Conservation Effects Assessment Project (CEAP)
Spring Creek Watershed Study

ANALYSIS OF BMP IMPLEMENTATION PERFORMANCE AND MAINTENANCE IN SPRING CREEK, AN AGRICULTURALLY-INFLUENCED WATERSHED IN PENNSYLVANIA

A CSREES* Competitive Grant Watershed (Now NIFA—National Institute of Food and Agriculture)

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Preface/Acknowledgements

This report summarizes the results from this collaborative research project. Each of the five components (ecological and ground-based monitoring, landscape characterization and coarse- vs. fine-grained assessment, hydrologic and landscape modeling, socio-economic analyses, and outreach) is covered separately. To avoid conflict with journal requirements, those intended for publication elsewhere (landscape characterization & coarse- vs. fine-grained assessment; and hydrologic and landscape modeling) include only a summary of the methods and results, along with supplemental information regarding the published citations and contacting authors. The remaining sections are covered entirely within this report.

We would like to thank the USDA's Conservation Effects Assessment Project (CEAP) for funding this project, which enabled us to determine: (1) not only the best indicators for monitoring water quality but also the time needed for obtaining accurate results; (2) the most appropriate spatial scales for assessing BMP effectiveness; (3) the best hydrologic and landscape models for evaluating BMP performance; (4) the factors affecting landowner adoption of BMPs; and (5) the most effective means of communicating water quality issues with the public.
We hope the information from this document will aid fellow researchers, land managers, extension agents, environmental regulators and others in their efforts to improve the science of best management practices for improving the water quality of our watersheds. We appreciate the cooperation of the citizens of the Spring Creek Watershed that provided access to their properties, gave freely of their time and insight during interviews and surveys, and who continue to strive to conserve the features, habitats, and services of this valuable resource.

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Additional information and future publications from this project will be listed at the following site, www.riparia.psu.edu, under a BMP label.

EXECUTIVE SUMMARY

The Spring Creek watershed is one of 13 Cooperative State Research Education and Extension Service (CSREES*) watershed studies initiated under the Conservation Effects Assessment Project (CEAP) in an effort to quantify the environmental effects of conservation practices by recognizing that environmental quality is the result of a complex interaction between these practices, the hydrology in the landscape, social and economic factors, and outreach efforts. In conjunction with this goal, the study has five inter-related components:

- Ground-based monitoring and ecological analyses
- Landscape characterization and coarse- vs. fine-grained assessment
- Hydrologic and landscape modeling
- Socio-economic analyses
- Outreach.

The Spring Creek watershed is located in central Pennsylvania within the Ridge and Valley physiographic province. Land use patterns follow the topography of the region, with the ridges mostly forested and the limestone valleys primarily in agricultural and urban land uses. This watershed is under increasing pressure from urbanization. As a result, agricultural land use decreased from 1938 to 2006, while areas of developed land increased and forest land use remained relatively stable during this time period.

Ground-based Monitoring and Ecological Analyses

Ground-based monitoring and ecological analyses are summarized as two parts. The first part covers monitoring and analysis of Best Management Practices (BMPs) implemented in two sub-basins of Spring Creek, both of which were thought to be the major sources of fine sediment impacting the mainstem.

Our main objectives were to determine: (1) the effectiveness of stream-bank fencing, stream crossings, and bank stabilizations in reducing fine sediment; (2) the utility of traditional water chemistry measures vs. substrate, aquatic macroinvertebrates, and fish as water quality surrogates; and (3) the optimal monitoring period for evaluating BMP performance. The study area included two treatment streams (Cedar Run and Slab Cabin Run) flowing through unfenced pastures prior to BMP implementation. Pre-treatment monitoring occurred from 1991 – 1992 and was followed by two phases of post-treatment monitoring (phase I: 2001 – 2002; phase II: 2005 – 2007). Comparison of pre-treatment vs. post-treatment results revealed the following:

- Baseflow and stormflow water quality results showed total suspended solids (TSS) in both treatment streams were substantially less in phase II monitoring period than during the pre-treatment and the initial phase I periods, while nutrient concentrations did not reveal any obvious trends.
- Fine sediment in stream substrates declined significantly in both treatment streams, but the response in Slab Cabin Run was delayed by drought.
- Macroinvertebrate densities increased in treatment streams following restoration in both phase I and phase II monitoring periods. All community metrics increased in both treatment streams by phase II, but only taxa richness was significantly higher. Reach-level differences within each stream revealed significant responses in all metrics at the farthest downstream sites. The Cedar Run site displayed a gradual increase throughout the monitoring period, while the Slab Cabin Run site response was delayed until phase II, most likely due to dry conditions.
- Fish community composition did not change in the treatment streams between pre- and post-treatment periods, but density of age-1 and older brown trout did increase in both streams following treatment, although the response was small for Slab Cabin Run where brown trout densities were lower than Cedar Run and the reference stream.

The second part of the ecological analyses was the assessment of cross vanes in Slab Cabin Run. Cross vanes (V-shaped structures of rocks or logs) were installed in an incised section of Slab Cabin Run flowing through Millbrook Marsh in an effort to elevate the stream surface and promote flooding of the wetland during storm flows, thus reducing non-point source pollution by filtering suspended pollutants. We deployed *in situ* water quality monitors during major storm flow events to measure the following variables both upstream and downstream of three cross vanes: pH, temperature, dissolved oxygen, and turbidity. Results from two storm events in Janurary and March (2010) indicated that cross vanes reduced sediment loads by 19.2 % and 5% respectively.

These ecological results led to the following take-home messages:

1) Specify which onsite stressors the BMP(s) will address and monitor the BMP with the appropriate indicator.
2) Be aware of the limitations of the BMP.
3) Identify the hydrologic nature of the stream and allow sufficient monitoring time to account for responses to hydrologic fluctuations and other stressors.
4) Consider cross vanes as an effective BMP for reducing non-point source pollution in areas where flooding is desired (e.g., stream-side wetlands).

Landscape Characterization and Coarse- vs. Fine-Grained Assessment

Optimal size and placement of BMPs down slope of animal heavy use areas is crucial for treatment success. However, visual inspection does not necessarily ensure BMPs are placed where pollutant flows will enter the stream. We summarized land use within a 100-m buffer on either side of the stream, and located agriculturally-based heavy use areas using high-resolution aerial photography. We then evaluated the effect of DEM resolution on flow path calculations by comparing coarse- (30 m), medium- (10 m), and fine-grained (1 m) elevation maps with each other and the straight-line distance.

Agricultural land use within the riparian buffer was highest in the watersheds with BMPs (Cedar Run 46%, Slab Cabin Run 37%) and lowest in the Upper Spring Creek watershed (17%). The BMP-treated watersheds had lower percentages of forest land use, but all three watersheds had similar percentages of residential and developed land. Across all watersheds, topographically-based flow paths were substantially longer than straight-line paths with a median value of 19 m longer than that of the straight-line path for both the 1-m (71% longer) and 10-m (49% longer) DEMs, and 48 m longer for the 30-m DEM (91% longer). The stream offsets were also considerably different, with median differences of 81 m for 1-m DEM, 25 m for 10-m DEM, and 85 m for the 30-m DEM.

Important findings from the coarse- vs. fine-grained assessment were as follows:

1) Current NRCS standards do not include recommendations to examine actual flow path from the heavy use area to the stream when determining related BMP placements.

2) Visual assessments and straight-line paths from heavy use areas to streams often give misleading estimates of flow path lengths and stream entry points, resulting in inefficiently placed BMPs.

3) When determining BMB placement for a particular site, the 10-m DEM appears to be adequate. While the 1-m DEM data provides more accurate results, these data are not widely available and require more intensive processing.

Hydrologic and Landscape Modeling

This section is composed of two parts: I) a landscape characterization evaluating the effect of topographic resolution on parameter estimates related to sediment transport and riparian buffers; and II) an examination of the utility of various simulation models for estimating the relative importance of upland, riparian, and in-stream sources on sediment loads.

The main objectives of the landscape characterization were to examine the effect of topographic data resolution on (1) estimates of stream corridor characteristics related to sediment transport and (2) watershed metrics related to riparian buffers. Results indicated the following:

- Finer resolution topographic data yield more accurate stream maps that better define stream channel location, length, and sinuosity than maps constructed from coarser topographic data.
- High-resolution topographic data from LiDAR remote sensing support detailed maps of stream and riparian characteristics, such as the stream cross-section and floodplain area. Coarser resolution topographic data are not precise enough to calculate such measures.
- The LiDAR-based maps of stream and riparian characteristics help identify riparian buffers and provide indicators of characteristics and processes that control how buffers affect stream water quality.

- Less than 4% of the riparian zone is forest or wetland.
- The effects of topographic resolution on watershed-average buffer potentials remain unclear; but at the scale of individual stream reaches, maps based on coarse resolution data can clearly provide misleading information on the positioning of buffers between croplands and streams.
- LiDAR data provide the topographic detail needed by stream simulation models to estimate gully erosion, streambank erosion, and floodplain deposition.

The objectives of the hydrologic and landscape modeling were to (1) estimate the relative importance of hillslope, gully, stream bank erosion, and floodplain deposition to stream sediment loads; and (2) compare results from the Sednet model to observed sediment loads and SWAT model predictions. We obtained the following results:

- A SWAT model with two slope classes and a low runoff factor for all land uses gave the best calibration to flow measurements from the Spring Creek Houserville gauge (Nash-Sutcliffe (NS) coefficient: 0.63).
- Massive hypothetical land use changes changed stream flow predictions of the SWAT model in the expected directions, but the changes were small relative to the prediction errors of the calibrated model. Land use scenarios restricting changes to the riparian zone produced much smaller changes in predicted discharges.
- Sednet estimates of bankfull discharge generally exceeded SWAT estimates by more than 100%. Sediment yields, however, were less than 50 percent of the SWAT predictions. The model suggested that incised stream reaches, which resulted in higher stream power, and limited floodplain deposition primarily contributed to the predicted loads.
- The Sednet model predictions were highly sensitive to land management practices in the stream corridor. Throughout the upper Spring Creek watershed, hypothetically converting all land cover in the mapped stream corridor to agricultural lands increased the predicted annual average load at the Houserville gauge by 26%. Converting the land use/land cover to forest/wetland resulted in

sediment loads almost 90% lower than the sediment load predicted under current land cover conditions.

