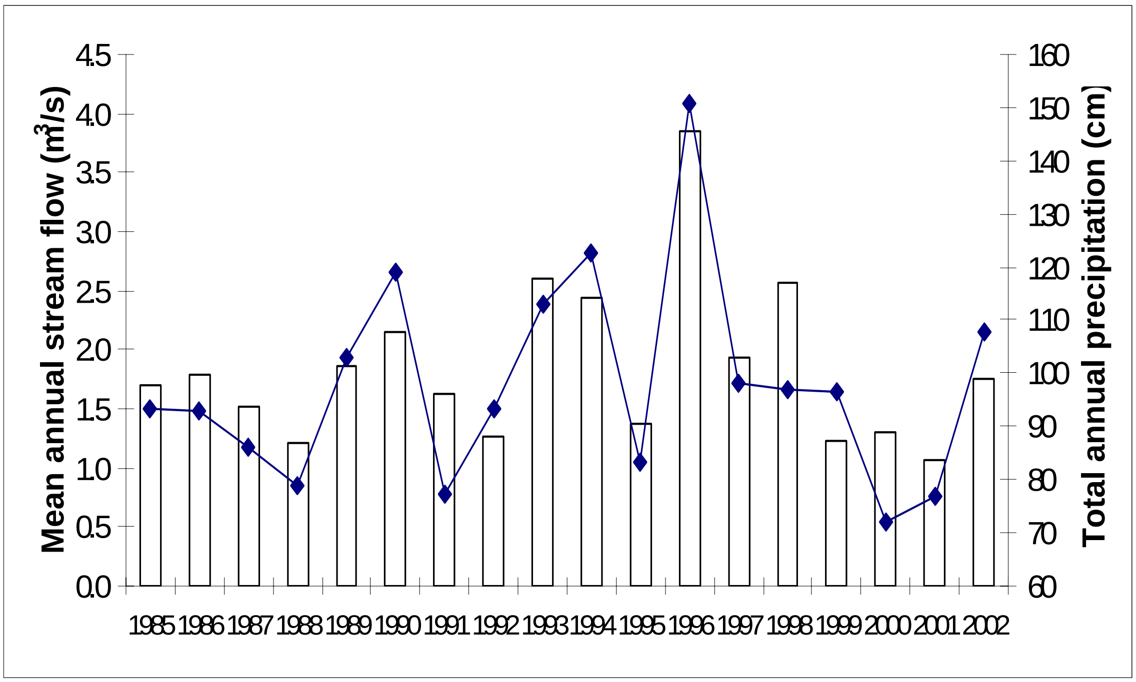

Table 1. Physical characteristics and land use of study stream basins within the Spring Creek watershed.

	Spring Creek	Cedar Run	Slab Cabin Run
Basin area (km^2)	34	46	44
Land use (% area)			
Agriculture	33	85	55
Forest	33	15	43
Urban	33	0	2
Channel length (km)	6.0	7.9	14.4
Median daily discharge[a] ($m^3 \cdot s^{-1}$)	0.38	0.19	0.10

[a]For the period September 1991 to August 1992, from Wohl and Carline (1996).

Two studies documented stream variables prior to restoration. In Spring Creek, Cedar Run, and Slab Cabin Run, in 1991-1992 (prior to riparian restoration) suspended sediment concentrations and sediment loading, stream channel morphology, water temperatures, macroinvertebrate communities, and fish communities were monitored (Wohl and Carline 1996). In 1993 and 1994, after restoration had commenced in Slab Cabin Run, but prior to restoration efforts on Cedar Run, nutrient and sediment concentrations and loadings in Cedar Run and Spring Creek were evaluated (Schnabel and Carline 1995). Results from both studies revealed that stream conditions were degraded due to grazing; stream temperatures were more variable and sediment and nutrient concentrations were greater in Cedar Run and Slab Cabin Run, relative to Spring Creek. Increased fine sediment in stream substrates decreased permeability in potential brown trout spawning areas in grazed streams; macroinvertebrate densities and brown trout densities were also depressed in Cedar Run and Slab Cabin Run (Wohl and Carline 1996).

Restoration Efforts and Techniques

The Centre County Conservation District, the Spring Creek Chapter of Trout Unlimited, and the Pennsylvania Cooperative Fish and Wildlife Research Unit jointly implemented riparian restoration efforts, including streambank fencing, streambank stabilization with rip-rap, and the installation of animal accesses and crossings.

The restoration effort in the Slab Cabin Run basin began in 1992 and was started later in 1994 in the Cedar Run basin. All restoration activities were concluded by 1998. In the Cedar Run basin, 4,019 m of fence were erected, 14 animal accesses were installed, and 245 m of bank were stabilized with rock, while in the Slab Cabin Run basin 5,488 m of fence were erected, 26 animal crossings and accesses were installed, and 1,875 m of bank were stabilized (Table 2). Fence material, which was mostly made of high tensile wire, though fencing materials varied according to the property owners' specifications. Approximately 3 m of buffer was created between fences and the streams. On Cedar Run and Slab Cabin Run, 98% and 67%, respectively, of pastured stream length was fenced. Landowners agreed to maintain fencing for 10 years after installation. Accesses permitted animals to reach the channel from only one side of the stream, while crossings spanned both streambanks. Accesses and crossings were installed at riffles or runs, where there was rocky substrate. Streambanks were graded (3:1) and a layer of 15-cm diameter limestone was placed on the slopes to create a ramp 4 to 5 m wide. Ramps were then covered with smaller rock, consisting of 3- to 5-cm diameter limestone. Contractors placed 15-cm limestone on vertical streambanks that were likely to continue eroding in the absence of disturbance by grazing animals.

Table 2. Landowners and locations of sampling sites and of streambank restoration projects in the Spring Creek, Cedar Run and Slab Cabin Run basins. Sampling site designations are shown in Figure 3. Length of fence installed included both banks. Typically, riprap was installed on only one bank. Sites CR1, CR4, SL2, and SL4 were ungrazed; no riparian restoration work was undertaken.

Basin, stream, and property owner	Sampling site	Year	Length of fence installed (m)	Number of crossings and accesses	Length of bank stablized with rip-rap (m)
Spring Creek Basin					
J. Westrick	SP1				
Cedar Run Basin					
Unnamed tributary to Mackey Run					
T. Potter		1993	457	1	0
Mackey Run					
R. Ackley		1994	244	2	15
L. Mothersbaugh		1994	518	2	59
R. Kreidler		1994	0	1	0
Cedar Run					
J. Williams	CR4				
C. Lingle	CR2	1997	1539	3	0
J. Meyer		1994	0	1	0
R. Gilliland	CR3	1998	1261	4	0
S. Smith	CR1				
Slab Cabin Run Basin					
Slab Cabin Run					
G. Driebelbis[a]		1996	216	0	0
F. Scott	SL4				

C. Homan[b]	SL3.5	1993	695	4	91
J. Musser[c]		1993	232	1	0
S. Everhart[d]	SL3	1993-1995	762	7	0
B. Pasquinelli		1994	0	2	23
R. Wasson	SL2.5	1993-1994	1981	2	244
R. Pifer		1992	244	3	0
R. Everhart		1994	0	2	137
State College Water Authority	SL2				
J. Meyer	SL1	1992	1358	3	15
Roaring Run					
C. Hess[e]		1993	0	2	61

Current Owners
[a] Joseph Dionisio
[b]Joseph and Delorse Homan
[c]Robert and Barbara Sorisio
[d]Dorothy Jodon
[e]Jeffery and Cindy Harding

Methods

The methods for data collection and the locations of gauging stations and sampling sites were identical to those used in pre-restoration studies (Schnabel and Carline 1995; Wohl and Carline 1996) (Figure 3 and Table 3) with one exception. Because sampling site SL1 on Slab Cabin Run was dry for extended periods during the post-restoration study, another site, SL3.5, was substituted for site SL1 for the purposes of macroinvertebrate and substrate sampling. Sites SL1 and SL3.5 were both located in riparian pastures and had severely eroding streambanks prior to restoration; restoration at both sites included streambank fencing, addition of animal accesses, and streambank stabilization.

Animal crossings and accesses may cause localized stream disturbance and can be sources of sediment (Fritz et al., 1999). The location of sampling relative to animal crossings and accesses could influence some stream variables measured in this study. Our sampling sites in pastured sections of Cedar Run and Slab Cabin Run were located downstream of crossings.

Channel Morphology

To determine any changes in channel morphology and streambank condition after restoration, surveys were conducted at 10 sites in July 2001 (Figure 3 and Table 3). Cross-sections of the stream channel were used to measure width, depth, velocity,

Figure 3. Map of sampling sites in the Spring Creek, Cedar Run, and Slab Cabin Run basins used in pre-restoration (1991-1992 and 1993-1994) and post-restoration (2000-2003) studies.

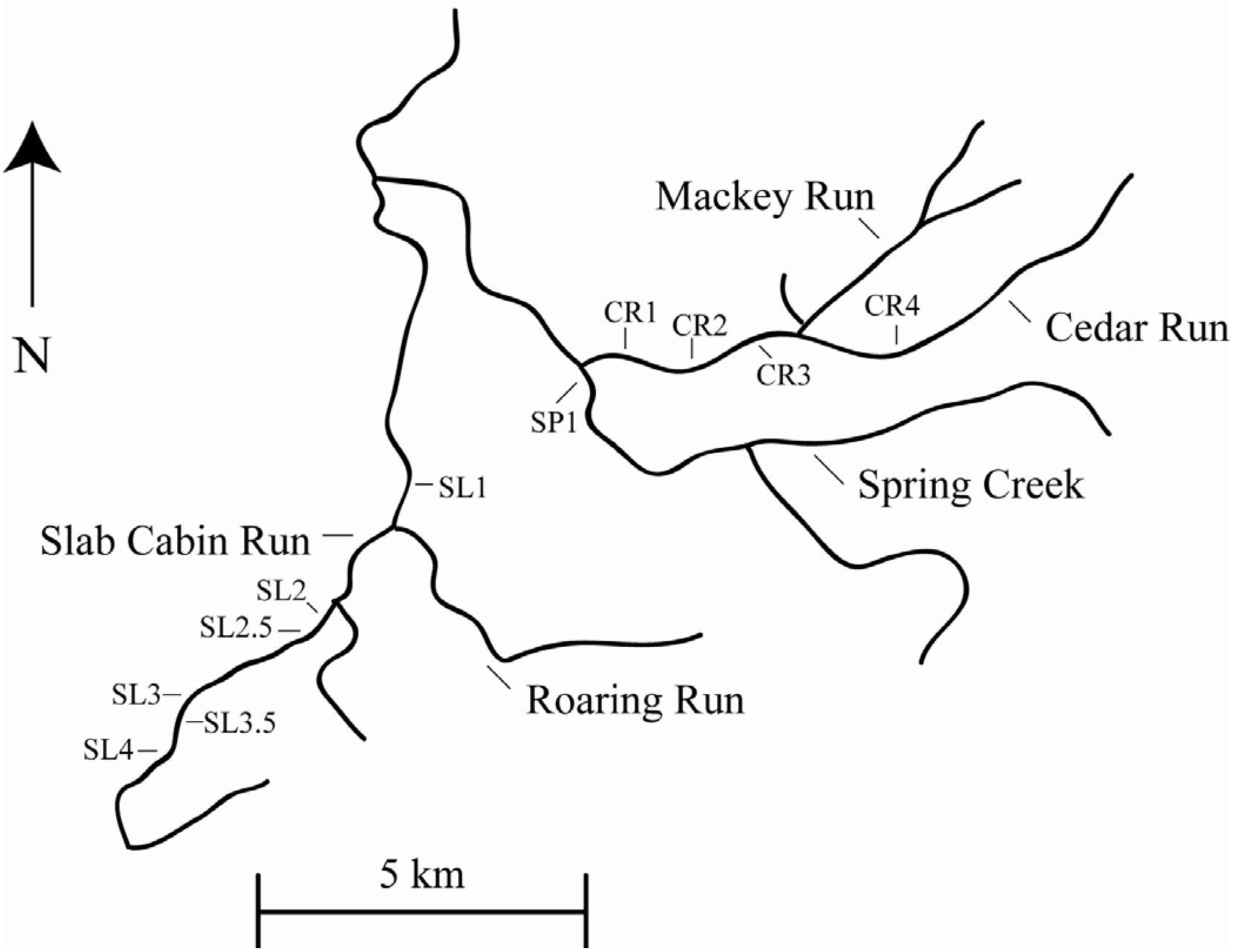

Table 3. Sampling sites and gauging station locations in study streams and types of data collected by location in the post-restoration study (2000-2003). Longitude and latitude are reported in decimal degrees. Types of data are represented by the following code: SD = stage and discharge, W = water quality, T = stream temperatures, C = channel morphology, S = substrate composition, M = macroinvertebrate communities, F = brown trout densities and fish communities.

Sampling sites and gauging stations.	Distance from gauging stations (km)	Longitude	Latitude	Type of data collected
Spring Creek				
SP1 and Gauging Station	0	-77.7982	40.7928	SD,W,T,C,S,M,F
Cedar Run				
CR1 and Gauging Station	0	-77.7968	40.7939	SD,W,T,C,S,M,F
CR2	2.2	-77.7764	40.7914	C,S,M,F
CR3	4.4	-77.7688	40.7955	C,S,M,F
CR4	5.1	-77.7554	40.7935	C,S,M,F
Slab Cabin Run				
Gauging Station	0	-77.8353	40.7850	SD,W,T
SL1	0.8	-778316	40.7813	C,F
SL2	2.7	-778391	40.7707	C
SL2.5	3.2	-77.8421	10.7678	C,S,M,F
SL3	5.7	-77.8539	40.7557	S,M
SL3.5	6.6	-77.8643	40.7520	S,M
SL4	7.9	-77.8684	40.7506	C,S,M,F

and percent un-vegetated streambank. The same cross-sections were used during the pre-restoration and post-restoration phases of this study. Width, depth, and velocity measurements were taken at 10 to 15 points across the stream channel at ten or more cross-sections per sampling site. Velocity was measured at 0.6 of total water depth from the water surface using a portable flowmeter (Marsh-McBirney Flow-mate 2000). Discharge was estimated using the current-meter method (Buchanan and Somers 1984). At both banks of the cross-section, the percentage of un-vegetated streambanks was estimated as 0, 25, 50, 75 or 100%; un-vegetated banks were defined as having exposed soil without vegetative cover. Type of vegetation was categorized as grass, shrubs, or trees on both streambanks. At ten points along each transect, substrate particle size was categorized as follows: silt-sand (< 2 mm), gravel (2-64 mm), cobble (> 64 mm), and bedrock.

Discharge

Gauging stations were established during pre-restoration phase of the study at the mouth of each basin. The gauging station on Spring Creek was located 50 m above the confluence with Cedar Run at site SP1 (Figure 3 and Table 3). The gauging station on Cedar Run (CR1) was located 75 m upstream of the confluence with Spring Creek. The gauging station on Slab Cabin Run was 15 m upstream of the Business Route 322 bridge and 0.3 km downstream from site SL1. Water levels were recorded at the gauging stations from January 1, 2001 to January 13, 2003. Pre-restoration water levels were recorded from September 1991 through August 1992 (Wohl and Carline 1996) and from October 1993 to October 1994 (Schnabel and Carline 1995).

Water level recorders (Design Analysis DH-21 Submersible Waterlogger), installed in stilling wells, were used to record stream stage at 0.5-h intervals. Staff gauges with 1-cm increments were used to calibrate water level recorders. Ice in the channel and 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. The water level recorder operated by the ClearWater Conservancy was the same model as used in this study and discharge was estimated using the same methods. Using regression analyses, discharge at the lower gauging station on Slab Cabin Run was a good predictor of discharge at the gauging station used in this study ($R^2 > 0.95$).