Important messages discerned from the landscape characterization and hydrologic/landscape modeling components of the project include:

1) Fine-scale LiDAR-based topography data are essential to modeling stream processes affecting sediment transport at the watershed scale. Coarser-scaled elevation data does not provide enough information to identify incised channels, estimate channel dimensions, or identify riparian and floodplain areas.

2) Hillslope processes affect stream sediment loads more by effects on channelized flow connections than by direct contributions of sediment delivery.

Socio-Economic Analyses

Our main objectives were to explore the factors that affect effective farmer adoption of BMPs and citizen perception of water quality. In 2009, we conducted a two-stage analysis of riparian landowners throughout all sub-basins of the Spring Creek watershed. The first stage included semi-structured interview of landowners and organization representatives; the second stage consisted of a mail-back questionnaire to riparian landowners. To assist our analysis of survey results, we created three landowner types based on land use: traditional farmers, hobby farmers, or non-farmers. Survey results were as follows:

- One third of respondents (33%) owned <1 acre.
- Respondents were well educated, with 30% holding graduate degrees.
- Respondents expressed agreement with pro-environment (70%), moderately pro-innovation (54%), and pro-private property rights (56%) attitudinal measures.
- Landowner respondent types were traditional farms (17%), hobby farms (15%), and non-farmers (63%).
- Sixty-one percent of respondents who cut lawn or vegetation near their stream do so within three feet of the stream.
- Traditional farmers perceived greater knowledge about the stream on their farm than hobby farmers.

- Non-farmers rated water quality of the stream on their property higher than traditional farmers.
- Thirty-eight percent of landowners were willing to have a buffer on their property, while 21% were not at all willing.
- Landowners identified five obstacles to buffer adoption: that buffers take up too much land and time to maintain, that buffers don't make sense for the size of their property, that buffer plants look messy, that buffers don't fit the appearance of their neighborhoods, and that buffers would bother their neighbors.

These were the take-home messages from the socio-economic portion of the project:

1) Non-farmers are a bottleneck for riparian buffer adoption across the watershed.
2) The amount a landowner hears about riparian buffers is also positively related to the amount heard about Chesapeake Bay water quality.
3) Willingness to adopt riparian buffers increases with perceived knowledge about stream water quality.
4) Baseline willingness will increase with more positive attitudes towards riparian constraints, suggesting the presence of a group of landowners who strongly support riparian buffers.
5) Riparian buffers are socially desirable based on the proportion of neighbors considered close friends. While this may encourage buffer adoption at the neighborhood scale, it may also discourage buffer adoption in areas where normative behaviors disapprove of riparian vegetation.
6) Stream flow permanence is positively related to landowners' perceptions of water quality, attitudes of stream importance, and perceptions of how buffers may improve environmental outcomes. This has important implications for riparian management and water quality in ephemeral stream reaches.
7) If landowners believe that buffers produce results, their willingness to adopt buffers will increase, suggesting a need for more education on local and downstream ecosystem services provided by riparian buffers.

Outreach

Members of the Spring Creek research team attended the annual meetings of the USDA-CSREES National Water Conference in 2008, 2009, and 2010. Technical presentations were made during these meetings and at other scientific venues.

In June 2010 Canaan Valley Institute (CVI) organized a workshop for the public and community officials within the Spring Creek Watershed. The goal of the workshop was twofold: to use the results of the socio-economic survey to educate those least likely to adopt riparian buffer practices on their importance and to disseminate the results of the USDA Best Management Practices in the Spring Creek Watershed Project to the residents of the watershed. Presentations on the results of the BMP project were given by key personnel from the project, followed by a question/answer session.

Despite the mailing of individual invitations followed by an email invitation, attendance at the workshop was lower than we anticipated. However, the homeowners in attendance were engaged in the conversation and made inquiries into how buffers may be established on a non-agricultural land. Due to feedback from attendees, we conclude that this type of public workshop is an effective means of communicating scientific research projects to the public. However, it is our recommendation that additional marketing is necessary to increase attendance at a voluntary workshop.

In May 2010, the Spring Creek research team hosted CEAP's National Synthesis Team for a two-day workshop and field tour to inform them of our findings.

In December 2010, Brooks presented the Spring Creek BMP findings to the Pennsylvania State Technical Committee monthly meeting for the Natural Resources Conservation Service. Following the presentation, there were questions and discussion from the members on how to incorporate these findings into future guidance.

Conclusions

This study illustrates the synergistic nature of watershed management, of which BMPs play an integral part. These results show the success of a BMP depends on a variety of factors including: (1) proper identification of the locations and sources of pollution, including the actual pathways by which it enters the stream; (2) proper alignment of the BMP with those locations; (3) effective monitoring techniques that target the stressor(s), match the BMP with the appropriate indicator, and allow sufficient time periods for capturing responses; and (4) capitalizing on proven factors that encourage landowner adoptions of BMPs, while effectively addressing impediments to BMP adoption. In addition, BMP success or failure is often dependent on external factors within the watershed, especially land use change and multiple-year weather patterns (e.g., drought, flooding). Thus, it is important to consider these additional impacts to water quality when attempting best management practices.

THE SPRING CREEK WATERSHED: LANDSCAPE SETTING AND LAND USE CHANGE

The Spring Creek watershed is one of 13 Cooperative State Research Education and Extension Service (CSREES*) watershed studies initiated under the CEAP program (Figure 1) in an effort to quantify the environmental effects of conservation practices by recognizing that environmental quality is the result of a complex interaction between these practices, the hydrology in the landscape, social and economic factors, and outreach efforts. In conjunction with this goal, the study has five inter-related components:

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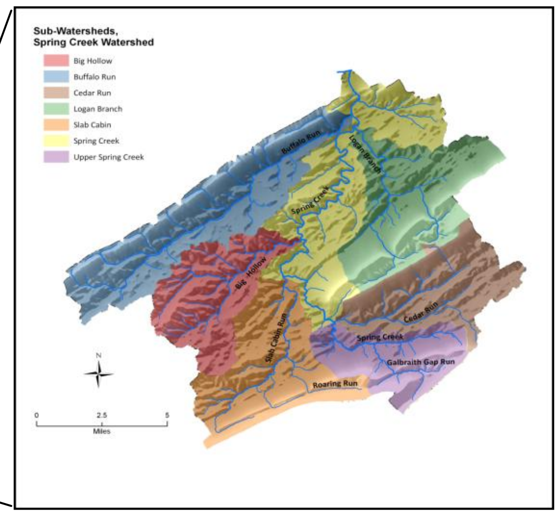
- Hydrologic and Landscape Modeling
- Socio-Economic Analyses
- Outreach



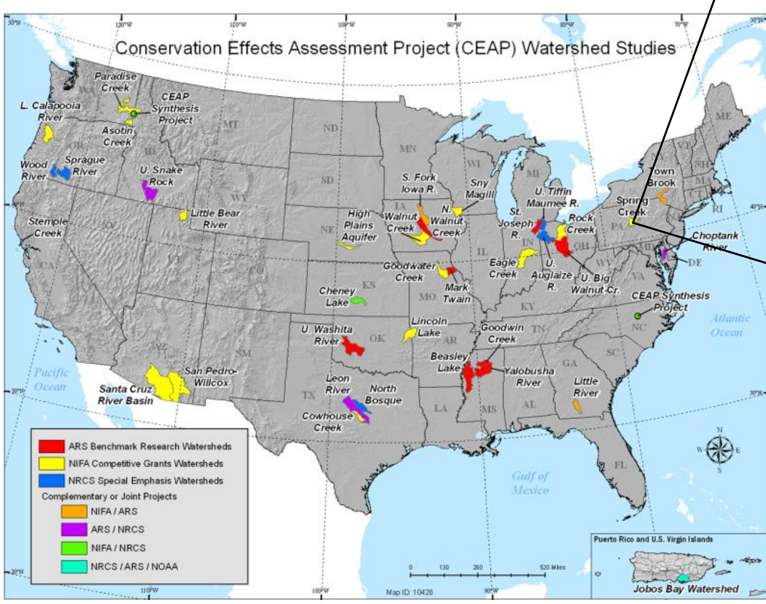


Figure 1. Maps depict the locations of the USDA-CEAP projects throughout the nation (http://www.nrcs.usda.gov/Technical/NRI/ceap/image/ceapmap.pn). The expanded view is of the Spring Creek watershed, located in Central Pennsylvania. Color shading and boundaries delineate the sub-watersheds within Spring Creek. Most of the on-the-ground activity took place in the three sub-watersheds to the south (Slab Cabin Run, Cedar Run, and Upper Spring Creek).

The Spring Creek watershed is located in central Pennsylvania within the Ridge and Valley physiographic province. Mean annual precipitation and temperature for State College, which lies within the watershed, are 97 cm and 49.4° F, respectively. Monthly averages for precipitation range from 6.2 cm (February) to 9.8 cm (May), while average monthly temperatures typically range from 26.5° F in January to 71.7° F in July (Pennsylvania State Climatologist 2010). Spring Creek is fed by seven major tributaries (Table 1), encompasses approximately 146 m^2 of surface-water drainage, and is part of the West Branch of the Susquehanna River Basin. The surrounding physiography produces a trellis drainage pattern where small headwaters often run down rocky sandstone ridges to join the mainstems in the limestone valleys (Kaktins and Delano 1999). During periods of low flow, much of the surface runoff is lost in fractures and sinkholes and returns to the valley via limestone springs, providing much of the base flow to Spring Creek (Fulton et al. 2005).

Land use patterns are also influenced by the topography of the region, with the ridges mostly as forest and the limestone valleys primarily as agriculture and urban land use. Through expansion of The Pennsylvania State University and nearby communities, the Spring Creek watershed is under increasing pressure from urbanization, and land use patterns have changed considerably over time. Aerial photography was interpreted to track land use change mapping the recent history of the Spring Creek watershed in Central Pennsylvania. We collected current and historic digital aerial photography from seven dates (1938, 1949, 1957, 1971, 1993, 2000, and 2006) and interpreted each to map land use conditions across time (Figure 2). Studies have shown that land use condition has direct relationships to water quality measures. As land condition changes and developed areas expand, the effects of urban and suburban areas that result in increased impervious surfaces have a direct relationship on erosion, storm water runoff rates, and in-stream energy. This in turn affects the biological integrity of the streams. For example, thresholds of as little as 10% imperviousness produce detrimental effects on trout populations (Schueler 1994). By 2000, the proportion of impervious surface in the Spring Creek watershed had surpassed this estimate, reaching 13.3% impervious surface in 2006 (Table 2). However, Spring Creek is still capable of supporting wild trout populations, most likely due to the substantial amounts of groundwater input (Carline et al. –in prep).

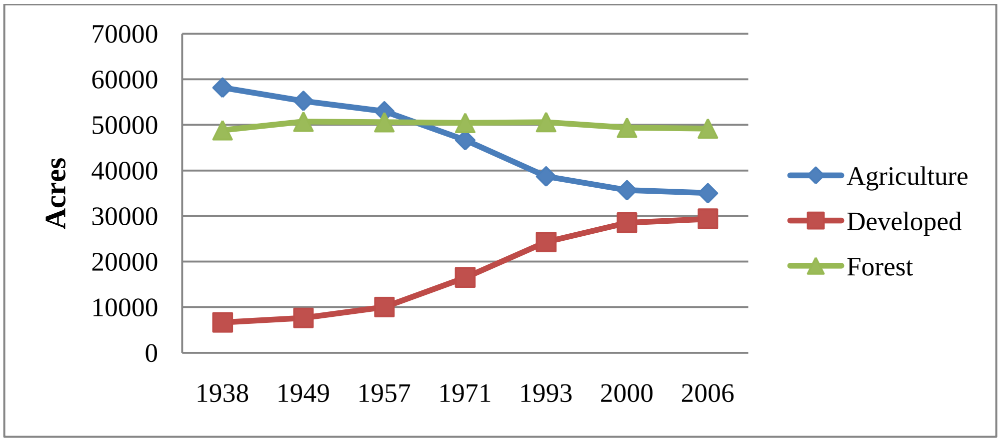


Figure 2. This line-graph tracks the land use change in the Spring Creek watershed from 1938 to 2006.

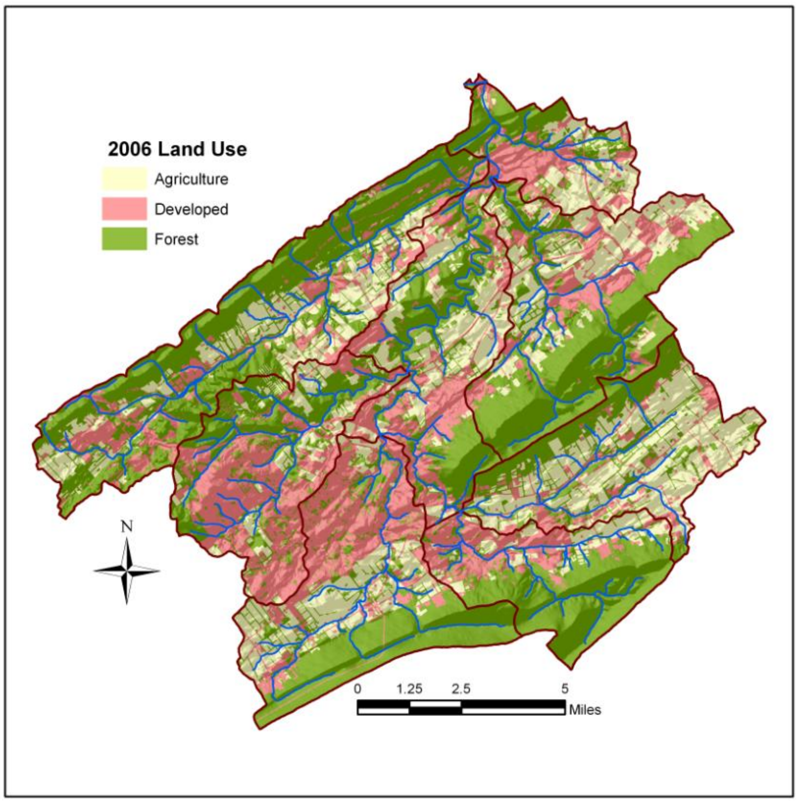


Figure 3. The map illustrates land use condition based on the 2006 aerial photography. Background shading is meant to help highlight the topography in the study area. While most of the headwater streams are under forested cover, almost all of the valleys have been converted to agricultural, suburban, and urban land uses.

Table 1. Percent land use for each of the sub-watersheds found within the Spring Creek watershed. Refer to Figures 1 and 3 to view the watershed areas.

	% Agriculture	% Developed	% Forest
Big Hollow	17.26	54.94	27.80
Buffalo Run	27.25	21.33	51.42
Cedar Run	62.05	14.03	23.92
Logan Branch	28.24	21.23	50.02
Lower Spring Creek	33.64	33.35	33.02
Upper Spring Creek	16.67	27.23	56.10
Slab Cabin Run	27.13	36.99	35.88

Table 2. Change in percent impervious surface in Spring Creek watershed from 1938 to 2006.

	% Impervious Surface
1938	3.06
1949	3.61
1957	4.46
1971	7.19
1993	10.80
2000	12.78
2006	13.30

GROUND-BASED MONITORING AND ECOLOGICAL ANALYSES

This component of the project is summarized as two parts: I) monitoring and ecological analysis of BMPs (stream-bank fencing, stream crossings, and bank stabilizations) implemented in two sub-basins of Spring Creek (Cedar Run and Slab Cabin Run), both of which were thought to be the major sources of fine sediment impacting brown trout (*Salmo trutta*) in the Spring Creek watershed; and II) installation of rock cross vanes in Slab Cabin Run to reconnect the incised stream channel with its floodplain and the surrounding Millbrook Marsh.

Part I: Monitoring and Ecological Analysis of Cedar Run and Slab Cabin Run BMPs

Background

The study area included two treatment streams (Cedar Run and Slab Cabin Run) and one reference stream (Upper Spring Creek), which did not contain any unfenced riparian pasture (Figure 4). Both treatment streams flowed through unfenced pastures and were the focus of a collaborative riparian restoration project conducted by several local private organizations and public agencies in order to reduce fine sediment loads to Spring Creek. Riparian landowners were willing to install narrow (3-4 m wide), grass buffer strips along the majority of unfenced pasture. Both streams were monitored prior to BMP installation in 1992. Post-restoration monitoring followed repeatedly from 2000 through 2007 (Carline and Walsh 2007, Wohl and Carline 1996).

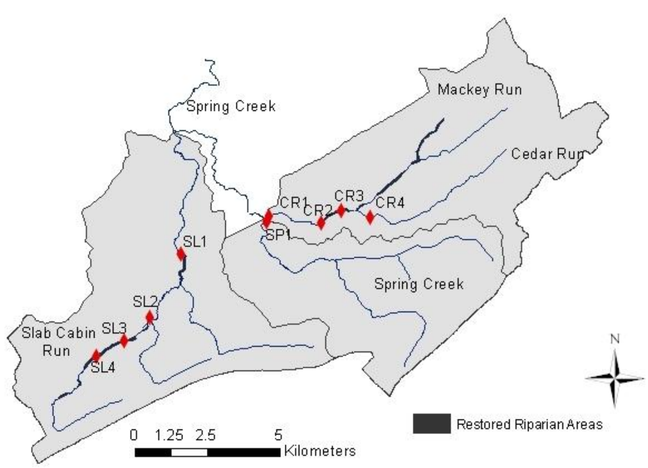


Figure 4. Spring Creek watershed map delineating sub-basins and sampling locations for evaluation of ecological responses to riparian restoration.

Treatment (BMPs) consisted of stream bank fencing (i.e. narrow grass buffers), stream crossings, and bank stabilization and resulted in 98% and 61% of the previously unfenced pastures being treated for Cedar Run and Slab Cabin Run, respectively. Consult Carline and Walsh (2007) for further information on restoration techniques. Four sampling sections (two grazed/two ungrazed) were established in each treatment stream (Figure 4). Pre- and post-restoration monitoring consisted of substrate, fish and macroinvertebrates sampling, as well as baseflow and stormflow monitoring for suspended solids and nutrients. The overall project timeline was as follows:

- Pre-treatment monitoring (1991-1992)
- Construction on Slab Cabin Run (1992-1995)
- Construction on Cedar Run (1993-1998)
- Phase I post-treatment monitoring (2001-2002)
- Phase II post-treatment monitoring (2007- 2008).

Main objectives were to determine:

- *The effectiveness of specific BMPs in reducing fine sediment;*
- *The utility of traditional water chemistry measures vs. substrate, aquatic macroinvertebrates, and fish as water quality surrogates;*
- *The optimal monitoring period for evaluating BMP performance.*

Figure 5. Stream-bank fencing and stream crossing along Cedar Run.

Methods

Water Quality: Water samples were collected during base flow and storm flow events. Between August 2007 and August 2008, reference and treatment streams were sampled during five storm flow events, which were defined as events raising discharge levels to 20% above base flow levels. During storms, hourly water samples were collected with automatic samplers (Hach American Sigma 900) at each of three gauging stations. Six of these water samples (two coinciding with the rising limb, two with the peak, and two with the falling limb of the hydrograph) were analyzed for total suspended solids (TSS), Ortho-Phosphate, Total-Phosphate, Nitrate-Nitrogen, and Total Nitrogen. Results were compared to pre-treatment and earlier post-treatment periods (Carline and Walsh 2007).

Substrate Composition: During previous studies, substrate similar to brown trout spawning habitat was sampled in May 1992 prior to riparian restoration and in 2001, 2002, and 2005 after riparian restoration. In this study, we again sampled substrates in May 2007. Samples were taken in areas where velocity ranged from 0.25 to 0.57 m/s and depth ranged from 0.2 m to 0.5 m, which corresponds to 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 substrate samples at each of nine sites (four sites each on test streams, one site on reference stream) (Figure 4). 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. Percent fines, defined as percent substrate sample weight of particles less than 1 mm, was used as the primary index of substrate permeability.