To develop rating curves, stream discharge was measured 10 to 15 times at a range of stage values at each gauging station. Methods used to estimate discharge were the same as those used in the channel morphology transects. Regression analyses of stage and discharge measurements were used to construct each rating curve.

Water Quality

To characterize post-restoration water quality, weekly baseflow water samples and storm water samples were taken at gauging stations from January 2001 to January 13, 2003. A total of 43 storm events were sampled. Storms were defined as periods when discharge increased by 25% or more from baseflow. Automatic water samplers (Hach American Sigma 900) typically took samples hourly for 24 hours during a storm event. Every attempt was made to sample both restored streams and Spring Creek concurrently during storm events. Because of equipment malfunctions not all storms were sampled concurrently on all streams. Six samples per storm, including two each from the rising limb, peak, and falling limb of the hydrograph, were analyzed for sediment and nutrient content. Mean sediment and nutrient concentrations in post-restoration stormflow samples were calculated for each storm to mimic the method of sample collection during pre-restoration sampling. Pre-restoration stormflow samples from the Schnabel and

Carline (1995) study were flow-weighted composite samples, which had been collected during the increase and decrease of streamflow. After Slab Cabin Run ceased flowing at the gauging station on July 25, 2001, no further grab samples or stormflow samples were taken until permanent flow resumed on April 2, 2002. Flow ceased again on August 26, 2002 and resumed on November 21, 2002.

Water samples were analyzed for sediment and nutrient content. Turbidity was measured on all samples with an Orbeco-Hellige Digital Turbiditimeter (Model 965). A subset of samples was analyzed for total suspended solids (TSS), using method 2540-D from Standard Methods for the Examination of Water and Wastewater (American Public Health Association 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. Then, the relationships were used to estimate TSS for samples when only turbidity was measured. The Penn State Environmental Resources Research Institute (ERRI) Water Laboratory performed nutrient analyses in accordance with Standard Methods for the Examination of Water and Wastewater (American Public Health Association 1995). Analyses included nitrate-nitrogen (dissolved; method 4500-NO_3-E), total nitrogen (particulate and dissolved; method 4500-N_{org}D), ortho-phosphate (dissolved; method 4500-P E ortho-phosphate), and total phosphorus (particulate and dissolved; method 4500-P B.5). All nutrient samples were analyzed within 24 h of collection or stored on ice until they could be analyzed, usually within 48 h.

Sediment and nitrate-nitrogen yields were calculated as the product of the concentration and stream flow. Daily mean constituent concentrations were calculated on storm event days. TSS and nitrate-nitrogen concentrations were linearly interpolated between sample days to estimate daily concentrations. Daily sediment and nitrate-nitrogen loads were summed over the entire sampling period, as an estimate of total annual yields.

Substrate Composition

Substrate similar to brown trout spawning habitat was sampled in May 2001 at each of nine sites (Figure 3 and Table 3); samples were taken in areas where velocity ranged from 0.25 to 0.57 $m \cdot s^{-1}$ and depth ranged from 0.2 m to 0.5 m, described as the ranges for brown trout redds in the Spring Creek watershed (Beard 1990). A stovepipe sampler (McNeil and Ahnell 1964) with a 10-cm diameter was used to collect four samples at each of nine sites. 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.

The following equation was used to calculate the Fredle index (F_i), a measure of substrate permeability (Lotspeich and Everest 1984):

$F_i = D_g/S_o$

where D_g = geometric mean particle size

S_o = sorting coefficient.

The variable, D_g, represents the central tendency of particle size. S_o represents particle size distribution; it is calculated as the square root of the quotient of the particle size at the 75^{th} percentile and the particle size at the 25^{th} percentile of the cumulative sample weight. The sorting coefficient is inversely proportional to permeability. By combining both variables, the Fredle index has a proportional relationship to salmonid embryo survival-to-emergence (Lotspeich and Everest 1981). Percent fines, defined as percent substrate sample weight of particles less than 1 mm, was also calculated.

Temperature

Stream temperatures were monitored at gauging stations from January 2001 through December 2002 in the post-restoration study. Pre-restoration temperature monitoring occurred from September 1991 to September 1992. Temperatures were measured to the nearest 0.5˚C at 1-h intervals using Ryan Instruments RL 100 Temperature Recorders. Recorders were placed in the streams adjacent to staff gauges, approximately 1 m from one streambank, and rested on the channel bottom. Instruments were checked against a stem thermometer bi-monthly. In May 2001, two temperature recorders required repair and recalibration. As a result, data from temperature recorders operated by the ClearWater Conservancy were used for temperature records in Spring Creek (May – August 2001) and in Cedar Run (May 2001). Temperature recorders are the same model as those used in this project and are located at the same gauging stations. The collection of temperature data in Slab Cabin Run was limited because of channel ice during winter months and no flows after a summer drought; water temperature data are available only from February 2001 through July 2001.

Macroinvertebrate Communities

Triplicate samples of the macroinvertebrate community were taken from riffles using a Surber sampler at each of nine sites in August 2000, May and August 2001, and May 2002 (Figure 3 and Table 3). Samples were fixed in 10% formalin and transferred to 90% ethanol. Insect taxa were identified to genus, except the Dipteran family Chironomidae, using a key by Merritt and Cummins (1996); all other invertebrates were identified to class or other lowest taxonomic level possible. The Shannon Diversity index (Pielou 1975), density, taxa richness, Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness were determined for all samples. The Shannon Diversity index (H′) is calculated as follows:

$$H' = -\Sigma\ p_i \log_e p_i$$

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 five is generally accepted as the highest score (Washington 1984).

Fish Communities and Brown Trout Densities

Direct current electrofishing gear (200 V) was used to survey fish communities in August 2000, May 2001, August 2001, and May 2002 at each of nine sites (Figure 3 and Table 3). Brown trout densities were estimated with the Zippin successive removal method (Zippin 1958) in 200-m sections. All brown trout were weighed, measured for total length, and released. In August surveys, age-0 brown trout were separated from age-1 and older fish using length frequency distributions; thus, densities were estimated separately for age-0 and age-1 and older fish. Computer software by Van Deventer and Platts (1989) was used to calculate brown trout densities with 95% confidence intervals. All fish species were collected in a 50-m sub-section to characterize the fish community. Most fish were identified in the field and released. Those that could not be identified with certainty were preserved and later identified using a key by Cooper (1983).

Quality Assurance

A Quality Assurance Project Plan was prepared and approved by the EPA Project Officer and Quality Assurance Officer. This plan documented project management, data acquisition, data assessment, and data validation. Protocols specified in the plan were followed.

Data Analyses

We tested null hypotheses that post-restoration data were equal to pre-restoration data. Because of non-normal distribution of data, non-parametric tests were used for most types of variables.

To determine if channel morphology characteristics were different after restoration, percent eroding streambank, mean width, depth, velocities, discharge, and percent substrate category were compared within sites using paired analyses of pre-restoration and post-restoration data.

Nutrient and sediment baseflow data were summarized as the differences between Spring Creek and Cedar Run concentrations and as the differences between Spring Creek and Slab Cabin Run concentrations from samples taken on the same date. Pre-restoration differences and post-restoration differences were compared with paired analyses. By computing differences of same date samples, we considered changes in concentrations in restored streams relative to the reference stream, where water levels were relatively similar. This method was intended to reduce any variation in concentrations due to differences in stream flows between study periods. In a similar manner, differences in mean stormflow concentrations between Spring Creek and Cedar Run and between Spring Creek and Slab Cabin Run from the same storms were calculated for pre-restoration and post-restoration studies; paired analyses compared differences in concentrations between pre-restoration and post-restoration storms. Storm events caused relatively similar increases in stream flows across the study basins. Therefore, any effects of flows on concentrations were reduced by comparing the concentrations in restored streams relative to those in Spring Creek, where flows from storm events of the same magnitude occurred.

Paired analyses tested whether Fredle indices and percent fines were significantly different between pre-restoration and post-restoration collections within each site. To compare stream temperatures between study periods, it was necessary to minimize differences between periods due to differences in ambient air temperatures and stream water levels. We attempted to incorporate this variation by calculating differences between average temperatures in Spring Creek and the restored streams on the same date. Stream temperatures in different streams from the same date would be under the influence of similar environmental conditions. Paired analyses tested whether differences in stream temperatures between the restored streams and Spring Creek were significantly different from pre-restoration to post-restoration studies.

To test whether macroinvertebrate taxa richness had changed in restored streams relative to Spring Creek, at each study site the differences between average number of taxa from pre-restoration and post-restoration samples were calculated for each season. Multiple comparisons were used to test whether changes in the number of taxa in Spring Creek were significantly different from changes in sites in restored streams. Similar analyses were used to test whether diversities had changed in restored streams relative to Spring Creek. To test whether the magnitude of any changes in densities in restored streams were different from changes in Spring Creek, we computed the ratio of macroinvertebrate density at Spring Creek to each of the stations in the treated streams and used a Kruskal-Wallis test to determine if these ratios were different in the pre-restoration period compared to the post-restoration period. We used a similar procedure to determine if density of age-1 and older trout were different in pre- versus post-restoration periods.

Results

Condition of Riparian Properties

In December 2003 and January 2004 each of the properties that had received some type of riparian restoration were inspected to assess condition of the improvements, stream bank stability, and riparian vegetation.

Cedar Run Basin

All of the seven properties that had been subjected to some riparian restoration work have been well maintained. Fences were intact, crossings were in good repair, and banks were vegetated and stable. There are two properties in this watershed that are in need of restoration. When property owners were being contacted in the 1990s and asked if they would be interested in participating in the program, these two property owners declined. Since then, one of the properties has changed hands and the new owners expressed interest in restoration work.

Slab Cabin Run

Five properties that had been subjected to some riparian restoration work have been well maintained. Fences were intact, crossings were in good repair, and banks were vegetated and stable. Two properties changed ownership after restoration work was completed. In one instance, stream bank fencing had been removed, but light grazing had not caused noticeable damage to the banks. In another instance, some stream bank fencing had been removed or not repaired and damage to stream banks was evident. On another farm, much of the original stream bank fencing had been removed or damaged by flooding and the owner had not restored the fence. A part of the pasture was being grazed, though no erosion was evident. On two properties, animal crossings were installed and short lengths of eroding banks were stabilized with rock, but no fences were erected. On one of these properties, a fence was subsequently installed by the riparian owner and banks were well vegetated. On another property, the improvements were intact, but bank erosion was evident where animals began crossing the stream at new locations; this farm merits consideration for fence construction because of new owners.

Riparian Cover

Riparian restoration greatly improved streambank conditions. Dense vegetation developed along the streambanks in fenced areas in Cedar Run sites CR2 and CR3 and Slab Cabin Run sites SL1, SL2.5, and SL3, creating a narrow riparian buffer. In the buffer along fenced sites, grasses and shrubs were present in transects in proportions of 86% and 14% respectively. No trees were observed in the riparian buffers.

Due to re-vegetation and stabilization of streambanks with rip-rap, streambanks were less susceptible to erosion at restored sites along Cedar Run and Slab Cabin Run than during pre-restoration surveys (Table 4). The percent of un-vegetated bank significantly decreased since restoration in Cedar Run sites CR2 and CR3 and Slab Cabin Run sites SL1, SL2.5, and SL3 (Mann-Whitney, $p < 0.05$). Other sites that had been not been previously grazed, including the reference site SP1 on Spring Creek, had proportions of un-vegetated streambank similar to pre-restoration levels, with the exception of Cedar Run site CR1, where stream bank erosion has worsened. At CR1, a dense tree cover on streambanks precluded other vegetative growth and streambanks had exposed soil. A shift in plant cover at CR1 since pre-restoration studies may be responsible for the change in the amount of exposed soil.

Channel Characteristics

During channel morphology surveys in July 2001, stream widths, depths, and discharge had declined in all study streams (Table 4), owing to the low precipitation during 2001. Despite the low precipitation, Spring Creek had the most constant channel characteristics; mean transect widths, velocities, and discharges were not significantly different than those in 1992. In Cedar Run, low precipitation resulted in significantly decreased discharges at all sites (Mann-Whitney, $p < 0.05$). Slab Cabin Run sites had similar changes in channel characteristics; discharges in three sites were significantly lower than during pre-restoration surveys (Mann-Whitney, $p < 0.05$).

In post-restoration channel morphology surveys, a larger proportion of fine particles and fewer coarse particles were found in study streams (Table 5). The relative proportions of silt and sand particles increased at the reference site in Spring Creek, although it was not significantly different from pre-restoration surveys. But, the proportion of cobble in Spring Creek was significantly lower than in pre-restoration surveys (Mann-Whitney, $p < 0.05$). Cedar Run sites, CR1 and CR4, and Slab Cabin Run site, SL4, had significantly increased proportions of silt-sand and significantly decreased proportions of gravel from pre-restoration surveys (Mann-Whitney, $p < 0.05$). Slab Cabin Run site, SL3, had significantly decreased sand-silt and significantly increased proportions of gravel and cobble (Mann-Whitney, $p < 0.05$). Generally, restoration neither decreased the proportion of silt-sand nor increased proportions of gravel and cobble in restored sites. Most sites that had been restored had at least as much or more silt-sand as prior to restoration.

Table 4. Mean values (SE in parentheses) for percent un-vegetated stream banks, channel morphology, stream velocity, and discharge at restored and ungrazed sites in Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1992) and post-restoration (2001) study periods. Pairs of data that are significantly different (Mann-Whitney, $p < 0.05$) are noted with an asterisk.

Stream, site type, and site number	Percent un-vegetated streambank		Mean width (m)		Mean depth (cm)		Mean velocity ($cm \cdot s^{-1}$)		Discharge ($m^3 \cdot s^{-1}$)	
	1992	2001	1992	2001	1992	2001	1992	2001	1992	2001
Spring Creek Ungrazed										
SP1	10 (6) N = 10	9 (4) N = 10	5.6 (0.6) N = 10	5.8 (0.7) N = 10	21 (4) N = 6	21 (3) N = 10	17 (4) N = 6	12 (4) N = 5	0.197 (0.013) N = 6	0.137 (0.030) N = 5
Cedar Run Ungrazed										
CR1	0 (0) N = 9	33* (9) N = 9	8.1 (0.5) N = 9	7.7 (0.5) N = 9	18 (3) N = 9	21 (3) N = 9	20 (3) N = 9	10* (2) N = 8	0.248 (0.006) N = 9	0.168* (0.008) N = 8
CR4	10 (6) N = 11	4 (1) N = 10	6.3 (0.5) N = 11	6.1 (0.6) N = 10	22 (2) N = 10	17* (2) N = 10	11 (1) N = 10	9 (2) N = 5	0.159 (0.003) N = 10	0.087* (1.013) N = 5
Restored CR2	48 (8) N = 10	1* (1) N = 10	6.6 (0.4) N = 10	6.7 (0.4) N = 10	31 (4) N = 10	20* (3) N = 10	14 (2) N = 10	9 (3) N = 5	0.225 (0.013) N = 10	0.109* (0.025) N = 5

CR3	59 (9) N = 10	3* (3) N = 10	6.4 (0.4) N = 10	6.1 (0.5) N = 10	21 (2) N = 10	17 (1) N = 10	19 (2) N = 10	14 (2) N = 5	0.223 (0.011) N = 10	0.106* (0.010) N = 5
Slab Cabin Run Ungrazed										
SL2	12 (5) N = 12	8 (2) N = 10	4.3 (0.4) N = 12	4.2 (0.3) N = 10	21 (2) N = 12	19 (2) N = 10	11 (2) N = 12	2* (1) N = 5	0.085 (0.003) N = 11	0.016* (0.003) N = 5
SL4	0 (0) N = 9	0 (0) N = 8	2.4 (0.2) N = 9	3.1 (0.4) N = 8	11 (1) N = 8	12 (2) N = 8	17 (2) N = 8	14 (2) N = 4	0.042 (0.003) N = 8	0.039 (0.004) N = 4
Restored SL1	100 (0) N = 9	0* (0) N = 12	6.7 (0.6) N = 9	5.5 (0.8) N = 12	12 (2) N = 10	11 (2) N = 10	12 (2) N = 10	4* (3) N = 5	0.112 (0.009) N = 10	0.035 (0.035) N = 5
SL2.5	100 (0) N = 14	3* (2.5) N = 10	4.9 (0.2) N = 14	3.0 (0.3) N = 10	21 (2) N = 14	16 (2) N = 10	4 (0) N = 14	7* (1) N = 5	0.046 (0.004) N = 14	0.026* (0.001) N = 5
SL3	100 (0) N = 20	28* (10) N = 10	2.5 (0.5) N = 20	3.4* (0.5) N = 10	19 (2) N = 20	13* (2) N = 10	19 (9) N = 20	6 (2) N = 5	0.051 (0.003) N = 20	0.017* (0.002) N = 5

sand particles increased at the reference site in Spring Creek, although it was not significantly different from pre-restoration surveys. But, the proportion of cobble in Spring Creek was significantly lower than in pre-restoration surveys (Mann-Whitney, $p < 0.05$). Cedar Run sites, CR1 and CR4, and Slab Cabin Run site, SL4, had significantly increased proportions of silt-sand and significantly decreased proportions of gravel from pre-restoration surveys (Mann-Whitney, $p < 0.05$). Slab Cabin Run site, SL3, had significantly decreased sand-silt and significantly increased proportions of gravel and cobble (Mann-Whitney, $p < 0.05$). Generally, restoration neither decreased the proportion of silt-sand nor increased proportions of gravel and cobble in restored sites. Most sites that had been restored had at least as much or more silt-sand as prior to restoration.