Macroinvertebrate Community: We used a Surber sampler to collect triplicate macroinvertebrate benthic samples from riffle habitats at each of the nine sites in May and August for both the current study year (2007) and previous study years (1992, 2000, 2001, 2002, and 2005). All samples were fixed in 10% formalin and transferred to 90% ethyl alcohol before sorting. Insect taxa were identified to genus-level whenever possible (except Chironomidae to family); non-insect taxa were typically identified to class or lowest taxonomic level possible.

Seasonal sampling results were analyzed separately. We described changes in macroinvertebrate community composition between years and streams through qualitative comparisons of taxa relative abundances and yearly trends in ratios of reference stream to treatment stream macroinvertebrate densities. For a more comprehensive quantitative statistical analysis of community responses to restoration, we applied nested general linear models to macroinvertebrate density, taxa richness, EPT richness, and Shannon diversity index (Pielou 1975) metrics to test whether communities in treatment streams were significantly different between years, streams, and/or sites. The equation for the general linear model was as follows:

Y = Year (Stream) + Stream + Site (Stream) + Error.

Significant changes in metric values across years would imply a significant year effect within each stream. Significant changes in metric values between streams or between sites within a stream would imply either that streams/sites responded differently to restoration or that streams/sites had inherently different communities. Either result would suggest the need to analyze streams/sites separately to control for this variation. In the event of significance, we applied one-way ANOVAs followed by Dunnett's multiple comparisons (which allow the specification of the pre-restoration data as a control) to determine which specific factors were significantly different between pre- (1992) and post-restoration years. To meet normality and equal variance assumptions, density data were transformed by natural log. For most combinations of year/stream/site/season, community composition and diversity metrics did not require transformation to meet these assumptions. Those that did were analyzed for significant differences between years through nonparametric (Kruskal-Wallis) methods, since transformations were unsuccessful.

To differentiate between community responses to restoration and natural variation due to climate or other regional environmental factors, we also compared the reference stream metric data between years. Lack of significant changes in reference post-restoration years or the lack of similar temporal trends between the reference stream and treatment streams would imply that changes in the treatment stream communities were due to restoration and not natural environmental changes.

Fish Community and Brown Trout Density: Direct current electrofishing gear (200 V) was used to survey fish communities in May and August. During previous studies, fish communities were sampled in 1992 and 2000-2002. In the present study, fish were sampled in 2007. Brown trout densities were estimated with the successive removal method in approximately 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).

Results

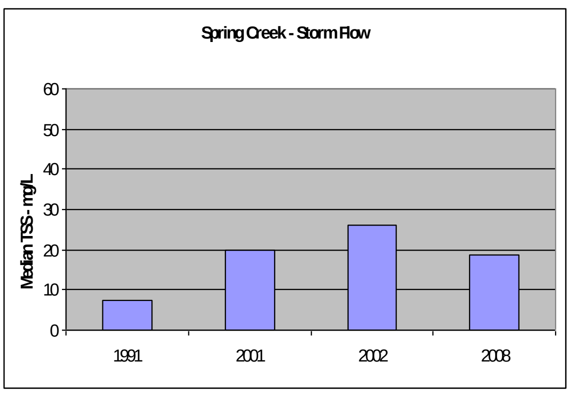
Water Quality: Base-flow and storm-flow water quality results showed total suspended solids in Cedar Run and Slab Cabin Run were substantially less in 2007-2008 (phase II) than during the pre-treatment and the 2001-2002 (phase I) post-treatment periods. By phase II, mean base-flow TSS levels for Cedar Run had decreased from 17.75 mg/L in 1992 to 1.0 mg/L. This reduction was even more dramatic in Slab Cabin Run, where average base-flow TSS levels decreased from 29.3 mg/L to 1.0 mg/L (Table 3). Although base-flow TSS levels also declined at the reference stream (4 mg/L to 1 mg/L), mean stormflow TSS levels actually increased during post-treatment monitoring years (Table 4). Unlike TSS, nutrient concentration did not reveal any obvious trends. Both baseflow and stormflow Ortho-phosporus and total phosphorus concentrations were relatively low throughout the monitoring period (Tables 3 and 4). Nitrate concentrations also changed little between pre- and post-treatment years. Although Cedar Run had higher base-flow and storm-flow nitrate concentrations (Nitrate-N and Total Nitrogen) than both Upper Spring Creek and Slab Cabin Run, these values were similar to the average for the mainstem of Spring Creek (4.1 mg/L) (Spring Creek Watershed Association 2009).

Table 3. Median sediment and nutrient concentrations (mg/L) and interquartile ranges in baseflow samples from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1991-1992 for TSS; 1993-1994 for nutrients) and post-restoration (2001-2002, 2007-2008) study periods. No nutrient data are available for Slab Cabin Run during 1993-1994.

	Spring Creek				Cedar Run				Slab Cabin Run			
	Pre-	2001	2002	2007-2008	Pre-	2001	2002	2007-2008	Pre-	2001	2002	2007-2008
TSS	4	1.9	2.7	1.0	17.75	9.8	11.4	1.0	29.3	6.6	5.4	1.0
	(2.2-6.0)	(1.1-2.9)	(1.7-3.7)	(1.0-1.0)	(13.3-27.1)	(6.5-14.9)	(9.4-14.2)	(1.0-2.75)	(17.6-46.3)	(5.4-9.0)	(4.0-6.6)	(1.0-3.0)
	N = 61	N = 48	N = 50	N = 8	N = 62	N = 48	N = 48	N = 8	N = 52	N = 28	N = 27	N = 7
Ortho-P	0.003	0.003	0.003	0.005	0.003	0.004	0.005	0.005		0.041	0.009	0.016
	(0.003-0.004)	(0.003-0.009)	(0.003-0.023)	(0.005-0.008)	(0.003-0.003)	(0.003-0.012)	(0.003-0.024)	(0.005-0.007)		(0.007-0.119)	(0.003-0.034)	(0.011-0.029)
	N = 188	N = 48	N = 50	N = 8	N = 181	N = 48	N = 50	N = 8		N = 28	N = 27	N = 7
Nitrate-N	2.4	2.4	1.68	2.85	4.45	4.34	4.31	4.58		2.44	2.43	3.17
	(1.80-3.20)	(1.72-2.88)	(1.33-2.52)	(2.42-3.33)	(4.20-4.80)	(4.17-4.46)	(3.64-4.72)	(4.39-4.78)		(1.93-3.00)	(1.60-3.24)	(2.76-3.75)
	N = 190	N = 48	N = 50	N =7	N = 182	N = 48	N = 50	N = 8		N = 28	N = 27	N = 6

Table 4. Median sediment and nutrient concentrations (mg/L) and inter-quartile ranges in storm flow 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 (2001-2001, 2007-2008) 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	2007-2008	Pre-	2001	2002	2007-2008	Pre-	2001	2002	2007-2008
TSS	7.5 (6.2, 9.1) N = 9	20 (9.1, 42.0) N = 21	26 (18.8, 61.4) N = 23	18.6 (14.0-21.7) N=5	29.4 (20.7, 45.9) N = 8	20.6 (15.1, 34.3) N = 19	33.1 (21.2, 53.9) N = 24	9.7 (4.3-10.0) N=5	62.1 (28.2, 86.2) N = 9	18.2 (10.5, 46.7) N = 12	16.9 (10.8, 51.0) N = 22	9.0 (7.2-29.3) N=5
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	0.004 (0.002-0.012) N=5	0.005 (0.005, 0.008) N = 49	0.003 (0.003, 0.006) N = 15	0.006 (0.003, 0.043) N = 23	0.004 (0.003-0.004) N=5		0.011 (0.001, 0.132) N = 8	0.035 (0.003, 0.099) N = 20	0.031 (0.008-0.035) N=5
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	0.07 (0.06-0.09) N=5	0.05 (0.013, 0.100) N = 45	0.066 (0.022, 0.079) N = 15	0.071 (0.041, 0.154) N = 23	0.022 (0.020-0.038) N=5		0.187 (0.071, 0.273) N = 9	0.107 (0.062, 0.232) N = 20	0.079 (0.076-0.136) N=5
Nitrate-N	1.6 (1.40, 2.20) N = 52	1.78 (1.12, 2.06) N = 15	1.14 (0.81, 1.47) N = 21	1.9 (1.5-2.0) N=5	4.2 (3.95, 4.40) N = 50	3.9 (3.63, 3.97) N = 15	3.57 (3.17, 4.13) N = 23	3.84 (3.76-4.02) N=5		1.48 (1.09, 2.26) N = 9	1.58 (1.07, 2.17) N = 20	1.13 (1.11-3.01) N=5
Total-N	1.9 (1.52, 2.48) N = 52	2.37 (1.78, 3.17) N = 15	1.57 (1.17, 2.16) N = 21	2.2 (1.9-2.3) N=5	4.35 (4.15, 4.64) N = 50	4.25 (4.11, 6.31) N = 15	3.88 (3.68, 4.29) N = 23	4.2 (4.08-4.54) N=5		2.76 (1.98, 4.80) N = 8	2.21 (1.55, 2.53) N = 20	2.02 (1.51-3.55) N = 5



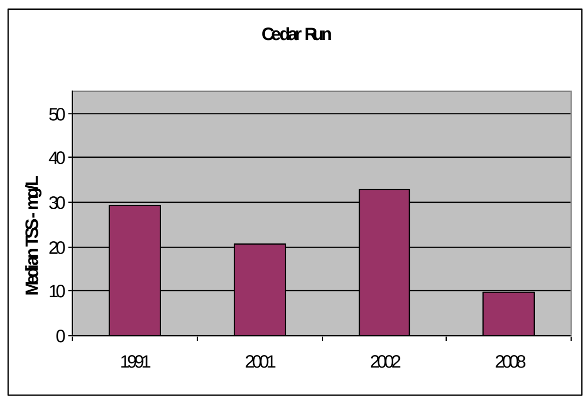
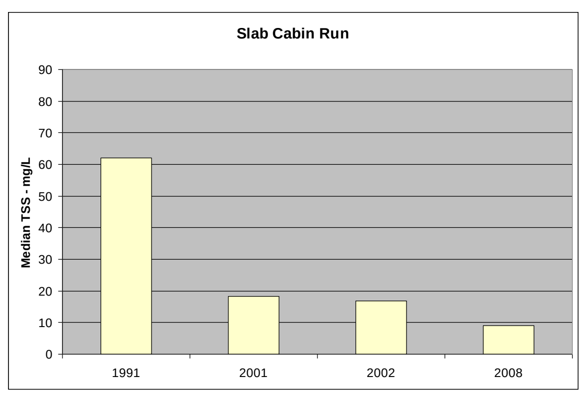
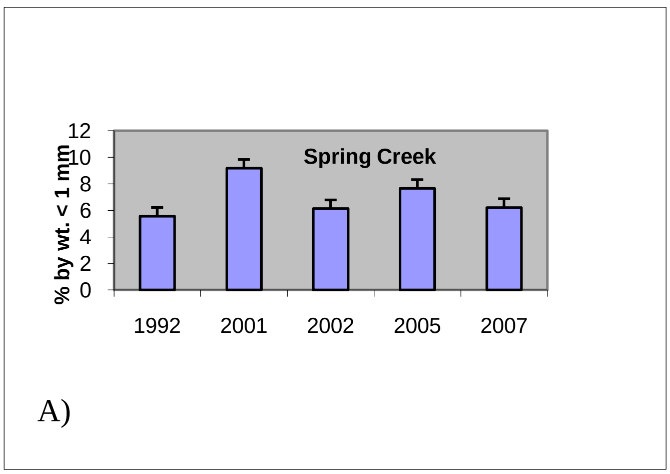


Figure 6. Pre-treatment (1991) vs. post-treatment storm-flow TSS levels for reference (Spring Creek) and treatment (Cedar Run, Slab Cabin Run) streams.



Substrate Composition: Percentage fines in substrate samples from the reference site in Spring Creek from 1992 to 2007 are displayed in Figure 7A. During this time period, median percentage fines ranged 5.6 to 8.2, and none of the pairwise comparisons were significantly different (Mann Whitney test; P<0.05). In contrast, percentage fines in substrates from Cedar Run changed substantially after riparian restoration was completed in the watershed (Figure 7B). Prior to construction in 1992, median percentage fines from all sites were 25.95, and during the post construction years of 2001-2007, median percentage fines continued to decrease from 14.4 to 10.0. Median percentage fines for each post-construction year were significantly different (P<0.001) from the 1992 samples, but were not different (P<0.05) from each other. It is noteworthy that percentage fines prior to construction in Cedar Run were nearly five times higher than that for Spring Creek and that after construction, percentage fines in Cedar Run were approaching values for fines in Spring Creek.



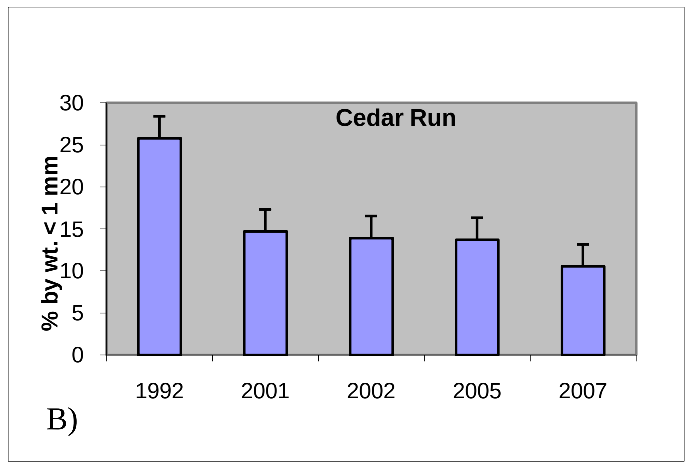
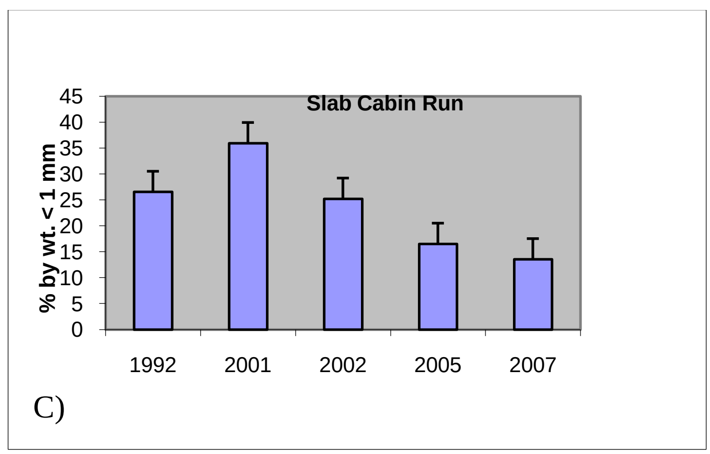


Figure 7. Mean percent by weight of substrate < 1mm from A) the Spring Creek reference site; B) four sampling sites in Cedar Run; and C) four sampling sites in Slab Cabin Run. 1992 = pre-treatment year; 2001-2002 = Phase I monitoring period; 2005-2007 = Phase II monitoring period.



Percentage fines in substrates from Slab Cabin Run, like those in Cedar Run, were high in 1992 prior to construction (Figure 7C). After construction, percentage fines in Slab Cabin Run increased in 2001, which was probably the result of a drought that began 1999. Precipitation returned to normal levels in 2002 and percentage fines decreased, but not significantly from previous years. Percentage fines continued to decrease in 2005 and 2007, when median values were significantly less ($P<0.01$) than those in 1992. Thus, percentage fines in substrates in Slab Cabin Run seemed to respond to riparian restoration, but the response was delayed by drought conditions.

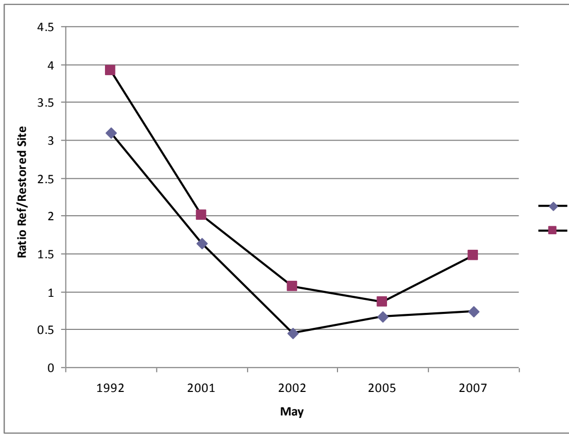
Macroinvertebrate Communities: Macroinvertebrate community composition varied between streams but was similar between sampling seasons (Table 5). In both May and August/ September, Diptera (primarily Chironomidae) were abundant in all streams during both pre-restoration and post-restoration periods. Unlike the reference stream, treatment streams did show declines in dipteran relative abundances during post-restoration periods. Amphipoda were abundant in both Spring Creek and Cedar Run, but represented only a small proportion of total individuals (< 10%) collected from Slab Cabin Run at any time period, with the exception of the farthest upstream site. Isopoda were also highly abundant in Cedar Run; together these three orders comprised at least two-thirds of the individuals collected from this stream from 1992 through 2007. The dominance of drought-tolerant dipterans in Slab Cabin Run is typical of perched streams that tend to dry out during part of the year. Spring Creek experienced spring Oligochaeta blooms during two years of the post-construction period. The presence of Ephemeroptera, Trichoptera, and Coleoptera was consistent for all streams; however, relative abundances fluctuated from rare (< 1%) to common (17 – 22%) regardless of the restoration period. For all streams, Plecoptera were either rare or not present during various years and showed no apparent response to restoration (Table 5).

Prior to restoration (1992), Spring Creek contained 3.1 and 3.9 times the number of macroinvertebrate individuals per square meter in May than Cedar Run and Slab Cabin Run, respectively (Figure 8A). By 2007, this ratio had fallen considerably in both streams, with Cedar Run actually supporting more macroinvertebrates per unit area than Spring Creek (mean density ratio = 0.73). May ratios of macroinvertebrate densities in Slab Cabin Run relative to Spring Creek had fallen to 1.5 by 2007 (Figure 8A). August/September temporal trends in ratios were similar to May with pre-restoration densities in Spring Creek at least twice that of the treatment streams (Figure 8B). By 2002, densities in both treatment streams exceeded those of the reference stream and maintained these ratios through 2007 (Figure 8B).

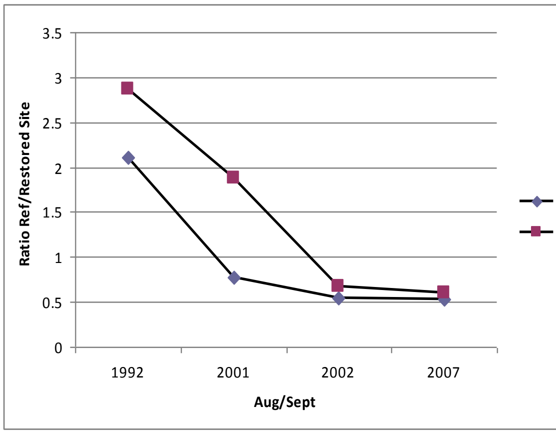
Table 5. Relative abundance of macroinvertebrate taxa (order or class) collected in a) May and b) Fall from Spring Creek, Cedar Run, and Slab Cabin Run during pre-restoration (1992) and post-restoration (2001 thru 2007) periods. In each year, three samples were collected from one reach location in Spring Creek, while Cedar Run and Slab Cabin Run each contained four reach locations where samples were collected, totaling twelve samples per stream.