Discharge

Stage and discharge were well correlated ($R^2 > 0.95$; Figure 4) in Spring Creek, Cedar Run, and Slab Cabin Run. From recorded stage values, mean daily discharge was estimated. Discharge was predicted from stage values that were within the range used in the rating curves.

During the entire study, discharge was highest in Spring Creek, intermediate in Cedar Run, and lowest in Slab Cabin Run (Table 6). Annual variations in discharge were largely due to variations in precipitation. Precipitation in 2001 was 77 cm or 16% less than during the pre-restoration period, 1991-1992, and discharge in all streams in 2001 was less than in 1991-1992. In 2002, precipitation was 108 cm or 17% higher than in 1991-1992, and discharge in Spring Creek and Cedar Run in 2002 was higher than in 1991-1992. Discharge in Slab Cabin Run in 2002 was less than 1991-1992, even though precipitation was greater in 2002 than in 1991-1992. Presumably, the ground water reserves in the Slab Cabin Run basin was so depleted

Table 5. Mean percent composition of stream substrates (SE in parentheses) in restored and ungrazed sites in Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1992) and post-restoration (2001) periods. Pairs of data that are significantly different (Mann-Whitney, $p < 0.05$) are noted by asterisks.

Stream, site type, and site number	Substrate (%)					
	Silt-sand		Gravel		Cobble	
	1992	2001	1992	2001	1992	2001
Spring Creek						
Ungrazed						
SP1	4 (3) N = 6	20 (9) N = 10	17 (5) N = 6	22 (6) N = 10	79 (7) N = 6	58* (10) N = 10
Cedar Run						
Ungrazed						
CR1	5 (2) N = 9	42* (10) N = 10	59 (10) N = 9	26* (6) N = 10	36 (10) N = 9	33 (9) N = 10
CR4	37 (9) N = 10	63* (8) N = 10	60 (7) N = 10	21* (5) N = 10	3 (2) N = 10	16 (6) N = 10
Restored CR2	53 (13) N = 10	57 (11) N = 10	9 (3) N = 10	11 (5) N = 10	40 (12) N = 10	32 911) N = 10
CR3	33 (10) N = 10	32 (7) N = 10	35 (7) N = 10	47 (6) N = 10	32 (9) N = 10	21 (6) N = 10
Slab Cabin Run						
Ungrazed						
SL2	82 (11) N = 10	93 (8) N = 10	17 (11) N = 10	5 (3) N = 10	1 (1) N = 12	3 (2) N = 10
SL4	6 (11) N = 8	46* (8) N = 8	77 (10) N = 8	24* (7) N = 8	17 (10) N = 8	30 (9) N = 8
Restored SL1	58 (10) N = 10	66 (7) N = 10	19 (6) N = 10	2* (1) N = 10	23 (5) N = 10	32 (7) N = 10

SL2.5	90 (5) N = 14	90 (5) N = 10	0 (0) N = 14	5 (5) N = 10	10 (5) N = 14	5 (2) N = 10
SL3	82 (8) N = 20	45* (11) N = 10	1 (10 N = 20	19* (6) N = 10	17 (8) N = 20	36* (9) N = 10

Figure 4. Discharge rating curves for Spring Creek, Cedar Run, and Slab Cabin Run. Curves are based on discharge (Q) and stage data (s) collected through 2002.

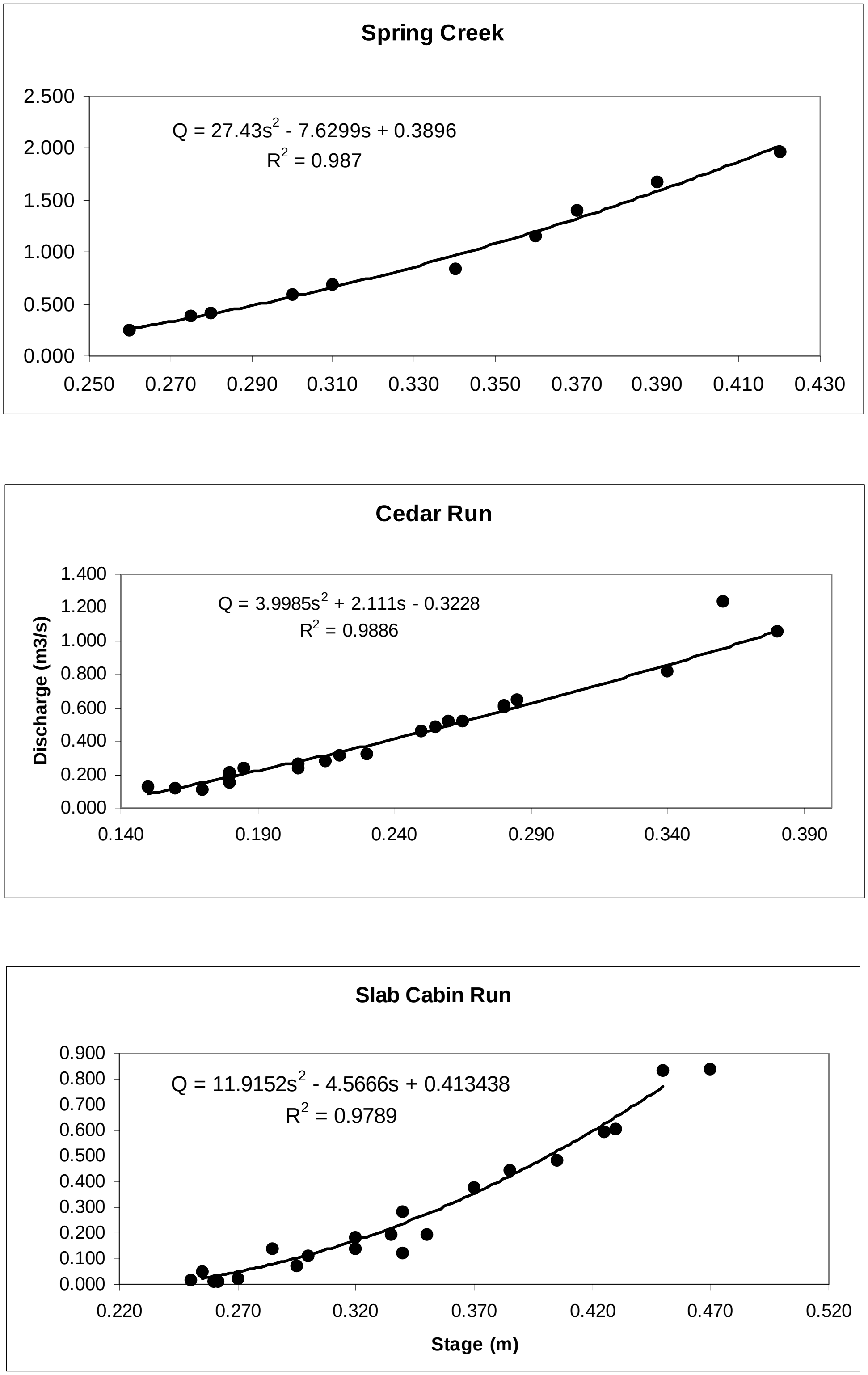

Table 6. Median, minimum, and maximum daily discharge ($m^3 \cdot s^{-1}$) in Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994) and post-restoration (2001-2002) study periods. No discharge data are available for Slab Cabin Run during 1993-1994.

	Spring Creek				Cedar Run				Slab Cabin Run		
	1991-1992[a]	1993-1994[b]	2001	2002	1991-1992[a]	1993-1994[b]	2001	2002	1991-1992[a]	2001	2002
Median	0.37	0.37	0.31	0.42	0.20	0.35	0.17	0.36	0.06	0.01	0.03
Minimum	0.16	0.19	0.19	0.19	0.11	0.28	0.10	0.12	0.03	0	0
Maximum	2.61	10.80	1.57	8.13	0.99	3.31	0.84	1.74	0.88	0.92	3.10

[a]1991-1992 data from Wohl and Carline (1996).
[b]1993-1994 data from Schnabel and Carline (1995).

from the below average precipitation in 2001 that the deficit was not made up in 2002. The severity of the 2001 drought was evident when permanent flow ceased at the gauging station on Slab Cabin Run on July 25, 2001 and did not resume until April 2, 2002. Precipitation in July and August 2002 was about 50% below normal, which resulted in cessation of flow on August 26, 2002, and permanent flow did not resume until November 21, 2002.

At sampling sites on Slab Cabin Run upstream of the gauging station at SL1, baseflow continued throughout the year, although water levels were low at those sections. Wohl (1993) identified the section in Slab Cabin Run between SL1 and the gauging station as losing stream flow.

Water Quality

Suspended Sediment

From turbidity and total suspended solids (TSS) measurements, polynomial relationships were developed between the variables for each stream. Turbidity was a good predictor of TSS in all streams ($R^2 > 0.90$; Figure 5). The relationships were used to estimate TSS for samples when only turbidity was analyzed.

At baseflow discharge, Spring Creek had the lowest TSS levels during both pre- and post-restoration periods (Table 7). Median values of TSS ranged from 1.9 to 4.0 mg/L. Cedar Run had intermediate levels of TSS during the pre-restoration period and during the post-restoration years, TSS declined by 36 to 45%. Prior to restoration, median TSS at baseflow in Slab Cabin Run was quite high, 29.3 mg/L, but after restoration median values had declined by 77 to 82%. These large differences in reductions in TSS in Cedar Run and Slab Cabin Run seemed to be at least partly related to differences in stream discharge during post-restoration years.

Figure 5. Relationships between turbidity (NTU) and total suspended solids (TSS) for Spring Creek, Cedar Run, and Slab Cabin Run. Data were collected in 2001.

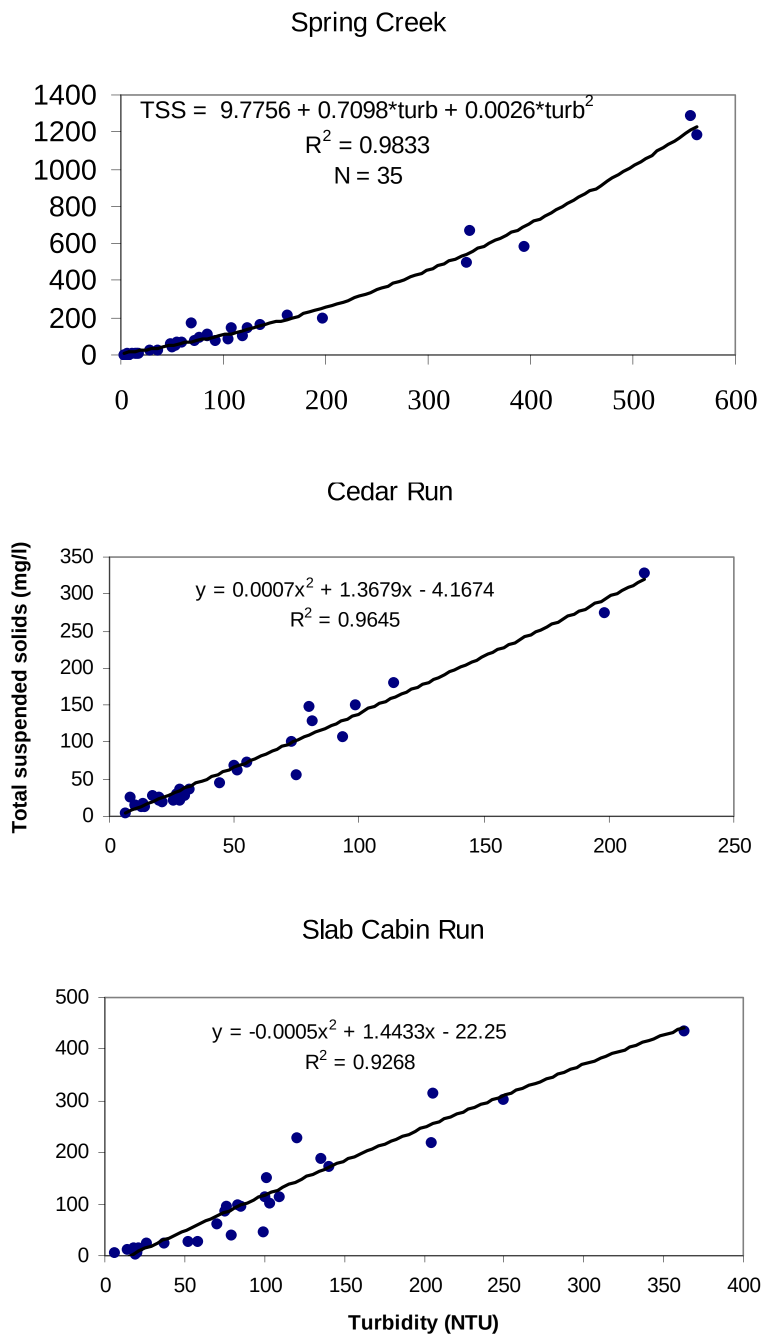

Table 7. Median sediment and nutrient concentrations (mg·L^{-1}) and interquartile ranges in baseflow samples form Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 for TSS; 1993-1994 for nutrients) and post-restoration (2001-2002) study periods. No nutrient data are available for Slab Cabin Run during 1993-1994.

	Spring Creek			Cedar Run			Slab Cabin Run		
	Pre-	2001	2002	Pre-	2001	2002	Pre-	2001	2002
TSS	4.0 (2.2-6.0) N = 61	1.9 (1.1-2.9) N = 48	2.7 (1.7-3.7) N = 50	17.75 (13.3-27.1) N = 62	9.8 (6.5-14.9) N = 48	11.4 (9.4-14.2) N = 48	29.3 (17.6-46.3) N = 52	6.6 (5.4-9.0) N = 28	5.4 (4.0-6.6) N = 27
Ortho-P	0.003 (0.003-0.004) N = 188	0.003 0.003-0.009 N = 48	0.003 (0.003-0.023) N = 50	0.003 (0.003-0.003) N = 181	0.004 (0.003-0.012) N = 48	0.005 (0.003-0.024) N = 50		0.041 (0.007-0.119) N = 28	0.009 (0.003-0.034) N = 27
Total P	0.100 (0.032-0.100) N = 178	0.024 (0.016-0.042) N = 48	0.031 (0.021-0.039) N = 50	0.100 (0.038-0.100) N = 168	0.034 (0.019-0.046) n = 48	0.037 (0.028-0.050) n = 50		0.104 (0.049-0.279) N = 28	0.046 (0.037-0.093) N = 27
Nitrate-N	2.40 (1.80-3.20) N = 190	2.40 (1.72-2.88) N = 48	1.68 (1.33-2.52) N = 50	4.45 (4.20-4.80) N = 182	4.34 (4.17-4.46) N = 48	4.31 (3.64-4.72) N = 50		2.44 (1.93-3.00) N = 28	2.43 (1.60-3.24) N = 27
Total-N	2.65 (1.90-3.30) N = 192	2.90 (2.33-3.80) N = 48	1.87 (1.52-2.92) N = 50	4.70 (4.50-4.90) N = 164	4.70 (4.38-5.29) N = 48	4.64 (4.27-4.93) N = 50		3.22 (2.77-4.55) N = 28	2.79 (1.80-4.15) N = 27

Below normal levels of precipitation in 2001 and part of 2002 resulted in little runoff and much reduced baseflow in Slab Cabin Run. The channel was dry at the gauging station from July 25, 2001 until April 2, 2002 and from August 26, 2002 to November 21, 2002. Even if riparian restoration reduced sediment loading to the stream, it would be difficult to judge the effect of restored riparian areas on sediment load by simply comparing median values of TSS.

We tested for changes in TSS in the restored streams by taking the difference in TSS between a restored stream and Spring Creek, the reference stream, for each day a water sample was collected during baseflow. We then compared the median difference during the pre-restoration period with the median differences during the two post-restoration periods. These analyses showed that differences in TSS during both post-restoration years were significantly ($P < 0.05$) less than during the pre-restoration period in both restored streams (Table 8). Differences in stream discharge among years do not affect this analysis, because flow was changing in a similar fashion at the same time in all streams. Therefore, we can conclude that there was a significant reduction in TSS at baseflow in both restored streams, despite variations in discharge among pre- and post-restoration periods.

We examined plots of discharge versus TSS at baseflow to determine if this relation changed from pre- to post-restoration periods (Appendix A). In Spring Creek, TSS increased with discharge though the slope was small. Results of ANOVA indicated a significant ($P = 0.016$) effect of flow, but no significant effect of year ($P = 0.16$) or of the flow*year interaction ($P = 0.40$). In Cedar Run there was a more pronounced increase in TSS with discharge (Appendix 1). Results of ANOVA indicated significant effects of flow ($P = 0.0003$), year ($P < 0.0001$), and of the flow*year interaction ($P = 0.0016$). A significant flow*year interaction indicates a change in slope of the relation between TSS and discharge (Figure 6). In Slab Cabin Run TSS at baseflow was not positively related to

Table 8. Median differences of baseflow sediment and nutrient concentrations ($mg \cdot L^{-1}$) between Spring Creek and Cedar Run and between Spring Creek and Slab Cabin Run and inter-quartile ranges during pre-restoration (1991-1992 and 1993-1994 for TSS; 1993-1994 for nutrient constituents) and post-restoration (2001-2002) study periods. Asterisks indicate that values for 2001 or 2002 are significantly different from pre-restoration values (Mann-Whitney, $p < 0.05$). No nutrient data are available for Slab Cabin Run during 1993-1994.

	Cedar Run - Spring Creek			Slab Cabin Run - Spring Creek		
	Pre-	2001	2002	Pre-	2001	2002
TSS	14.0 (11.0-20.8) N = 61	7.4 * (4.6, 11.8) N = 48	8.6* (6.6, 11.0) N = 48	25.25 (12.88, 42.13) N = 54	3.9* (2.4, 6.7) N = 28	1.533* (0.22, 3.7) N = 27
Ortho-P	0 (0.000, 0.000) N = 119	0 (0.000, 0.002) N = 48	0 (0.000, 0.0033) N = 50		0.033 (0.001-0.108) N = 28	0 (0.000, 0.016) N = 27
Total P	0 (0.000, 0.000) N = 107	0.001* (-0.0038, 0.0188) N = 48	0.008* (-0.0013, 0.0190) N = 50		0.074 (0.026, 0.247) N = 28	0.016 (0.007, 0.073) N = 27
Nitrate-N	2.00 (1.4, 2.6) N = 119	1.80 (1.42, 2.37) N = 48	2.13 (1.52, 2.71) N = 50		0.53 (0.13, 0.98) N = 28	0.69 (0.33, 1.21) N = 27
Total-N	1.91 (1.40, 2.40) N = 121	1.82 (1.31, 2.44) N = 48	2.43* (1.74, 3.16) N = 50		0.69 (0.13, 1.14) N = 28	0.84 (0.47, 1.33) N = 27

discharge, rather it was negatively related, owing to the large number of variable observations at low flow. The erratic flow in Slab Cabin Run during post-restoration years may have also contributed to this anomalous relation. As in Cedar Run, ANOVA indicated that year had a significant ($P = 0.01$) effect on TSS, such that at any given discharge, TSS in post-restoration years was less than during the pre-restoration period.

Changes in suspended sediment during storm flow was different among streams and years. In Spring Creek, TSS during storms in 1991-1992 was substantially lower than in either of the restored streams (Table 9). But, during post-restoration periods TSS was much higher than during the pre-restoration period. Even though sample size during the pre-restoration period was relatively small, we believe these differences are real, because of a housing construction project in the upper Spring Creek catchment.

Total suspended solids in Cedar Run during storm flow was three times higher than in Spring Creek and about one-half that in Slab Cabin Run in 1991-1992 (Table 9). During 2001, TSS in Cedar Run was lower than in the pre-restoration period, but in 2002 it was higher. When we compared median differences in TSS between Cedar Run and Spring Creek, differences were significant in 2001, but not in 2002, even though median TSS in Cedar Run was only 6 mg/L higher than in Spring Creek (Table 10). Hence, this analysis indicates that there was a significant reduction in TSS in Cedar Run during storm flow in one of the two post-restoration years.

In Slab Cabin Run, TSS during storm flow was substantially lower in 2001 and 2002 than during the pre-restoration period (Table 9). Median differences in TSS between Slab Cabin Run and Spring Creek in 2001 were different than in 1991-1992, but these differences were not significant in 2002 compared to 1991-1992 (Table 10). Thus, as we found in Cedar Run, TSS during storm flow in Slab Cabin Run was significantly less in 2001 but not in 2002.

Figure 6. Predicted relationship between TSS and discharge in Cedar Run during pre- and post-restoration periods. Equations were derived from a fixed effects general linear model with a period*flow interaction term. $F = 17.1$, $P < 0.0001$, $R^2 = 0.22$.

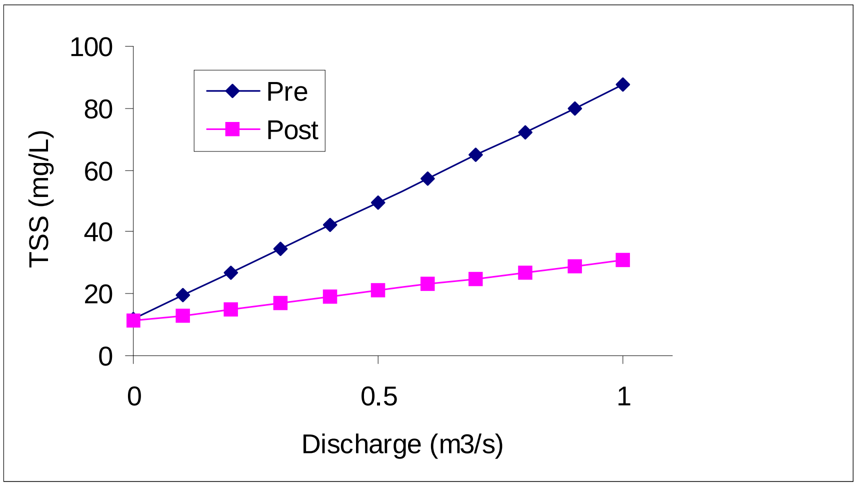

Table 9. Median sediment and nutrient concentrations ($mg \cdot L^{-1}$) and inter-quartile ranges in stormflow samples from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration study periods (1991-1992 for TSS; 1993-1994 for nutrients) and post-restoration (2001-2002) study periods. No nutrient data are available for Slab Cabin Run during 1993-1994. *2002 data includes four 2003 storms* .

	Spring Creek			Cedar Run			Slab Cabin Run		
	Pre-	2001	2002	Pre-	2001	2002	Pre-	2001	2002
TSS	7.5 (6.2, 9.1) N = 9	20 (9.1, 42.0) N = 21	26 (18.8, 61.4) N = 23	29.4 (20.7, 45.9) N = 8	20.6 (15.1, 34.3) N = 19	33.1 (21.2, 53.9) N = 24	62.1 (28.2, 86.2) N = 9	18.2 (10.5, 46.7) N = 12	16.9 (10.8, 51.0) N = 22
Ortho-P	0.005 (0.005, 0.008) N = 51	0.004 (0.003, 0.015) N = 15	0.021 (0.003, 0.045) N = 21	0.005 (0.005, 0.008) N = 49	0.003 (0.003, 0.006) N = 15	0.006 (0.003, 0.043) N = 23		0.011 (0.001, 0.132) N = 8	0.035 (0.003, 0.099) N = 20
Total P	0.082 (0.050, 0.100) N = 40	0.069 (0.044, 0.137) N = 15	0.078 (0.049, 0.176) N = 21	0.050 (0.013, 0.100) N = 45	0.066 (0.022, 0.079) N = 15	0.071 (0.041, 0.154) N = 23		0.187 (0.071, 0.273) N = 9	0.107 (0.062, 0.232) N = 20
Nitrate-N	1.60 (1.40, 2.20) N = 52	1.78 (1.12, 2.06) N = 15	1.14 (0.81, 1.47) N = 21	4.20 (3.95, 4.40) N = 50	3.90 (3.63, 3.97) N = 15	3.57 (3.17, 4.13) N = 23		1.48 (1.09, 2.26) N = 9	1.58 (1.07, 2.17) N = 20
Total-N	1.90 (1.52, 2.48) N = 52	2.37 (1.78, 3.17) N = 15	1.57 (1.17, 2.16) N = 21	4.35 (4.15, 4.64) N = 50	4.25 (4.11, 6.31) N = 15	3.88 (3.68, 4.29) N = 23		2.76 (1.98, 4.80) N = 8	2.21 (1.55, 2.53) N = 20

Total annual sediment yields, which integrate baseflow and storm flow TSS plus stream discharge, indicate that a positive effect of riparian restoration in the Cedar Run basin, but low flows in Slab Cabin Run complicate interpretations of the data. During the pre-restorations years, annual sediment loads in Spring Creek were less than one-half of those in Cedar Run and Slab Cabin Run (Table 11). In Cedar Run sediment load in 2001 was only 18% higher than in Spring Creek, and in 2002 sediment load in Cedar Run was 46% less than in Spring Creek. In Slab Cabin Run sediment load in 2001 was about one-half of that in Spring Creek, but this was largely due to extended periods of no flow in Slab Cabin Run. Interestingly, in 2002 sediment load in Slab Cabin Run was 135 tonnes compared to 794 tonnes in Spring Creek. Here again, there were three months of no flow in Slab Cabin Run during late summer and autumn 2002, which no doubt contributed to the reduced sediment load. Land-use disturbances in the Spring Creek basin probably influenced the increased sediment load.

Phosphorus

In baseflow and stormflow samples, ortho-phosphate and total phosphorus concentrations were extremely low in all streams before and after restoration (Tables 7 and 9). In fact, in more than 85% of pre-restoration samples ortho-phosphate concentrations were below detection limit (0.005 $mg \cdot L^{-1}$) in Spring Creek and Cedar Run and in post-restoration samples more than 47% of samples from both streams were below the detection limit. Likewise, total phosphorus concentrations were below detection limit (0.05 $mg \cdot L^{-1}$) in more than 77% of pre-restoration samples in Spring Creek and Cedar Run, but were measured in all post-restoration samples. In instances where concentrations were below detection limits, sample concentrations were estimated as one-half the detection limit.

Table 10. Median differences of stormflow sediment and nutrient concentrations (mg·L^{-1}) between Spring Creek and Cedar Run and Spring Creek and Slab Cabin Run and inter-quartile ranges during pre-restoration study (1991-1992 for TSS; 1993-1994 for nutrients constituents) and post-restoration (2001-2002) study periods. Asterisks indicate that values for 2001 and 2002 are significantly different from those for the pre-restoration period (Mann-Whitney, $p < 0.05$). No nutrient data are available for Slab Cabin Run during 1993-1994. The data for 2002 includes four storms sampled in early 2003.

	Cedar Run Values - Spring Creek Values			Slab Cabin Run Values - Spring Creek Values		
	Pre-	2001	2002	Pre-	2001	2002
TSS	20.6 (14.7-37.5) N = 8	-3.8 * (-27.0. 10.8) N = 18	6.1 (-2.8, 33.5) N = 21	55.7 (19.9, 78.4) N = 9	1.9 * (-29.4, 13.0) N = 10	17.9 (11.2, 41.5) N = 20
Ortho-P	0 (0.000, 0.001) N = 33	0 (-0.013, 0.000) N = 14	0 (-0.005, 0.004) N 20		-0.011 (-0.128, 0.000) N = 7	-0.003 (-0.036, 0.000) N = 19
Total P	0.000 (0.000, 0.015) N = 26	-0.018 * (-0.062, 0.003) N = 14	0.000 (-0.031, 0.057) N = 20		-0.122 (-0.174, -0.046) N = 8	-0.007 (-0029, 0.043) N = 19
Nitrate-N	2.4 (2.050, 2.675) N = 36	1.976 (2.050, 2.675) N = 14	2.4 (1.928, 2.770) N = 20		2.586 (1.650, 2.915) N = 8	2.048 (1.767, 2.352) N = 19
Total-N	2.52 (2.270, 3.300) N = 30	1.97 (2.270, 3.300) N = 14	2.251 (2.012, 2.535) N = 20		1.972 (0.672, 2.365) N = 7	1.909 (1.394, 2.193) N = 19

Table 11. Total annual yields (tonnes) of suspended solids and nitrate-N in Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994 for suspended solids; 1993-1994 for nitrate-N) and post-restoration (2001) study periods. No nitrate-N data are available for Spring Creek and Cedar Run during 1991-1992 and for Slab Cabin Run during 1993-1994.

	Spring Creek				Cedar Run				Slab Cabin Run		
	1991-1992	1993-1994	2001	2002	1991-1992	1993-1994	2001	2002	1991-1992	2001	2002
Suspended Solids	113	464	117	794	255	536	138	431	273	50	135
Nitrate-N		49.2	23.5	28.5		64.1	31.1	50.7		4.98	10.1

Among streams, median ortho-phosphate and total phosphorus concentrations were highest in Slab Cabin Run in post-restoration baseflow, especially at low flows (Tables 7). Concentrations of ortho-phosphate and total phosphorus were similar between Spring Creek and Cedar Run even prior to restoration. Median differences between Spring Creek and Cedar Run baseflow concentrations of ortho-phosphate and total phosphorus were zero for pre-restoration periods and near zero for post-restoration periods (Table 8). Median differences in total phosphorus between Cedar Run and Spring Creek in 2001 and 2002 were quite small, but significant based on Mann-Whitney tests. But, the power of this test may influenced by the large number of ties in ranks of the data, owing to many zeros or one-half detection limits. Concentrations of ortho-phosphate and total phosphorus at baseflow in Cedar Run and Spring Creek have no obvious relationship to discharge (Appendix B and Appendix C). In Slab Cabin Run, baseflow concentrations of ortho-phosphate and total phosphorus increase as discharge approaches zero (Appendix B and Appendix C). This anomalous result may have been caused by periodic flushing of isolated pools of water, which were common when flow ceased at the gauging station.