A)	Spring Creek					Cedar Run					Slab Cabin Run				
	1992	2001	2002	2005	2007	1992	2001	2002	2005	2007	1992	2001	2002	2005	2007
Amphipoda	61	9	23	10	13	17	20	14	26	28	2	5	5	8	10
Isopoda	0	<1	<1	0	<1	30	32	45	25	47	1	20	27	6	9
Coleoptera	3	2	10	2	3	2	9	9	9	8	<1	19	1	4	6
Diptera	21	28	31	62	14	21	19	10	15	7	70	33	40	43	61
Ephemeroptera	3	2	3	14	1	1	2	1	7	1	1	3	<1	22	1
Trichoptera	7	2	17	11	4	14	6	17	17	5	2	2	4	16	3
Plecoptera	0	<1	0	1	<1	0	<1	<1	0	0	<1	0	0	<1	<1
Oligochaeta	2	45	2	0	63	6	8	<1	<1	<1	21	2	14	<1	7
Turbellaria	3	6	5	0	1	1	1	2	0	2	1	1	3	0	2
Other	0	6	10	<1	2	8	3	3	<1	<1	1	15	6	<1	<1

B)	Spring Creek					Cedar Run					Slab Cabin Run				
	1992	2000	2001	2005	2007	1992	2000	2001	2005	2007	1992	2000	2001	2005	2007
Amphipoda	56	35	27	*	11	17	8	8	36	20	14	5	8	23	5
Isopoda	0	< 1	0	*	< 1	22	40	40	13	48	2	56	60	15	41
Coleoptera	5	4	9	*	7	11	19	19	17	14	1	5	3	12	9
Diptera	29	47	19	*	61	35	20	20	20	7	47	18	12	21	29
Ephemeroptera	3	5	4	*	7	1	3	3	2	1	2	5	< 1	2	1
Trichoptera	5	4	5	*	4	11	3	3	8	5	15	4	5	19	9
Plecoptera	0	< 1	0	*	0	0	< 1	< 1	0	0	0	0	0	< 1	0
Oligochaeta	0	< 1	< 1	*	< 1	0	1	1	< 1	1	3	3	4	3	1
Turbellaria	0	2	27	*	5	1	3	3	0	2	4	3	2	0	3
Other	2	1	9	*	5	2	2	2	3	2	12	1	5	5	2



A)



B)

Figure 8. A) May and B) August/September mean macroinvertebrate density ratios of reference stream (Spring Creek) to restored stream (Cedar Run or Slab Cabin Run) from pre-restoration (1992) through post-restoration (2000 - 2007) years.

General linear models on the log-transformed mean density data for Cedar Run and Slab Cabin Run May sampling seasons revealed significant year ($p = 0.000$) and stream effects ($p = 0.074$) but not a significant site effect ($\alpha < 0.10$). Analysis from the general linear models on the August density data indicated year ($p = 0.000$), stream (0.000) and site ($p = 0.003$) effects; however, models applied to each stream separately did not result in significant site effects for either stream. Thus, one-way ANOVAs and Dunnett's multiple comparisons were applied to density results at the stream level only. Mean macroinvertebrate densities for Spring Creek were not significantly different between years for both the May and August/September sampling seasons (Figure 9). Conversely, May samples collected from Cedar Run and Slab Cabin Run had significantly higher ($p = 0.000$) post-restoration than pre-restoration densities for all years except 2002 (Figure 9A). Densities at the reference stream were also depressed in 2002, though not significantly. For August/September, all post-restoration years contained higher macroinvertebrate densities for Cedar Run, while only the latter post-restoration years were significantly higher than 1992 for Slab Cabin Run (Figure 9B). This lack of a significant response in macroinvertebrate density during the initial post-restoration years at Slab Cabin Run is most likely due to dry conditions.

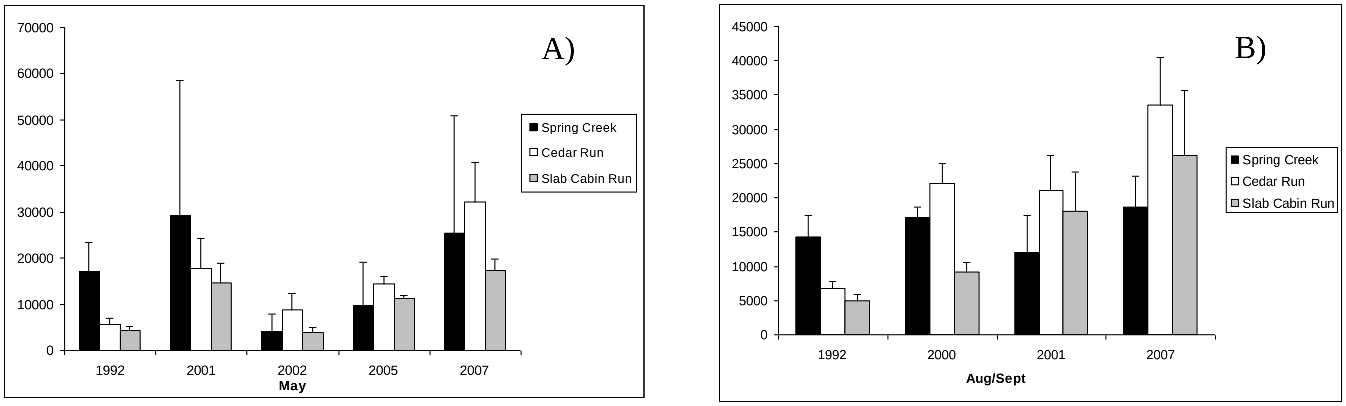


Figure 9. Mean macroinvertebrate densities and SE for (a) May and (b) August/September sampling seasons during pre-restoration (1992) and post-restoration (2000-2007) periods.

Spring Creek had no significant differences in any metric values between years (Table 6). Although community composition did not change remarkably in the treatment streams following restoration, site-level differences within each stream did reveal changes in community composition at the farthest downstream sites. As a result, general linear models run on the May community metrics (taxa richness, EPT richness, and diversity index) all displayed significant year, stream, and site effects (all metrics and factors: $p = 0.000$). Results from the August/ September data were not as consistent (EPT richness—no significant year effect; diversity—no significant stream effect), but we analyzed all sites separately for consistency.

At Cedar Run, the farthest downstream site (ungrazed CR1) displayed a positive response to restoration in both the May and August/September data (Tables 6 and 7). Several EPT genera (*Ephemerella, Drunella, Ceratopsyche, Cheumatopsyche)*, Diptera (*Chelifera,* Tabanidae), and Gastropoda were collected repeatedly from CR1 during post-restoration years but not prior to restoration. As a result, taxa richness at CR1 was significantly higher ($p = 0.000$) in 2000, 2001, 2005, and 2007 than pre-restoration levels for both sampling seasons (Tables 6 and 7). EPT richness increased significantly from pre-restoration in May 2005 and 2007 ($p = 0.001$) and August/September 2000, while diversity increased significantly for both seasons in 2007 (Tables 6 and 7). The other sites (2 grazed and 1 ungrazed but upstream of treatment area) did not show any trends in significant community metric responses following restoration for both May and August/September periods (Tables 6 and 7).

Table 6. Mean macroinvertebrate taxa richness, EPT richness, Shannon diveristy index (H'), and SE for May samples collected from Cedar Run, Slab Cabin Run, and Spring Creek in select years from 1992 through 2007. Bold, italicized values represent significant differences from pre-restoration values (α<0.05).

MAY

Stream (Site)	*Macroinvertebrate Metric*	*1992*	*2001*	*2002*	*2005*	*2007*	*Stream (Site)*	*Macroinvertebrate Metric*	*1992*	*2001*	*2002*	*2005*	*2007*
Cedar Run	Taxa Richness	13.33	***20.00***	15.00	17.43	19.27	Slab Cabin Run	Taxa Richness	12.00	14.67	9.33	***17.25***	***16.95***
	SE	1.170	2.120	1.410	3.040	2.170		SE	0.913	2.910	3.180	0.494	0.380
	EPT Richness	6.42	7.00	5.50	8.50	8.00		EPT Richness	3.50	2.67	1.67	***7.75***	5.32
	SE	0.825	1.080	1.320	2.510	1.610		SE	0.645	1.200	1.200	0.441	0.766
	Diversity (*H'*)	1.78	1.98	1.81	1.69	1.67		Diversity (*H'*)	1.32	1.14	1.04	1.60	1.41
	SE	0.121	0.064	0.174	0.265	0.240		SE	0.126	0.280	0.309	0.122	0.099
(CR1)	Taxa Richness	11.67	***24.33***	17	***23.33***	***25***	(SL1)	Taxa Richness	10.00			***17.67***	***16.33***
	SE	1.86	0.882	1.73	0.667	0.577		SE	1.00			0.333	2.19
	EPT Richness	7.67	7.67	7.67	***12.67***	***12***		EPT Richness	2.67			***9.00***	***5.67***
	SE	0.667	0.667	0.882	0.333	0		SE	0.667			0.577	0.882
	Diversity (*H'*)	1.68	2.11	2.03	2.04	***2.28***		Diversity (*H'*)	1.00			1.46	1.32
	SE	0.206	0.0467	0.0889	0.0608	0.0656		SE	0.095			0.0872	0.157
(CR2)	Taxa Richness	14	***22.67***	14.67	12.67	19.67	(SL2)	Taxa Richness	10.67	13.67	8.67	18.33	17
	SE	2.52	2.33	1.45	0.882	1.2		SE	2.33	5.17	2.19	1.76	0.577
	EPT Richness	4.67	9	5	4	8.67		EPT Richness	2.33	2.33	0	***7.67***	3.33
	SE	1.2	1.53	0	0.577	1.2		SE	0.882	0.882	0	1.45	0.333
	Diversity (*H'*)	1.92	1.95	1.69	1.32	1.37		Diversity (*H'*)	1.38	1.03	0.993	1.35	1.17
	SE	0.026	0.0473	0.15	0.0933	0.127		SE	0.231	0.171	0.0437	0.0611	0.192
(CR3)	Taxa Richness	16.33	18.33	17.33	22	17.67	(SL3)	Taxa Richness	12.67	***19.67***	3.5	16	16.33
	SE	2.19	1.33	0.667	1.73	0.882		SE	2.67	0.333	1.5	1.73	0.667
	EPT Richness	7.67	7.33	6.67	13	7		EPT Richness	4.00	1.00	0.50	7.00	5.33
	SE	1.76	0.333	0.333	1.53	0.577		SE	1.15	0.577	0.5	1.00	1.20
	Diversity (*H'*)	2.02	2.06	2.15	2.24	1.81		Diversity (*H'*)	1.28	0.717	0.51	***1.89***	1.56
	SE	0.07	0.0593	0.0463	0.0338	0.223		SE	0.104	0.0788	0.33	0.0203	0.114
(CR4)	Taxa Richness	14.33	15	11	11.67	14.67	(SL4)	Taxa Richness	13.67	16.00	14.67	17.00	18.00
	SE	2.96	2.08	2.52	1.33	0.333		SE	2.33	2.31	1.33	0.577	1.53
	EPT Richness	5.33	4.33	2.33	4.33	4.33		EPT Richness	5.00	5.33	4.33	7.33	7.00
	SE	2.03	0.333	0.333	0.667	0.333		SE	0.577	0.882	0.333	0.333	0.577
	Diversity (*H'*)	1.48	1.82	1.38	1.17	1.22		Diversity (*H'*)	1.61	1.67	1.59	1.71	1.58
	SE	0.118	0.0437	0.224	0.131	0.0698		SE	0.0521	0.0985	0.105	0.0788	0.0426
Spring Creek	Taxa Richness	20.67	23.33	16.33	17.67	18							
	SE	1.45	2.91	2.19	0.333	2.65							
	EPT Richness	9.33	11.33	6	10.67	7.33							
	SE	0.882	1.2	1.53	0.333	1.67							
	Diversity (*H'*)	1.29	1.76	1.95	1.58	2.32							
	SE	0.105	0.093	0.155	0.11	0.0764							