During storm flow, concentrations of ortho-phosphate and total phosphorus in pre- and post-restoration samples were similar in Spring Creek and Cedar Run (Table 9). The only apparent difference occurred in 2002 storm flow when median concentration of ortho-phosphate in Spring Creek was about three times higher than in Cedar Run. Median differences between the Spring Creek and Cedar Run stormflow concentrations of ortho-phosphate and total phosphorus were zero, except in 2001, when differences were negative, i.e., concentrations were less in Cedar Run than in Spring Creek (Table 10). Overall, there was no indication that phosphorus concentrations in Cedar Run changed after restoration.

Nitrogen

Restoration had no effect on nitrate-nitrogen and total nitrogen concentrations in Cedar Run. In pre- and post-restoration samples, Cedar Run had the highest concentrations of nitrate-nitrogen and total nitrogen at baseflow (Table 7). There was little difference in median nitrogen concentrations between pre- and post-restoration periods. Median differences between Spring Creek and Cedar Run in baseflow concentrations of nitrate-nitrogen did not change significantly after restoration (Table 8). The only significant difference between pre-restoration and post-restoration study periods was in 2002 when differences in concentration of total nitrogen was higher than during the pre-restoration period. During the pre- and post-restoration periods, nitrate-nitrogen and total nitrogen concentrations did not have any clear relationship to stream discharge at baseflow in study streams (Appendix D and Appendix E). Any changes in concentrations are not apparently related to changes in discharge.

Similar to baseflow samples, concentrations of nitrate-nitrogen and total nitrogen in stormflow samples were highest in Cedar Run during the pre-restoration study period (Table 9). Spring Creek had the lowest concentrations of total nitrogen and intermediate concentrations of nitrate-nitrogen, while Slab Cabin Run had the lowest concentrations of nitrate-nitrogen and intermediate concentrations of total nitrogen. Median concentrations of nitrate-nitrogen and total nitrogen decreased modestly in Cedar Run stormflow samples after restoration. Although median differences between Cedar Run and Spring Creek stormflow sample concentrations of nitrate nitrogen and total nitrogen declined after restoration, there were no significant differences between pre-restoration and post-restoration periods (Table 10).

During the post-restoration study period, Cedar Run had the highest annual yield of nitrate-nitrogen, while the nitrate-nitrogen yield was the lowest in Slab Cabin Run (Table 11). Among year differences in nitrate yields seemed to be partly related to differences in discharge, which was most evident in Cedar Run and Slab Cabin Run. In Cedar Run the lowest nitrate yield occurred in 2001 when discharge (0.17 $m^3 \cdot s$) was considerably lower than in 1993-1994 (0.35 $m^3 \cdot s$) and 2002 (0.36 $m^3 \cdot s$). But, nitrate yield in 2002 was 21% less than in 1993-1994, when discharges were nearly the same - a possible response to restoration. Nitrate yields in Slab Cabin Run in 2001 and 2002 were quite low, as were discharges; hence, it is impossible to relate these low yields to restoration.

Substrate Composition

Substrate composition, as described by percent fines and the Fredle index, at the Spring Creek station changed little from 1992 to the post-restoration period, 2001-2002 (Table 12). In contrast, in Cedar Run percent fines declined significantly at all sampling sites in 2001 and at three of four sites in 2002. The Fredle index increased at all stations, but differences were significant in only two of eight comparisons. In Slab Cabin Run, there were no obvious trends in measures of substrate composition. At station SL3 there was a significant increase in percent fines from 1992 to 2001, but none of the other comparisons was significant.

Stream Temperatures

Temperatures in Spring Creek tended to be cooler in summer and warmer in winter than in the other two streams, because large inputs of groundwater entered the stream within 500 m of the gauging station (Table 13). Cedar Run temperatures were intermediate, and those in Slab Cabin Run

Table 12. Median Fredle index, median percent fines (particles less than 1 mm), and inter-quartile ranges for Spring Creek, Cedar Run, and Spring Creek sites during pre-restoration (1992) and post-restoration (2001 and 2002) study periods. Asterisks indicate that 2001 or 2002 values are significantly different from 1992 values for all sites derived from four samples from each sampling site (Mann-Whitney, N = 4, p < 0.05).

Stream and sampling site	Percent fines			Fredle Index		
	1992	2001	2002	1992	2001	2002
Spring Creek						
SP1	5.6 (4.3, 6.9)	8.2 (5.4, 14.0)	7.0 (2.4, 9.1)	4.62 (4.5, 6.3)	4.15 (3.4, 7.4)	5.4 (3.5, 6.3)
Cedar Run						
CR1	18.2* (14.0, 23.5)	11.7* (10.0, 13.7)	14.3 (6.6, 20.7)	1.62 (1.24, 2.34)	2.11 (1.80, 2.36)	2.5 (0.8, 12.6)
CR2	28.9* (22.9, 37.0)	6.9* (5.5, 17.7)	16.9* (5.8, 19.4)	0.85 (0.74, 1.31)	3.8 (1.86, 4.91)	2.2 (0.6, 6.1)
CR3	33.5* (28.8, 37.8)	23.3* (18.6, 26.4)	15.7* (6.5, 22.8)	0.8* (0.67, 1.01)	1.36* (1.12. 1.96)	2.5 (0.9, 5.4)
CR4	20.3* (18.9, 25.9)	15.6* (10.0, 16.9)	11.9* (10.6, 15.9)	1.47 (1.03, 2.03)	2.21 (1.92, 2.98)	2.6 (1.0, 3.5)
All sites	26.0* (19.9, 32.3)	14.4* (8.8, 20.3)	14.1* (10.6, 18.6)	1.13* (0.82, 1.59)	2.11* (1.59, 3.03)	2.5 (0.9, 3.6)
Slab Cabin Run						

SL1	18.6 (13.0, 26.4)			1.00 (0.72, 1.80)		
SL2	34.6 (31.4, 37.5)	35.4 (26.8, 48.4)	26.4 (20.0, 41.2)	0.77 (0.57, 1.0)	0.6 (0.29, 1.08)	1.2 (0.4, 1.7)
SL3	35.7* (24.8, 38.5)	51.9* (43.0, 65.0)	33.2 (23.8, 49.8)	0.83 (0.67, 1.10)	0.43 (0.24, 0.73)	0.96 (0.6, 1.60)
SL3.5		22.3 (10.0, 25.0)	13.9 (7.8, 23.3)		1.55 (1.08, 4.90)	3.43 (1.6, 6.30)
SL4	18.6 (13.0, 26.4)	16 (8.5, 28.3)	11.6 (2.4, 20.2)	1.53 (1.13, 2.25)	2.25 (1.29, 4.37)	2.88 (1.9, 8.2)
All sites	26.8 (17.4, 35.1)	27.8 (19.8, 46.4)	20.8 (10.9, 28.6)	0.86 (0.72, 1.46)	1.19 (0.40, 1.76)	1.8 (0.9, 3.4)

Table 13. Mean daily temperatures by month (˚C) for Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (September 1, August 31, 1991-1992) and post restoration (January to December 2001-2002) periods.

	Spring Creek			Cedar Run			Slab Cabin Run		
	1991-1992	2001	2002	1991-1992	2001	2002	1991-1992	2001	2002
Jan	7.2	8.3	8.9	4.2	4.2	5.1	1.6		
Feb	7.4	7.9	8.4	4.8	6.3	6.5	2.4	3.3	
Mar	6.7	7.5	8.7	6.9	7.1	8.1	5.2	4.8	
Apr	8.9	10.0	10.7	10.1	11.3	11.8	9.4	10.3	11.7
May	10.6	11.7	11.9	12.3	13.4	13.0	12.8	18.4	13.1
Jun	11.5	12.6	13.8	14.4	16.2	14.8	16.2	20.1	15.1
Jul	12.0	12.8	13.3	16.3	17.0	15.9	17.9		18.2
Aug	11.5	12.9	13.0	15.9	18.7	16.6	16.6		21.0
Sept	11.0	12.4	12.2	14.5	15.8	15.7	17.5		
Oct	10.2	11.6	11.5	9.4	11.8	11.4	9.6		
Nov	9.1	11.2	9.6	7.2	8.8	8.6	6.3		
Dec	6.8	9.9	7.6	5.0	6.4	6.3	3 3.0		

were the most extreme, owing to the least amount of groundwater influence.

During post-restoration years, stream temperatures tended to be warmer in both winter and summer months in all streams. When we compared differences in temperature between each restored stream and Spring Creek, we found some significant differences, but there were no consistent trends (Table 14). If restoration benefitted streams, we would have expected stream temperatures to more closely simulate those in Spring Creek. Hence, we concluded that riparian restoration did not significantly influence stream temperatures at the gauging stations.

Macroinvertebrate Communities

Macroinvertebrate community composition was variable in all study streams; some proportions of taxa increased greatly since restoration while others declined. Two to three taxa typically dominated stream communities, making up more than one-half the individuals collected. In Spring Creek, Amphipoda and Diptera composed a large proportion of taxa in pre- and post-restoration periods (Tables 15 and 16). Other taxa, like Coleoptera, Ephemeroptera, and Trichoptera, were consistently present in Spring Creek throughout all sampling periods. Absent from pre-restoration samples, the stonefly, Nemouridae *Amphinemura*, was found in 2001, but not 2002. In Cedar Run, the most common taxa were Amphipoda, Isopoda, and Diptera during all sampling periods. *Amphinemura* was also found after restoration in Cedar Run in 2001 and 2002.

The macroinvertebrate community in Slab Cabin Run was mostly made up of Isopoda, Coleoptera, and Diptera. Amphipoda were not present in large numbers. Although Diptera comprised a large proportion of the macroinvertebrates at all sampling periods, the relative abundance of Dipterans decreased during both seasons after restoration. Taxa categorized as “Other”, including Gastropoda, Decapoda, Hirudinea, and Hydracarina, made up 15% of individuals

Table 14. Median and interquartile ranges (in parentheses) of daily differences in temperatures by month (˚C) for Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (September 1, 1991-August 31, 1992) and post-restoration (January to December 2001-2002) periods. Asterisks indicate that values for 2001 or 2002 were significantly different from those in 1991-1992 (Mann-Whitney test, $p < 0.05$).

	Cedar Run - Spring Creek			Slab Cabin Run - Spring Creek		
	1991-1992	2001	2002	1991-1992	2001	2002
Jan	-2.9 (-2.0, -4.0)	-3.3* (-1.9, -4.6)	-0.8* (-0.1, -2.9)	-5.8 (-4.6, -6.8)		
Feb	-2.4 (-1.6, -3.7)	-1.6* (-1.0, -2.3)	-1.7* (-1.1, -2.4)	-5.6 (-3.5, -6.5)	-3.7 (-3.2, -5.3)	
Mar	0.0 (-0.8, 1.4)	0.3 (-1.8, 1.1)	-0.2 (-2.5, 1.2)	-1.1 (-0.2, -2.8)	-2.1* (-1.3, -4.3)	
Apr	1.2 (0.8, 1.7)	1.3 (1.0, 1.7)	1.0 (0.4, 1.4)	0.6 (0.1, 1.2)	0.4 (-0.5, 0.9)	0.8 (-0.1, 1.9)
May	1.6 (1.0, 2.5)	1.7 (1.3, 2.4)	1.1* (0.9, 1.5)	2.0 (1.3, 3.3)	7.2* (3.2, 9.5)	0.9* (0.7, 1.8)
Jun	3.1 (2.3, 3.6)	3.6 (2.9, 4.7)	1.1* (O.6, 1.6)	4.6 (3.9, 5.9)	7.6* (6.2, 9.6)	1.5* (0.3, 2.1)
Jul	4.2 (3.5, 5.0)	4.0 (3.5, 5.0)	2.6* (2.0, 3.1)	6.1 (5.4, 6.4)		4.6* (3.7, 5.8)
Aug	4.4 (3.4, 4.9)	5.7* (5.0, 6.6)	3.7* (3.1, 4.2)	5.1 (4.0, 6.0)		8.1* (6.7, 8.6)
Sep	3.6 (2.7, 4.9)	3.7 (2.1, 4.3)	3.8 (2.7, 4.2)	6.9 (4.8, 8.6)		
Oct	-0.8 (-2.2, 0.6)	0.3 (-2.0, 2.4)	-0.8 (-1.3, 1.0)	-0.7 (-2.6, 1.0)		
Nov	-1.9 (-3.3, 0.3)	-2.3 (-3.6, -1.1)	-1.1* (-1.4, -0.6)	-3.1 (-4.4, 0.9)		
Dec	-1.9 (-3.1, 0.1)	-3.3* (-4.6, -1.9)	-0.8 (-2.9, -0.1)	-3.5 (-4.8, -2.7)		

Table 15. Relative abundance (percent of total individuals collected) of macroinvertebrate taxa in May from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1992) and post-restoration (2001 and 2002) study periods. Three samples were collected annually in Spring Creek and twelve samples total were collected annually in Cedar Run and Slab Cabin Run during pre- and post-restoration periods.

	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

Table 16. Relative abundance (percent of total individuals collected) of macroinvertebrate taxa in August from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1992) and post-restoration (2000 and 2001) periods. Three samples were collected in Spring Creek and twelve samples total were collected in Cedar Run and Slab Cabin Run during pre-restoration and post-restoration periods.

	Spring Creek			Cedar Run			Slab Cabin Run		
	1992	2001	2002	1992	2001	2002	1992	2001	2002
Amphipoda	56	35	27	17	8	16	14	5	8
Isopoda	0	< 1	0	22	40	44	2	56	60
Coleoptera	5	4	9	11	19	14	1	5	3
Diptera	29	47	19	35	20	10	47	18	12
Ephemeroptera	3	5	4	1	3	< 1	2	5	< 1
Trichoptera	5	4	5	11	3	5	15	4	5
Plecoptera	0	< 1	0	0	< 1	< 1	0	0	0
Oligochaeta	0	< 1	< 1	0	1	1	3	3	4
Turbellaria	0	2	27	1	3	3	4	3	2
Other	2	1	9	2	2	7	12	1	5

in spring 2001 samples. Plecopterans, which were found at two sites in Slab Cabin Run prior to restoration, were not present after restoration. Ephemeroptera and Trichoptera were present in small proportions in Slab Cabin Run in all sampling periods, but did not increase greatly after restoration in either stream.

The number of macroinvertebrate genera in Spring Creek varied during post-restoration periods, increased slightly in Cedar Run, and varied in Slab Cabin Run (Table 17). Ephemeroptera, Trichoptera, and Plecoptera (EPT) taxa richness followed these same trends (Table 18). The Shannon diversity index failed to show any trends from 1992 compared to samples collected during the post-restoration period (Table 19). Thus, we conclude that riparian restoration in Cedar Run and Slab Cabin Run had no demonstrable influence on the composition of macroinvertebrate communities.

Densities of macroinvertebrates in May and August samples from Spring Creek increased somewhat from 1992 to the first year of the post-restoration period, but then decreased during the second year of the post-restoration period (Table 20). Changes in macroinvertebrate densities in Cedar Run varied greatly among sampling stations in the May samples, and no trends were evident. In August samples there was a consistent and large increase in macroinvertebrate densities from 1992 to the post-restoration periods. In Slab Cabin Run, densities in May did not show any trend through time. But, like August samples in Cedar Run, there were marked increases in macroinvertebrate densities from 1992 to post-restoration periods.

We computed ratios of macroinvertebrate densities in Spring Creek to densities at each station in treated streams and compared these ratios between pre- and post-restoration periods. The median ratio of invertebrates densities in Cedar Run during pre-restoration was 2.55, i.e., there were 2.55 times more invertebrates in Spring Creek than in the four stations in Cedar Run. This ratio fell to

Table 17. Median number and ranges of macroinvertebrate taxa from Spring Creek, Cedar Run, and Slab Cabin Run sites during pre-restoration (1992) and post-restoration (2000 and 2002) periods. Three samples were collected at each site during pre- and post-restoration periods.

Stream and sampling site	May			August		
	1992	2001	2002	1992	2000	2001
Spring Creek						
SP1	21 (18-23)	24 (18-28)	18 (12-19)	16 (14-19)	18 (16-22)	19 (18-20)
Cedar Run						
CR1	13 (8-14)	24 (23-26)	17 (14-20)	11 (9-12)	20 (20-27)	19 16-23
CR2	12 (11-19)	25 (18-25)	15 (12-17)	14 (12-17)	15 (12-19)	17 (17-20)
CR3	18 (12-19)	17 (17-21)	18 (16-18)	15 (13-17)	12 (10-16)	21 (17-21)
CR4	13 (10-20)	14 (12-19)	13 (6-14)	14 (12-14)	16 (14-19)	13 (12-15)
Slab Cabin Run						
SL1	9 (9-12)			10 (10-15)		

SL2	13 (6-13)	9 (8-24)	7 (6-13)	13 (11-15)	11 (8-13)	13 (7-18)
SL3	10 (10-18)	10 (9-10)	(2-5)	14 (13-16)	17 (13-18)	13 (11-19)
SL3.5		20 (19-20)	14 (14-14)		15 (11-15)	9 (8-11)
SL4	13 (10-18)	20 (18-21)	16 (12-16)	14 (10-16)	15 (13-20)	18 (18-20)

Table 18. Median number of EPT macroinvertebrate taxa and ranges from Spring Creek, Cedar Run, and Slab Cabin Run sites during pre-restoration (1992) and post-restoration (2000 and 2001) periods. Three samples were collected at each site during pre-restoration and post-restoration periods.

Stream and sampling site	May			August		
	1992	2001	2002	1992	2000	2001
Spring Creek						
SP1	9 (8-11)	12 (9-13)	7 (3-8)	5 (4-6)	8 (6-9)	6 (5-7)
Cedar Run						
CR1	7 (7-9)	7 (7-9)	8 (6-9)	3 (2-4)	8 (6-12)	8 (4-10)
CR2	4 (3-7)	8 (7-12)	5 (5-5)	4 (3-5)	5 (2-7)	5 (5-8)
CR3	7 (5-11)	7 (7-8)	7 (6-7)	5 (4-6)	1 (1-3)	5 (4-6)
CR4	5 (2-9)	4 (4-5)	2 (2-3)	4 (3-5)	4 (2-5)	2 (1-3)
Slab Cabin Run						
SL1	2 (2-4)			3 (3-6)		
SL2	2 (1-4)	2 (1-4)	0 (0-0)	2 (2-4)	0 (0-1)	2 (1-4)

SL3	4	1		7	5	3
	(2-6)	(0-2)	(0-1)	(6-7)	(4-6)	(3-3)
SL3.5		5	5		3	1
		(4-5)	(2-5)		(2-7)	(0-3)
SL4	5	5	4	2	3	5
	(4-6)	(4-7)	(4-5)	(1-3)	(3-8)	(5-6)

Table 19. Median Shannon diversity index and ranges for macroinvertebrates from Spring Creek, Cedar Run, and Slab Cabin Run sites during pre-restoration (1992) and post-restoration (2000 and 2001) periods. Three samples were collected at each site during pre- and post-restoration periods.

Stream and sampling site	May			August		
	1992	2001	2002	1992	2000	2001
Spring Creek						
SP1	1.20 (1.17-1.50)	1.78 (1.59-1.91)	2.06 (1.64-2.14)	1.41 (1.24-1.46)	1.20 (1.00-1.55)	1.95 (1.72-2.11)
Cedar Run						
CR1	1.65 (1.34-2.05)	2.10 (2.04-2.20)	2.07 (1.86-2.16)	1.60 (1.40-1.88)	2.31 (1.97-2.36)	1.47 (1.15-1.57)
CR2	1.92 (1.87-1.96)	1.93 (1.88-2.04)	1.55 (1.53-1.99)	1.69 (1.49-1.80)	1.55 (1.47-1.66)	1.80 (0.72-1.94)
CR3	2.00 (1.91-2.15)	2.03 (1.97-2.17)	2.16 (2.07-2.23)	1.84 (1.51-1.85)	1.25 (1.16-1.57)	2.08 (1.99-2.10)
CR4	1.47 (1.28-1.69)	1.85 (1.73-1.87)	1.18 (1.14-1.83)	1.37 (1.26-1.48)	1.68 (1.38-1.77)	1.13 (1.10-1.39)
Slab Cabin Run						
SL1	0.91 (0.90-1.19)			1.71 (1.57-1.92)		
SL2	1.37 (0.98-1.78)	0.96 (0.77-1.35)	0.96 (0. 94-1.08)	1.60 (1.53-1.66)	1.17 (1.03-1.30)	1.08 (1.08-1.32)

SL3	1.27 (1.10-1.46)	0.78 (0.56-0.81)	(0.18-0.84)	1.70 (1.41-1.73)	1.51 (1.04-1.67)	1.31 (1.28-1.63)
SL3.5		1.22 (0.74-1.47)	1.53 (1.38-1.60)		0.94 (0.68-1.13)	0.94 (0.06-0.96)
SL4	1.60 (1.52-1.70)	1.72 (1 .48-1.81)	1.53 (1.44-1.79)	1.63 (1.33-1.66)	1.70 (1.65-1.87)	1.44 (1.34-1.76)

Table 20. Median macroinvertebrate density (number per m^2) and ranges from Spring Creek, Cedar Run, and Slab Cabin Run sites during pre-restoration (1992) and post-restoration (2000-2002) periods. Three samples were collected at each site during pre-restoration and post-restoration periods.

Stream and sampling Site	May			August		
	1992	2001	2002	1992	2000	2001
Spring Creek						
SP1	12,980 (9,171- 29,430)	14,472 (4,347-69,079)	3,767 (1,410-6,749)	15,170 (8,558-19,300)	17,481 (14,176-19,773)	7,050 (6,275-23,573)
Cedar Run						
CR1	1630 (344-2,940)	16,355 (11,083-22,790)	7,459 (7,072-9,666)	2,790 (8,558-19,300)	24,822 (20,118-36447)	18,869 (8, 633-23,261)
CR2	4,540 (1,620-9,930)	47,204 (15,667-75,116)	5,167 (5,102-5,188)	6,660 (3,780-15,900)	29,063 (25,704-29902)	18,288 (17,556-20,785)
CR3	4,350 (2,540-4,630)	4,487 (1,851-5,552)	3,111 (3,111-7,685)	7,450 (6,510-9,850)	16,322 (6,268-30,934)	11, 011 (6,415-52,915)
CR4	13,580 (6,000-14,700)	3,949 (3,798-6,693)	4,790 (506-47,544)	6,560 (5,940-10,000)	16,501 (5,102-25,844)	28,148 (6,706-60,095)
Slab Cabin Run						
SL1	7,480 (2,160-8,570)			2,920 (1,310-4,080)		

SL2	3,840 (3,150-4,970)	8,382 (1,636-32,764)	1,421 (743-9,881)	3,310 (1,490-3,300)	5,210 (3,757-9,978)	6,394 (1,184-9,429)
SL3	4,710 (3,080-7,960)	4,724 (4,595-5,036)	2,928 (1,033-4,822)	7,590 (7,320-10,250)	10,915 (10,616-14,951)	12,475 (8,105-29,891)
SL3.5		40,856 (23,780-45,859)	4,994 (4,166-7,319)		20,042 (18,772-23,831)	16,232 (6,361-44,444)
SL4	2,470 (860-4,060)	26,534 (14,989-33,334)	4,489 (1,356-7,083)	7,230 (2,400-8,364)	10,624 (3,746-13,057)	22,174 (18,858-56,532)

0.70 during post-restoration, and the ratios were significantly different (Kruskal-Wallis test, P = 0.004). We reran the test and omitted data from CR4, which was upstream of all restoration work in Cedar Run. Here again, ratios, 3.85 vs. 0.70, were significantly different (P = 0.006). We observed similar changes in Slab Cabin Run; ratios declined significantly (P = 0.002) 3.85 to 1.10. These results indicate that macroinvertebrate densities in both treatment streams increased after restoration.

Fish Communities and Brown Trout Densities

The number of fish species present in study streams was relatively low during pre-restoration and post-restoration surveys (Table 21). Slab Cabin Run had the highest number of fish species; aside from brown trout, several cyprinid species, tessellated darter, banded killifish, slimy sculpin, and white sucker were present before and after restoration. Some species collected in pre-restoration surveys, like creek chub, fallfish, and pearl dace, were absent after restoration, though they did not occur in large numbers in 1992 surveys. A small number of bluegills were found in Slab Cabin Run after restoration; they had not been previously collected there. In Cedar Run, species composition remained unchanged over time; brown trout, slimy sculpins, and white suckers were all collected before and after restoration. Likewise, the community fish in Spring Creek was similar between sampling periods, consisting of brown trout, slimy sculpins, and white suckers.

During May fish surveys, densities of age-1 and older brown trout in Spring Creek declined by about 30% from pre-restoration to post-restoration periods, while at all Cedar Run sites, the post-restoration densities of age-1+ brown trout increased from pre-restoration densities (Table 22). Among all stations in Cedar Run, brown trout densities in 2001 were 103% higher than in 1992 and in 2002 they were 46% higher than in 1992. In contrast, there was no evidence that brown trout densities in Slab Cabin Run changed after restoration. No brown trout were captured in study site

Table 21. Mean number of fish per 50 m in all sites from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1992) and post-restoration (2000-2002) study periods. One site was sampled on Spring Creek and four sites were sampled on Cedar Run and Slab Cabin Run. Sites were sampled in May 1992, 2001, 2002 and August 1992, 2000, and 2001.

	1992			2000-2002		
	Spring Creek	Cedar Run	Slab Cabin Run	Spring Creek	Cedar Run	Slab Cabin Run
Brown Trout (*Salmo trutta*)	44	10	1	38	13	1
Common shiner (*Luxilus cornutus*)	0	0	<1	0	0	0
Pearl dace (*Margariscus Margarita)*	0	0	<1	0	0	0
Fathead minnow (*Pimephales promelas*)	0	0	7	0	0	9
Blacknose dace (*Rhinichthys atratulus*)	0	0	3	0	0	2
Longnose dace (*Rhinichthys cataractae*)	0	0	30	0	0	1
Creek chub (*Semotilus atromaculatus*)	0	0	2	0	0	0
Fallfish (*Semotilus corporalis*)	0	0	<1	0	0	0

White sucker (*Catostomus commersoni*)	2	5	14	<1	4	4
Banded killfish (*Fundulus diaphanus*)	0	0	3	0	0	1
Bluegill (*Lepomis macrochirus*)	0	<1	0	0	0	<1
Tessellated darter (*Etheostoma olmstedi*)	0	0	1	0	0	1
Slimy sculpin (*Cottus cognatus*)	76	78	8	28	43	16

Table 22. Estimate densities (number per 100 m) of age-1 and older brown trout (95% confidence intervals in parentheses) in May from Spring Creek, Cedar Run, and Slab Cabin Run sites during pre-restoration (1992) and post-restoration (2001-2002) study periods. Where no brown trout were captured on the final pass, the total number of fish captured on all passes was considered the density estimate; no confidence intervals are given.

Stream and sampling site	1992	2001	2002
Spring Creek			
SP1	106 (106-126)	77 (67-87)	70 (67-74)
Cedar Run			
CR1	29 (29-30)	46 (45-47)	50 (49-52)
CR2	34 (34-35)	74 (66-82)	41 (40-43)
CR3	16 (16-18)	33 (31-38)	21 (19-24)
CR4	8 (8-9)	24 (20-31.5)	15 (15-15)
Slab Cabin Run			
SL1	1 (1-2)	0	0
SL2	2 (2-3)	3	1
SL3	1 (1-2)	2	3
SL4	3 (3-5)	7	8 (8-9)

SL1 during spring post-restoration surveys. At all sites in Slab Cabin Run, density estimates in May were extremely low (<10 individuals per site).

During August surveys, densities of age-0 brown trout varied considerably among years and streams (Table 23). In Spring Creek, densities of age-0 brown trout were about 94% lower in post-restorations surveys than in the pre-restoration survey. In Cedar Run during August, densities of age-0 brown trout fluctuated among years and sites, but total density among all stations was 53% higher in 2000 and 162% higher in 2001 than in 1992. In Slab Cabin Run age-0 brown trout were found in just a few surveys before and after restoration.

During August surveys, densities of age-1 and older brown trout varied considerably among streams and years (Table 23). In Spring Creek, density in 2000 was nearly identical to that in 1992, but in 2001 density was nearly 50% than in 1992. In Cedar Run, densities of age-1 and older brown in all sampling sites after restoration were generally lower than in 1992, averaging 25% less in post-restoration versus pre-restoration surveys. In Slab Cabin Run, densities of age-1 and older brown trout were low in all stations before and after restoration.

Ratios of densities age-1 and older brown trout in Spring Creek relative to Cedar Run changed little from pre- to post-restoration (5.10 to 4.45) and differences were not significant (Kruskal-Wallis test, $P = 0.64$). Similarly, in Slab Cabin Run there was little change in ratios (4.30 to 3.75; $P = 0.39$) from pre- to post-restoration. These tests confirm observations that there were no significant changes in brown trout densities after riparian restoration.

Table 23. Estimated densities (number per 100 m) of age-0 and age-1 and older brown trout (95% confidence intervals in parentheses) in August from Spring Creek, Cedar Run, and Slab Cabin Run sites during pre-restoration (1992) and post-restoration (2000-2001) study periods. Where no brown trout were captured on the final pass, the total number of fish captured for all passes was considered the density estimate; no confidence intervals are given. During summer 2001, no sampling occurred at sampling station SL1.

Stream and sampling site	Age Class	1992	2000	2001
Spring Creek				
SP1	0	197 (39-1009)	18 (18-19)	7 (17-19)
	1+	69 (59-187)	70 (66-75)	104 (103-107)
Cedar Run				
CR1	0	22 (14-43)	4	39 (31-51)
	1+	14 (14-15)	14 (14-14)	22 (22-23)
CR2	0	22 (16-50)	14 (14-20)	95 (19-558)
	1+	44 (43-48)	10	25 (25-25)
CR3	0	26 (24-30)	63 (62-65)	43 (41-45)
	1+	13 (13-14)	14 (14-15)	7
CR4	0	2 (2-3)	29 (29-31)	12 (12-13)
	1+	6 (6-7)	10 (10-11)	13 (13-13)
Slab Cabin Run				
SL1	0	0	23 (19-31)	
	1+	1 (1-2)	1	

SL2	0	0	0	0
	1+	2 (2-3)	1	0
SL3	0	0	1	6
	1+	1 (1-2)	2	3
SL4	0	14 (14-19)	9 (9-10)	8
	1+	6 (6-8)	7 (7-7)	3

Discussion

Proportion of Restored Riparia

When this riparian restoration project was designed in 1991, our goal was to install streamside fences along at least 75% of the unfenced pastures. We met this goal on Slab Cabin Run, but a change of ownership of one property and a change in management of another led to a reduction in the length of streamside fencing. Currently, 61% of the original unfenced pastures have well maintained fences and stable riparian areas. Renewed efforts will be needed to convince landowners to participate in fencing programs.

In the Cedar Run basin we were able to obtain voluntary participation by all but two landowners, both of whom have short reaches of unfenced pastures. About 98% of the former unfenced pastures have well maintained fences and stable banks. The remaining unfenced pastures, though small, contribute noticeable amounts of sediment to the stream.