Similar to the density data, metric responses at Slab Cabin Run were not significant until the latter post-restoration years and, like Cedar Run, only occurred at the downstream site. At Slab Cabin Run the farthest downstream site (grazed SL1) also had significantly higher taxa richness and EPT richness from pre-restoration levels in May 2005, 2007 and August 2005 (taxa richness only) (Tables 6 and 7). This was due primarily to certain taxa [some mayfly genera (*Ephemerella, Epeorus)*, caddisfly genera (*Cheumatopsyche)*, riffle beetles (*Optioservus, Promoresia)*, and snails (Gastropoda)] that were absent in pre-restoration samples but present in the latter post-restoration years. The other grazed site along Slab Cabin Run (SL3) had taxa richness values significantly higher than pre-restoration values in May

2001 but not in proceeding post-restoration years (Table 6). It is important to note, however, that fencing at the SL3 site was not maintained during the latter post-restoration period. No other significant trends in community metric changes from pre-restoration levels were detected at the Slab Cabin Run sites.

Table 7. Mean macroinvertevrate taxa richness, EPT richness, Shannon diversity index (H'), and SE for Fall samples collected from Cedar Run, Slab Cabin Run, and Spring Creek in select years from 1992 through 2007. Bold, italicized values represent significant differences from pre-restoration values (α<0.05).

				FALL									
Stream (Site)	*Macroinvertebrate Metric*	*1992*	*2000*	*2001*	*2005*	*2007*	*Stream (Site)*	*Macroinvertebrate Metric*	*1992*	*2000*	*2001*	*2005*	*2007*
Cedar Run	Taxa Richness	13.33	16.67	***17.58***	16.92	***18.67***	Slab Cabin Run	Taxa Richness	13.08	14.22	15.22	***17.17***	15.83
	SE	0.678	1.340	0.965	1.150	0.873		SE	0.668	1.230	1.470	0.638	0.454
	EPT Richness	4.00	4.67	5.08	5.17	6.17		EPT Richness	3.67	3.33	3.56	5.08	4.50
	SE	0.326	0.940	0.753	0.842	0.613		SE	0.632	0.913	0.530	0.452	0.428
	Diversity (*H'*)	1.60	1.68	1.54	1.36	1.62		Diversity (*H'*)	1.62	1.44	1.36	1.30	1.72
	SE	0.061	0.109	0.131	0.163	0.116		SE	0.044	0.104	0.075	0.096	0.111
(CR1)	Taxa Richness	10.67	***22.33***	***19.33***	17.33	***19.33***	(SL1)	Taxa Richness	11.67	*	*	***18.67***	*
	SE	0.882	2.330	2.030	0.882	1.670		SE	1.670	*	*	0.882	*
	EPT Richness	3.00	***8.67***	7.33	5.00	7.67		EPT Richness	4.00	*	*	5.33	*
	SE	0.577	1.760	1.760	0.577	0.882		SE	1.000	*	*	0.667	*
	Diversity (*H'*)	1.63	***2.21***	1.40	1.87	***2.08***		Diversity (*H'*)	1.73	*	*	1.23	*
	SE	0.139	0.123	0.127	0.007	0.003		SE	0.102	*	*	0.240	*
(CR2)	Taxa Richness	14.33	15.33	18.00	17.00	17.00	(SL2)	Taxa Richness	13.00	10.67	12.67	15.33	*
	SE	1.450	2.030	1.000	0.577	1.530		SE	1.150	1.450	3.180	0.882	*
	EPT Richness	4.00	4.67	6.00	4.67	6.00		EPT Richness	2.00	0.33	2.33	3.33	*
	SE	0.577	1.450	1.000	0.333	1.730		SE	0.000	0.333	0.882	0.333	*
	Diversity (*H'*)	1.67	1.56	1.48	0.97	1.21		Diversity (*H'*)	1.58	1.17	1.16	1.15	*
	SE	0.091	0.055	0.385	0.348	0.139		SE	0.023	0.078	0.080	0.054	*
(CR3)	Taxa Richness	15.00	12.67	19.67	21.33	17.00	(SL3)	Taxa Richness	14.33	16.00	14.33	18.67	15.33
	SE	1.150	1.760	1.330	2.190	1.530		SE	0.882	1.530	2.400	1.330	1.330
	EPT Richness	5.00	1.67	5.00	9.00	6.33		EPT Richness	6.67	5.00	3.00	5.67	4.67
	SE	0.577	0.667	0.577	1.530	1.450		SE	0.333	0.577	0.000	0.882	0.882
	Diversity (*H'*)	1.73	1.33	2.06	1.71	1.73		Diversity (*H'*)	1.61	1.41	1.41	1.48	1.53
	SE	0.112	0.124	0.034	0.203	0.267		SE	0.102	0.189	0.112	0.252	0.131
(CR4)	Taxa Richness	13.33	16.33	13.33	12.00	***21.33***	(SL4)	Taxa Richness	13.33	16.00	18.67	16.00	16.33
	SE	0.667	1.450	0.882	1.150	1.670		SE	1.760	2.080	0.667	1.000	0.3330
	EPT Richness	4.00	3.67	2.00	2.00	4.67		EPT Richness	2.00	4.67	5.33	6.00	4.33
	SE	0.577	0.882	0.577	0.577	0.333		SE	0.577	1.670	0.333	1.000	0.333
	Diversity (*H'*)	1.37	1.61	1.21	0.88	1.48		Diversity (*H'*)	1.54	1.74	1.51	1.35	1.92
	SE	0.064	0.118	0.092	0.201	0.030		SE	0.105	0.067	0.127	0.222	0.0841
Spring Creek	Taxa Richness	16.33	18.67	19.00	*	18.33							
	SE	1.450	1.760	0.577	*	2.960							
	EPT Richness	5.00	7.67	6.00	*	7.33							
	SE	0.577	0.882	0.577	*	1.860							
	Diversity (*H'*)	1.37	1.25	1.93	*	1.46							
	SE	0.067	0.161	0.113	*	0.225							

<u>Fish Communities and Brown Trout Densities:</u> In general, fish communities in the three study streams were rather simple, which is typical of coldwater limestone streams. Spring Creek and Cedar Run supported three species: slimy sculpins, brown trout, and white suckers in descending order of abundance (Table 8). In Slab Cabin Run, which is warmer during summer than the

other streams, we collected a total of 14 species during eight sampling events. Most species were represented by a few specimens, while white suckers, longnose dace, blacknose dace, and slimy sculpins were the most numerous species. There were no apparent changes in fish communities from 1992 through post-construction years in any of the streams.

Density of age-1 and older brown trout at the Spring Creek reference site in May was highest in 1992 and declined during the post construction years, when it ranged from 70 to 83 trout/100 m (Table 9). Trends in brown trout density at all four sites in Cedar Run were markedly different than those in the reference site. In 1992 prior to construction, brown trout density averaged 22/100m among the four survey sites. During the post construction years, 2001-2007, brown trout density was 91% than 1992 density and in three of four sites, density was significantly higher (95% CL did not overlap) in post construction years compared to the pre-construction survey. These data suggest a positive response of age-1 and older brown trout to riparian restoration.

During August trout surveys, we collected age-0 trout at most sites, and estimated their density separately from age-1 and older brown trout. At the Spring Creek reference site, density of age-0 trout varied widely among years, reflecting variable reproductive success. During post-treatment years, density of age-0 trout averaged 32/100 m (Table 10). At the Cedar Run sites, age-0 density ranged from 0 to 95/100m, suggesting substantial variations among sites and among years. During post-treatment years, age-0 density averaged 25/100m, which was about 38% higher than in 1992. Density of age-0 trout in Slab Cabin Run was low among all years. Average density during post-treatment years was 10/100 m, more than twice the density in 1992.

Among all streams, density of age-1 and older trout in August varied less than density of age-0 trout (Table 10). At the Spring Creek reference site, we estimated a density of 69/100m in 1992, and during post-treatment years density averaged 96/100m. Density of age-1 and older trout in Cedar Run in August we generally higher during post-treatment years compared to 1992. Average density was 19/100m in 1992 and average 31/100m in 2000-2007. Density of age-1 and older brown trout in Slab Cabin Run was rather low during pre- and post-treatment years. In 1992 we estimated only 3/100m in 1992 and 6/100m in post-treatment years. Thus, during August, the pattern of brown trout density was similar to that in May. Densities of trout were higher in Cedar Run and Slab Cabin Run after construction, but population response in Slab Cabin Run was rather small.

Table 8. 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 (2000-2007) 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, 2007 and August 1992, 2000, 2001, and 2007.