Channel Characteristics

Riparian restoration can potentially improve stream habitats through establishment of stream bank vegetation that narrows the stream channel, causes an increase in stream velocity, and induces a shift from fine to coarse stream substrates. Even though riparian vegetation, primarily grasses, quickly became established in restored sections, channel dimensions did not change after restoration. We made these measurements in 2001 when stream discharge was well below normal, which was reflected in reduced stream width, depths, velocities, and discharges in channel transects. Spring Creek had the most stable channel characteristics; discharge in channel transects was similar in both study years. In Cedar Run and Slab Cabin Run, the low water levels resulted in much lower discharges in 2001 compared to 1992. The study site in Spring Creek is located directly downstream of a large spring, providing a relatively stable source of ground water. Whereas, study sites in Cedar Run and Slab Cabin Run have varied interactions between stream flow and ground water. Wohl (1993) reported longitudinal variation in stream flows in Slab Cabin Run and Cedar Run; he identified sections along the channels where stream flow increased from springs and tributaries and where it decreased by entering the substrate.

In 2001 we found larger percentages of silt and sand in nearly all sample sites than in 1992. Here again, it is likely that this increase in fine substrates was the result of reduced stream flow for several consecutive years. Precipitation from 1999 through 2001 was below normal; hence stream flows were less than average. A resurvey of sample sites in 2004 would be useful to test this idea, because stream flows in 2003 were well above normal.

Intensive grazing has been reported to cause changes in channel characteristics and to decrease quality of trout habitat, by increasing stream widths and reducing depths (Knapp and Matthews 1996). Keller and Burnham (1982) found that grazing exclosures created narrower and deeper pools and improved fish habitat in stream sections adjacent to grazing exclosures. No similar changes in channel characteristics occurred in restored sections in this study. However, changes in channel morphology may develop over a longer period of time. There have been few high stream flow events since post restoration measurements were taken; the absence of channel-forming stream flows may explain the lack of change in channel characteristics.

Discharge

Daily stream discharges at gauging stations were markedly lower in 2001 in all study streams due to below average precipitation; precipitation returned to near normal levels in 2002 and stream discharge increased, particularly during the latter part of the year. The effect of low precipitation on discharge was most pronounced in Slab Cabin Run. Losses of groundwater pumped from the State College Water Authority wellfields adjacent the stream in conjunction with natural losses of stream water in Slab Cabin Run led to reduced discharge in the channel. Groundwater withdrawals from wellfields adjacent to the channel have changed throughout the study periods. In 1991-1992, 1993-1994, and 2001, average daily water withdrawals were 11.1, 12.9, and 10.3 million liters per day, respectively (D. Nevel, State College Water Authority, personal communication). Withdrawals of groundwater were less during drier years; but, losses of groundwater to wellfields may have reduced already low stream flows in 2001.

Water Quality

Restoration reduced suspended sediment in Cedar Run and Slab Cabin Run. Baseflow and stormflow sediment concentrations decreased in both restored streams. Relative to Spring Creek, Cedar Run baseflow TSS decreased by about 40% and Slab Cabin Run baseflow TSS decreased by about 80%. Changes in stormflow TSS relative to Spring Creek are more difficult to interpret, because storm flow TSS increased in Spring Creek between study periods. New urban development in the Spring Creek basin may have increased runoff since the pre-restoration studies.

Median TSS during storm flow in Cedar Run in 2001 was lower than in 1992, but the opposite was true in 2002; hence, we cannot conclude that restoration influenced sediment loading during storm flow. In Slab Cabin Run storm flow TSS was substantially lower in 2001 and 2002 than in 1992. But, much of this reduction could have resulted from the much reduced discharge. Annual sediment yields in all streams followed the same trend as that of TSS among years. Sediment yield in Spring Creek was substantially higher than in previous years. In Cedar Run, sediment yield was lower in 2001 and higher in 2002 than in 1992. The low sediment load in Slab Cabin Run in 2001 and 2002 was surely influenced by low stream flow. Other studies found large reductions in sediment concentrations and yield after streambank fencing. Owens et al. (1996) reported a 60% decrease in sediment concentrations during storms and a 40% decrease in annual sediment loss from an Ohio watershed after streambank fencing. In a Pennsylvania study, streambank fencing resulted in a reduction in concentrations of storm flow suspended sediment by 21-54% and a 10-25% reduction in sediment yield at fenced sites (Galeone 2000).

Restoration did not have any influence on the already low ortho-phosphate or total phosphorus concentrations. In Cedar Run, baseflow and storm flow concentrations of ortho-phosphate and total phosphorus changed slightly relative to Spring Creek. Because no nutrient data prior to streambank fencing were collected from Slab Cabin Run, no inferences about the effects of restoration on its phosphorus levels can be drawn. In the post-restoration study, ortho-phosphate and total phosphorus concentrations were relatively high when discharge was low in Slab Cabin Run. During low flows, temperatures may have increased in warm weather and caused dissolved oxygen levels to diminish. If anoxic conditions developed, iron hydroxides would have been reduced and the phosphorus bound to the hydroxides would have been released (Vadas and Sims 1999).

Restoration had no influence on nitrate-nitrogen and total nitrogen concentrations in Cedar Run. Annual yield of nitrate-nitrogen decreased after restoration in Cedar Run and Spring Creek. But, changes in discharge also contributed to decreased yields. In a similar study, Galeone (2000) reported that total nitrogen concentrations decreased by 20-30% after streambank fencing.

In this study, a 2- to 3-m riparian buffer reduced sediment loading but had no effect on nutrient loading. Other studies recommend a minimum 9- to 10-m buffer for filtering sediment from overland flow (Osborne and Kovacic 1993; Castelle et al. 1994). Fencing a relatively narrow buffer in Cedar Run and Slab Cabin Run achieved fairly large reductions in sediment. A wider buffer, however, would have been more effective at filtering nitrate-nitrogen and total nitrogen. After reviewing numerous studies on buffer strip width, Vought et al. (1994) recommended a minimum buffer strip width of 10-20 m for removing nutrients from overland flow. In another review, Fennessey and Cronk (1997) advocate using 20- to 30-m buffers for removing nitrate-nitrogen from subsurface flow.

Improved filtering capacity of nitrate-nitrogen might have also been achieved if trees had been planted in riparian buffers. Peterjohn and Corell (1984) found that a riparian forest retained 79% of nitrate-nitrogen from surface flow and an average of 94% from subsurface flow from an adjacent agricultural field. Osborne and Kovacic (1993) compared forest and grass buffer zone ability to remove nutrients from subsurface flow and retain them over time. Over all seasons, the forest buffer was more effective at reducing nitrate-nitrogen than was the grass buffer.

Complete exclusion of livestock from streams may also have increased nutrient reductions from restoration. At animal crossings and accesses, animal waste can still directly enter Cedar Run and Slab Cabin Run, adding some nutrients to the streams. In pasture systems such as these, where the stream bisects the pasture, livestock crossings cannot be avoided.

Substrate Composition

Restoration led to some reductions in fine sediment in stream substrate. Fine particles in areas judged to be potential spawning habitat were reduced and Fredle indices increased in Cedar Run, where flows were low but continuous in the post-restoration study. However, similar changes were not found in Slab Cabin Run. The extremely low flows in Slab Cabin Run during the post-restoration study may have permitted fine particles to accumulate even in sections with higher velocities. Spring Creek had some changes in fine particles and in Fredle index values, indicating that increased deposition of sediment may be occurring there.

Improvements in substrate may mean more favorable habitat for spawning brown trout and better embryo survival in Cedar Run. Survival-to-emergence is directly proportional to the Fredle index (Lotspeich and Everest 1981) for coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*). Another study found a similar relationship between brown trout embryo survival and the Fredle index (Beard 1990). Not only is survival related to substrate size, but condition of emerging brown trout also improves with larger particle size and increased permeability. Witzel and MacCrimmon (1983) reported that alevins from redds with larger gravel size emerged later and were further developed than those from redds with finer substrate.

Temperature

Restoration did not moderate stream temperatures in Cedar Run or Slab Cabin Run. Rather stream temperatures in restored streams and Spring Creek were more dissimilar in the post-restoration sampling period. Low water levels may have been responsible for the increased variability in stream temperatures in restored streams; Slab Cabin Run had the most extreme fluctuations in discharge and the most extreme temperatures. Differences in the volume and proximity of groundwater sources to stream gauging stations also may have influenced temperatures. A large contribution of groundwater to stream flow, as occurs in Spring Creek, provides more stable temperatures than in Cedar Run or Slab Cabin Run.

Even if water levels had been more favorable in the post-restoration study, the grass riparian buffers may not have been effective in moderating temperatures in restored streams. To better understand stream temperatures in response to riparian buffer type, Blann et al. (2002) modeled temperature regimens in streams with grazed, grass/forbs, and wooded riparian buffers. Results of their model show that grass/forb buffers have limited ability to sufficiently shade streams for temperature moderation; in summer, temperatures would actually increase as streams flow through grass/forbs and grazed areas.

Macroinvertebrate Communities

Despite some improvements in physical and chemical stream variables in restored streams, macroinvertebrate responses to restoration were limited. The presence of some Plecopteran individuals in Cedar Run and Spring Creek indicate that stoneflies may be colonizing these streams. Spring Creek also had some fluctuations in community composition, indicating that even in the absence of restoration, stream community composition changes occur. Taxa richness and diversity did not increase in restored streams relative to Spring Creek. Few other studies document the response of macroinvertebrate communities to restoration. However, Galeone (2000) reported that macroinvertebrate richness increased and the percent dominant taxa decreased within a year of streambank fencing. Densities of macroinvertebrates in Cedar Run and Slab Cabin Run significantly increased in 2001 and 2002 compared to 1992. Densities in the two restored streams were more similar to those in Spring Creek in 2001 and 2002 relative to 1992.

Variations in stream discharges during study periods may have influenced macroinvertebrate communities. Periods of high flows can result in a high rate of macroinvertebrate drift and ultimately decrease densities (Rosillon 1989). Low flows in the post-restoration study (2001 and early 2002 when macroinvertebrates were last sampled) may have had the opposite effect, allowing densities to accrue in Spring Creek, Cedar Run, and Slab Cabin Run. On the other hand, macroinvertebrate community composition and structure may have been limited by poor water quality at extremely low flows, as occurred in Slab Cabin Run. A study on drought-stressed streams in Portugal reported that dissolved oxygen concentrations decreased at low flows; the macroinvertebrate community was dominated by dipterans at sites that periodically went dry (Pires et al. 2000). In ephemeral streams in Pennsylvania, McManaman (2001) found that taxa richness, evenness, and diversity varied across sites with a gradient of water levels; variables had the lowest values at sites that were dry for the longest periods. The macroinvertebrate community in Slab Cabin Run may still recover when higher flows return, given the ability of macroinvertebrates to quickly recolonize. Pires et al. (2000) noted that macroinvertebrate diversity and density rebounded shortly after periods of low flow.

Fish Communities and Brown Trout Densities

The composition of fish communities remained stable from pre-restoration to post-restoration surveys; as typical of cold-water streams, study streams had few fish species. Cedar Run and Slab Cabin Run communities were similar to those in pre-restoration surveys. Slab Cabin Run had the fewest brown trout, but it has the highest richness of fish species, including cyprinids, bluegills, slimy sculpins, banded killifish, and tessellated darters. With the exception of brown trout, the fish species found in Slab Cabin Run typically occur in a cool or warm-water streams; those species may be more tolerant to warm stream temperatures in the summer than brown trout.

Densities of brown trout were not influenced by riparian restoration. There was considerable year-to-year variation in numbers of age-1 and older brown trout among sites, but it seems unlikely that this variation masked potential increases related to restoration. There was also considerable variation in numbers of age-0 trout among sites, but no indication that their densities were increasing after restoration, despite some improvements in water quality and in reductions in fine substrates in potential spawning habitat. If embryo survival increased after restoration, it would have been reflected in densities of age-0 and age-1+ brown trout. Beard (1990) found a strong relationship between embryo survival and densities of adult brown trout in Spring Creek.

Other habitat requirements besides spawning habitat may be constraining brown trout densities. Shirvell and Dungey (1983) described different microhabitats used by brown trout for feeding and spawning. Although spawning habitat may have been improved, other habitat needs may not have been met. If the availability of preferred microhabitats constrains brown trout populations, then a restoration aimed at improving habitat may result in increased brown trout densities. Additionally, we observed little overhead cover or undercut banks in restored sections. The use of rip-rap for bank stabilization has been criticized because it does not provide habitat complexity (Schmetterling et al. 2001). Banks stabilized with rip-rap do not undercut and they have limited ability to support riparian vegetation, which provides overhead cover.

The fluctuations in stream flows may also be confounding any effects of restoration on brown trout densities. The low stream flows in the post-restoration study could limit densities of brown trout, masking any potential improvements in spawning success. Drought conditions result in declines in densities of age-0 and age-1+ fish, as seen in a population of anadromous sea trout (*Salmo trutta*) in an English stream (Elliott et al. 1997). Authors cite mortality due to increased water temperatures during drought periods as limiting densities.

This study underscores the challenges of ecological research in a field setting. Changes in the amount of precipitation and, consequently, in stream flows between the study periods added uncontrolled variation to the study. Further variation was added due to changes at the landscape level. Urban development in the Spring Creek basin and changes in the amount water withdrawals from ground water in the Slab Cabin Run basin occurred from pre-restoration to post-restoration studies. We have only been able to speculate about the effects of these added sources of variation on results.

Additionally, natural variation, especially in stream communities, was difficult to separate from changes due to riparian restoration. Highly variable fish and macroinvertebrate communities may have changed in response to fluctuations in stream flows or other environmental variables. Because data used to describe pre-restoration and post-restoration periods were collected within relatively short time spans, it provides only a snapshot of stream communities under those conditions. Longer-term studies are needed to better characterize natural variation and to distinguish any changes in communities due to natural processes from those due to riparian restoration. Rinne (1988) recommended 10-year or longer studies to adequately evaluate the influences of grazing on fish populations; a similar time scale for pre-restoration and post-restoration studies may be necessary to fully assess the effects of streambank fencing on stream systems.

Conclusions

Since the completion of restoration efforts, a narrow vegetated buffer was established between riparian fences and the streams. The most notable effects of restoration were reductions in the proportion of un-vegetated streambank and sediment inputs from pastures. Suspended sediment at baseflow significantly declined after restoration. Decreased suspended sediment resulted in less substrate fine particles in potential brown trout spawning habitat in Cedar Run. Similar reductions in fine substrate particles did not occur in Slab Cabin Run. Low flows in the post-restoration period in Slab Cabin Run may have permitted fine particles to accumulate. A smaller proportion of pastures in the Slab Cabin Run basin were fenced compared to the Cedar Run basin.

Narrow riparian buffers were not effective in reducing nutrients in streams flowing through restored areas, nor were there any measurable changes in stream temperatures. Composition of macroinvertebrate communities did not change after restoration, but there were significant increases in densities of macroinvertebrates after restoration. Composition of fish communities in treated streams were unchanged, and there were no consistent changes in densities of young-of-the-year or adult brown trout after restoration.

Responses to restoration were confounded by the effects of low flows during the post-restoration study. Longer-term studies that capture natural variation in flows and, in turn, stream communities could further differentiate improvements in stream conditions due to restoration from changes due to other variables. Additionally, studies over a longer recovery time after restoration might capture some improvements in stream biota that were not seen in this investigation.

Despite the limitations of this study, some positive effects of riparian restoration were observed. In the short term, restoration was effective at reducing the amount of un-vegetated streambanks and, thereby, reducing suspended sediment and deposited substrate sediment. The improvements in water quality and substrate composition may have the potential to enhance macroinvertebrate communities and brown trout densities in the future.

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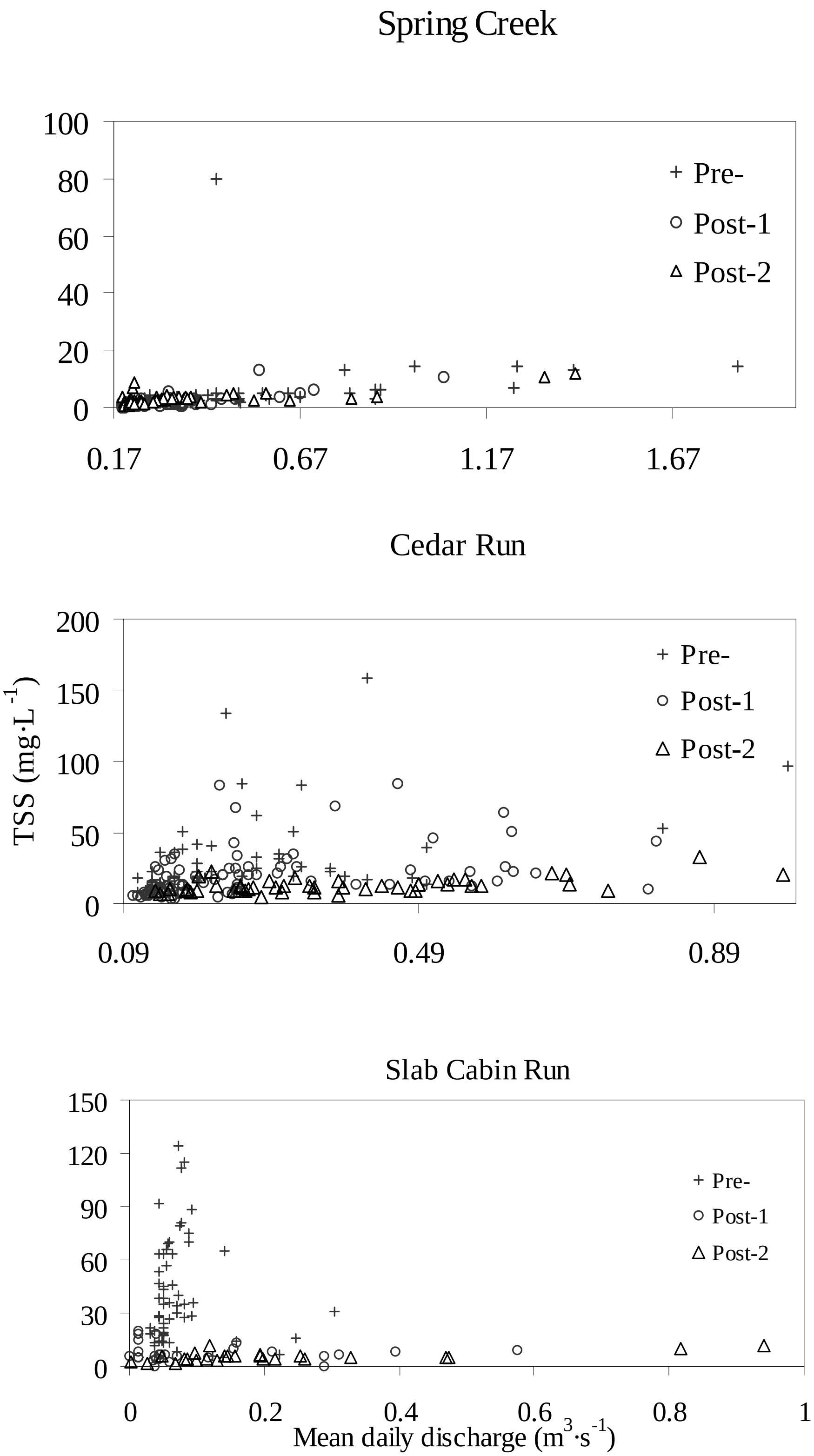
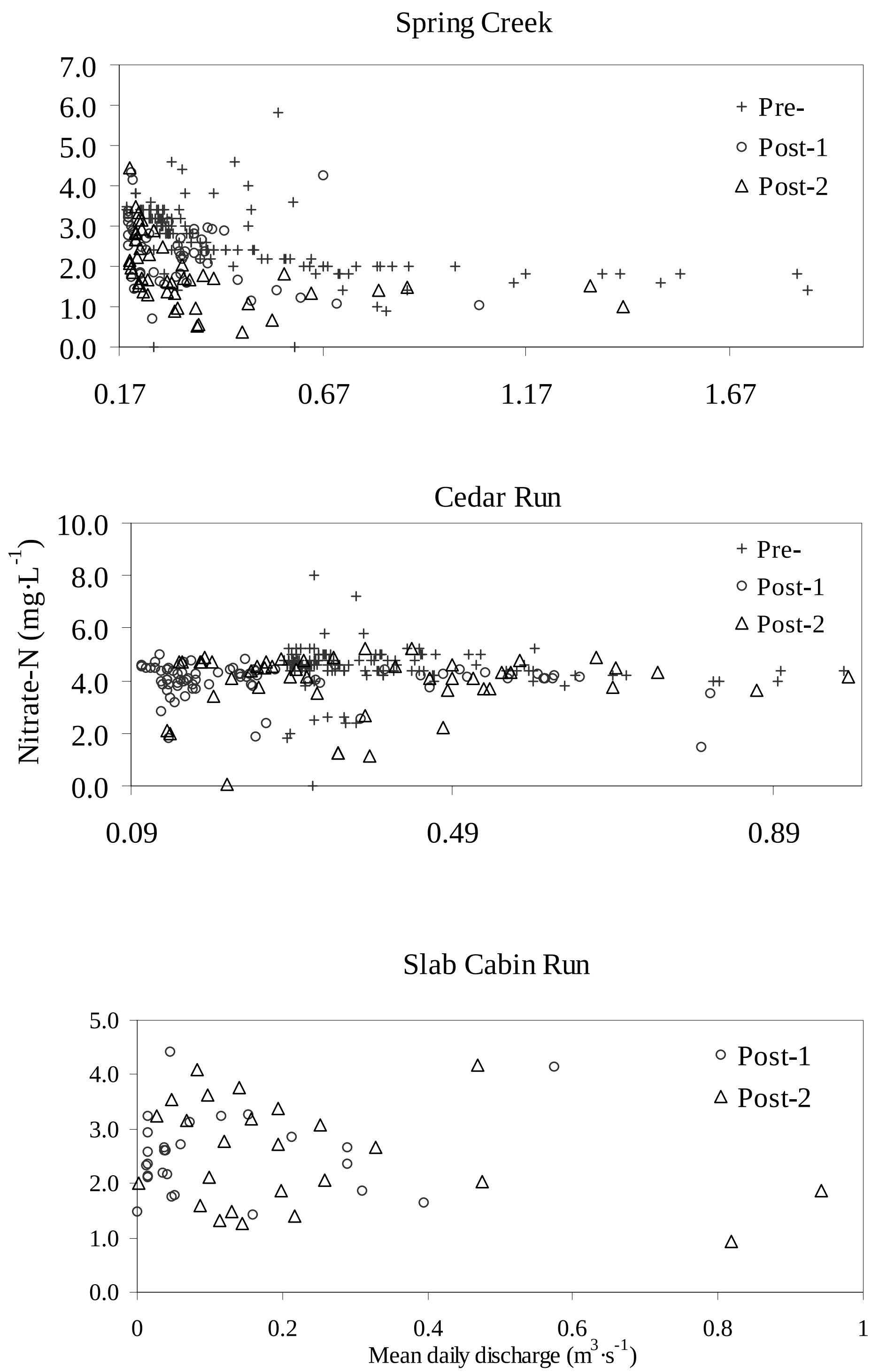
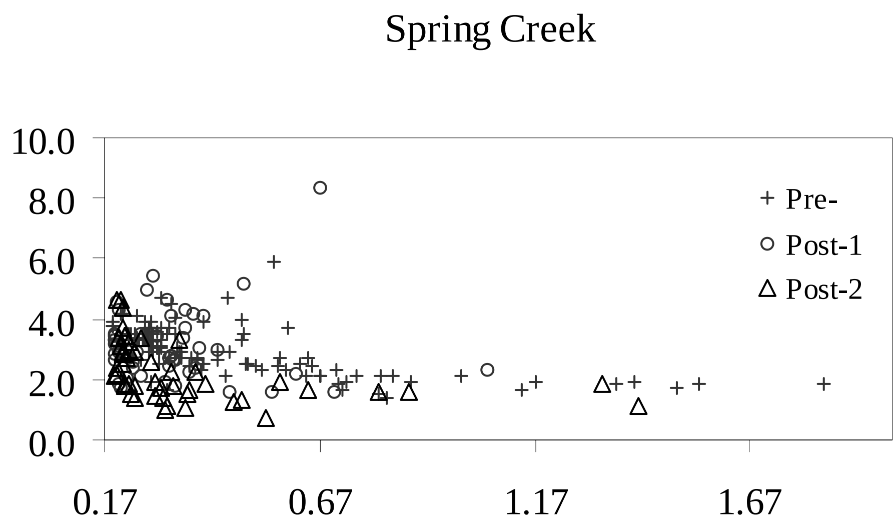
Appendix A. Baseflow TSS concentrations ($mg \cdot L^{-1}$) and mean daily discharge ($m^3 s^{-1}$) from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994) and post-restoration (2001-2002) study periods.

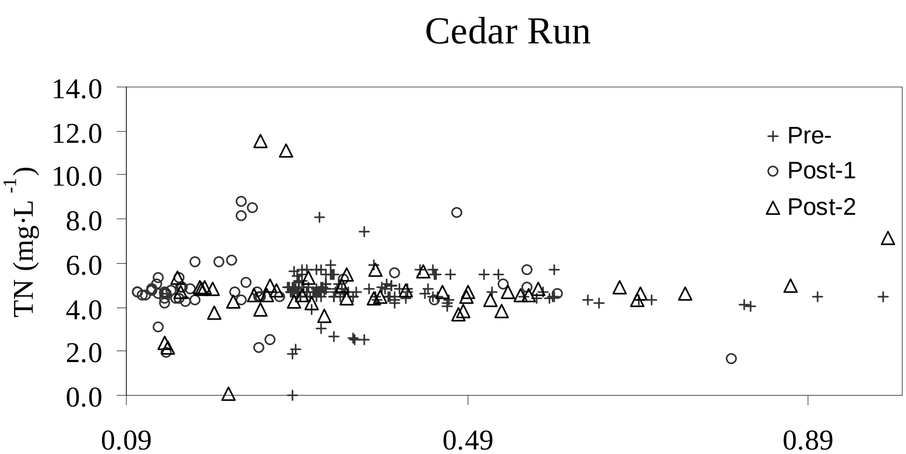


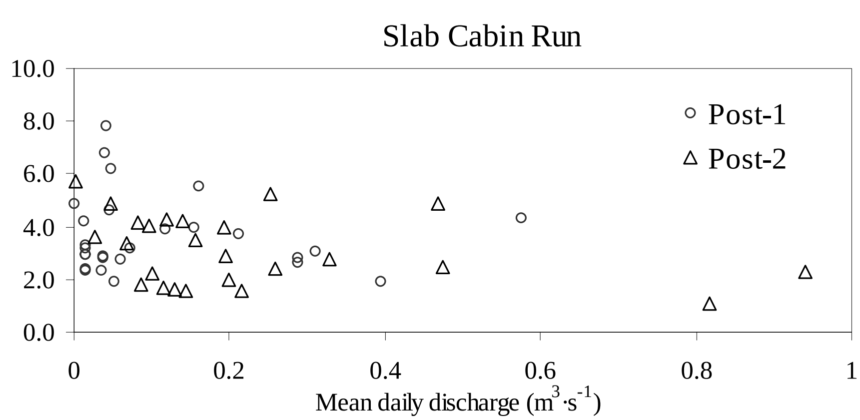
Appendix B. Baseflow ortho-phosphate concentrations (mg·L^{-1}) and mean daily discharge (m^{3}s^{-1}) from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994) and post-restoration (2001-2002) study periods.



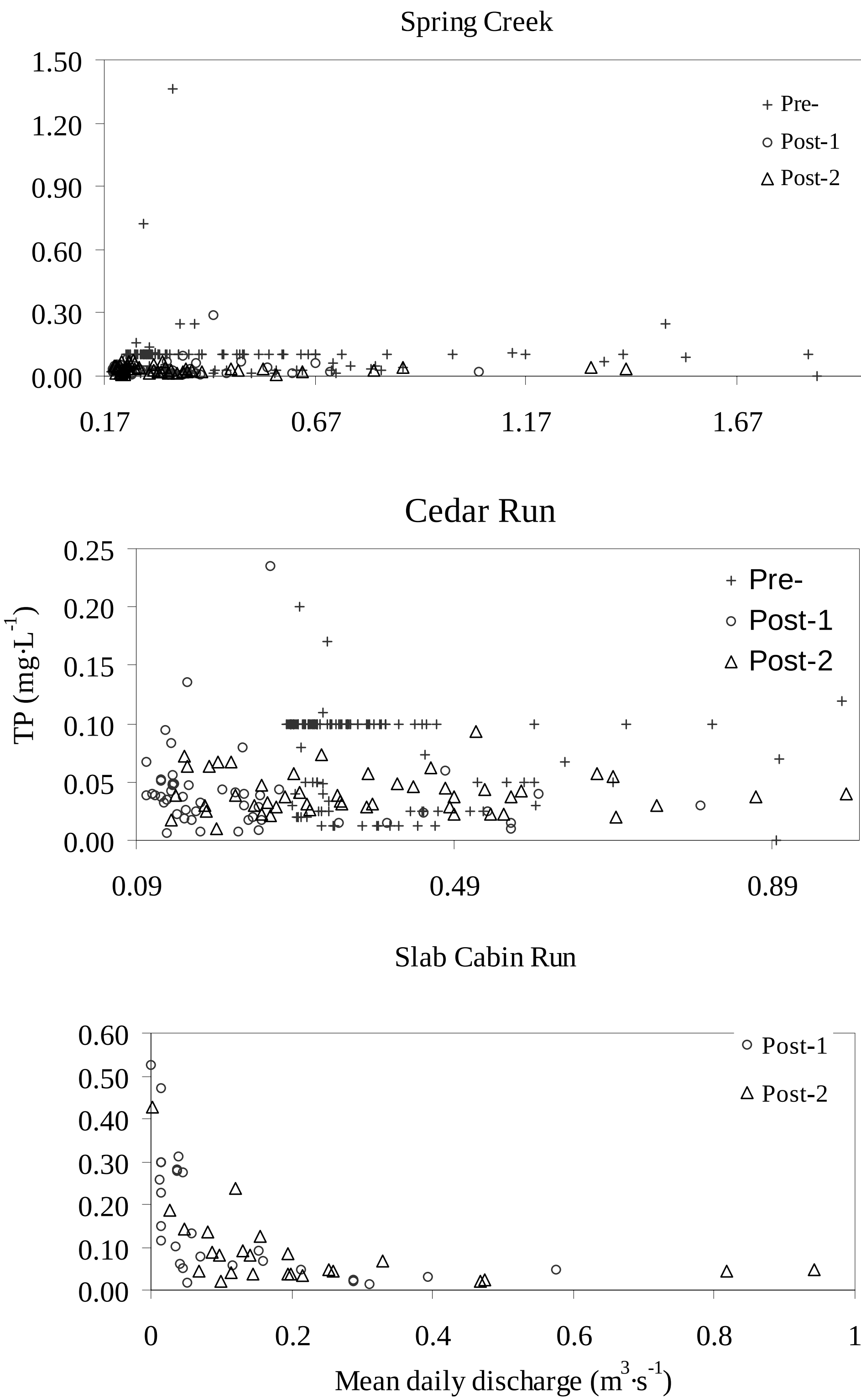
Appendix C. Baseflow total phosphorus concentrations (mg·L^{-1}) and mean daily discharge (m^{3}s^{-1}) from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994) and post-restoration (2001-2002) study periods.







Appendix D. Baseflow nitrate-nitrogen concentrations (mg·L^{-1}) and mean daily discharge (m^{3}s^{-1}) from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994) and post-restoration (2001-2002) study periods.



Appendix E. Baseflow total nitrogen concentrations (mg·L^{-1}) and mean daily discharge (m^{3}s^{-1}) from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 and 1993-1994) and post-restoration (2001-2002) study periods.