	1992			2000-2002			2007		
	Spring Creek	Cedar Run	Slab Cabin Run	Spring Creek	Cedar Run	Slab Cabin Run	Spring Creek	Cedar Run	Slab Cabin Run
Brown trout (*Salmo trutta*)	44	10	1	38	13	1	51	14	4
Common shiner (*Luxilus cornutus*)	0	0	<1	0	0	0	0	0	0
Pearl dace (*Margariscus margarita*)	0	0	<1	0	0	0	0	0	4
Fathead minnow (*Pimephales promelas*)	0	0	7	0	0	9	0	0	6
Blacknose dace (*Rhinichthys atratulus*)	0	0	3	0	0	2	0	0	23
Longnose dace (*Rhinichthys cataractae*)	0	0	30	0	0	1	0	0	2
Creek chub (*Semotilus atromaculatus*)	0	0	2	0	0	0	0	0	3
Fallfish (*Semotilus corporalis*)	0	0	<1	0	0	0	0	0	0
White sucker (*Catostomus commersoni*)	2	5	14	<1	4	4	3	6	42
Banded killfish (*Fundulus diaphanu* s)	0	0	3	0	0	1	0	0	1
Pumpkinseed (*Lepomis gibbosus*)	0	0	0	0	0	0	0	0	<1
Bluegill (*Lepomis macrochirus*)	0	<1	0	0	0	<1	0	0	<1
Tessellated darter (*Etheostoma olmstedi*)	0	0	1	0	0	1	0	0	4
Slimy sculpin (*Cottus cognatus*)	76	78	8	28	43	16	46	80	11

Table 9. Estimated 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 (2001-2007) 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	2005	2007
Spring Creek					
SP1	106 (106-126)	77 (67-87)	70 (67-74)	83 (74-92)	68 (67-71)
Cedar Run					
CR1	29 (29-30)	46 (45-47)	50 (49-52)	66 (63-70)	66 (59-75)
CR2	34 (34-35)	74 (66-82)	41 (40-43)	56 (51-63)	62 (48-80)
CR3	16 (16-18)	33 (31-38)	21 (19-24)	53 (53-55)	41 (40-44)
CR4	8 (8-9)	24 (20-31.5)	15 (15-15)	8 (8-9)	11 (11-14)
Slab Cabin Run					
SL1	1 (1-2)	0	0	11 (11-13)	2
SL2	2 (2-3)	3	1	14 (14-14)	6 (6-8)
SL3	1 (1-2)	2	3	35 (35-36)	4 (4-5)
SL4	3 (3-5)	7	8 (8-9)	29 (29-31)	24 (24-25)

Table 10. 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 (2000-2007) 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	2005	2007
Spring Creek						
SP1	0	197 (39-1009)	18 (18-19)	7 (17-19)	3	90 (25-352)
	1+	69 (59-187)	70 (66-75)	104 (103-107)	150 (148-153)	89 (86-93)
Cedar Run						
CR1	0	22 (14-43)	4	39 (31-51)	23 (a)	22 (19-27)
	1+	14 (14-15)	14 (14-14)	22 (22-23)	63 (63-64)	35 (35-36)
CR2	0	22 (16-50)	14 (14-20)	95 (19-558)	20 (20-23)	2
	1+	44 (43-48)	10	25 (25-25)	81 (81-82)	86 (83-90)
CR3	0	26 (24-30)	63 (62-65)	43 (41-45)	12 (9-21)	25 (25-27)
	1+	13 (13-14)	14 (14-15)	7	63 (60-66)	28 (28-28)
CR4	0	2 (2-3)	29 (29-31)	12 (12-13)	35 (16-114)	1
	1+	6 (6-7)	10 (10-11)	13 (13-13)	23 (23-24)	5
Slab Cabin Run						
SL1	0	0	23 (19-31)	Dry Channel	3	1
	1+	1 (1-2)	1		23 (23-24)	0
SL2	0	0	0	0	0	0
	1+	2 (2-3)	1	0	4	0
SL3	0	0	1	6	5 (5-7)	4
	1+	1 (1-2)	2	3	13 (13-13)	1
SL4	0	14 (14-19)	9 (9-10)	8	37 (a)	60 (53-71)
	1+	6 (6-8)	7 (7-7)	3	26 (26-27)	18 (18-20)

Part II: Assessment of Cross Vanes in Slab Cabin Run

Background

Main objective:

- *Elevate the stream surface of Slab Cabin Run to promote flooding of the wetland during storm flow and filter suspended pollutants.*

We capitalized on the opportunity to assess the efficacy of rock cross vanes to improve the connection of storm flows to adjacent wetlands, thereby promoting the filtering of pollutants by wetland vegetation. The use of cross vanes to reduce non-point source pollution is a new innovation that may be applicable to some agricultural areas. Cross vanes are V-shaped structures constructed of either large rocks or logs (Figure 10). The point of the V is oriented upstream and is lowest in elevation compared to the downstream base. Cross vanes divert stream flow to the middle of the channel and impound the channel, the degree to which is determined by the elevation of the base of the structure. Downcut stream channels are common in agricultural wetlands. If cross vanes are effective in flooding wetlands and reducing pollutant loads, they may represent another BMP that can be employed where circumstances permit.

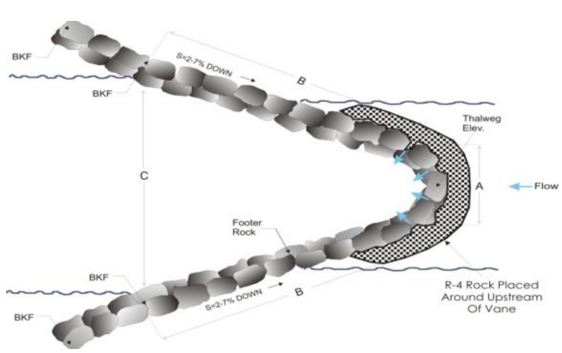


Figure 10. Diagram and photo of rock cross vane in Millbrook Marsh. Courtesy of T. Rightnour.

Methods

Eight cross vanes were installed in 2007 in a section of Slab Cabin Run that flows through Millbrook Marsh (Figure 11). This section of stream has been downcut, because stream length was shortened during road construction. Shortened stream length produced higher stream velocities, which produced the downcut. The channel now has an increased volume, owing to the downcut, and during storm flow the stream is less likely to flood the adjacent wetlands. The purpose of the cross vanes was to elevate the stream surface and promote flooding of the adjacent wetland during storm flow, which would presumably lead to filtering of suspended pollutants.

We deployed Hydrolab DS5X (Hach Co.) data sondes at sites upstream and downstream of three cross vanes in Slab Cabin Run (Figure 11). These *in situ* monitors were equipped to measure several variables including pH, temperature, dissolved oxygen and turbidity. They were programmed to measure selected variables at 15-min intervals. Monitors were deployed from September 1, 2009 to November 4, 2009, and data were downloaded at weekly intervals. Stream flow during this time period was never high enough to flood the wetland. Rather than continuing to monitor at base flow, we decided to closely watch weather reports and only deploy monitors when it seemed likely flows would be high enough to flood the wetlands.

Figure 11. Map of Millbrook Marsh showing locations of cross vanes and monitoring sites. Courtesy of T. Rightnour.

Rainfall on January 24, 2010 combined with melting of existing snowpack caused stream flows to rise rapidly. During the morning of January 25, 2010, we deployed the data sondes in Slab Cabin Run, which was bankfull at the time. By 1700 h the following day, turbidity had returned to near normal levels. We again deployed the data sondes on March 11, 2010, in response to a predicted large storm, which began early on March 13. Stream flow began increasing around 1000 h on March 13, peaked around 0400 h on March 14; turbidity returned to normal levels by 1700 h on March 14.

The Spring Creek Watershed Association has a newly installed stream gaging station where we deployed the downstream data sonde. A rating curve has not yet been developed for this site; hence we had to estimate stream discharge from the USGS gaging station on Spring Creek at Houserville. This gaging station is about 2.4 km downstream of our Millbrook Marsh sites. Measured stream discharge at the Millbrook Marsh site in November 2009 and February 2010 averaged 25.3% of the discharge at the Houserville gage on those dates. We downloaded discharge data (15-min intervals) from the Houserville gage for the January and March storm events and estimated discharge at Millbrook Marsh using the 25.3% conversion.

We converted the turbidity (NTU) measurements from the data sondes to total suspended solids (TSS) using the formula:

$$TSS = 0.0011*NTU^2 + 1.127*NTU,$$

which was developed from a study by Carline et al. (2003) on Spring Creek in the vicinity of Millbrook Marsh. This regression was highly significant ($R^2 = 0.96$). We then used estimated stream discharge and TSS to compute sediment load for each 15-min interval during the two monitored storm events.

Results

Judging from flow records from the USGS gage at Houserville and assuming 1-h travel time from Millbrook Marsh to the USGS gage, it seems that the data sondes were deployed in Millbrook Marsh at the time of peak flow for the January storm. Therefore, our data record spans the descending limb of the hydrograph. At the upstream site, turbidity peaked at 471 NTU at 1200 h, while at the downstream site turbidity peaked about the same time but at a substantially lower level of 337 NTU (Table 11 & Figure 12). Peak sediment load at the upstream site was 2,558 kg/15 min, compared to 1,684 kg/15 min at the downstream site. During the 18 h for which we estimated sediment load, total sediment load at the upstream site was nearly 55,000 kg compared to about 44,000 kg at the downstream site. These data suggest the three cross vanes between the two sampling sites reduced sediment load by 19.2%.

Table 11. Turbidity and computed sediment load at sites upstream and downstream of cross vanes in Slab Cabin Run for storms on January 25-56, 2010 and March 13-14, 2010.

Storm	Elapsed time (h)	Location	Median turbidity (NTU)	Peak turbidity (NTU)	Peak sediment load (kg/15 min)	Total sediment load (kg)
JAN	18.0	Upstream	132	471	2,558	54,865
		Downstream	120	337	1,684	44,322
MAR	31.25	Upstream	100	196	691	42,826
		Downstream	85	178	684	40,623

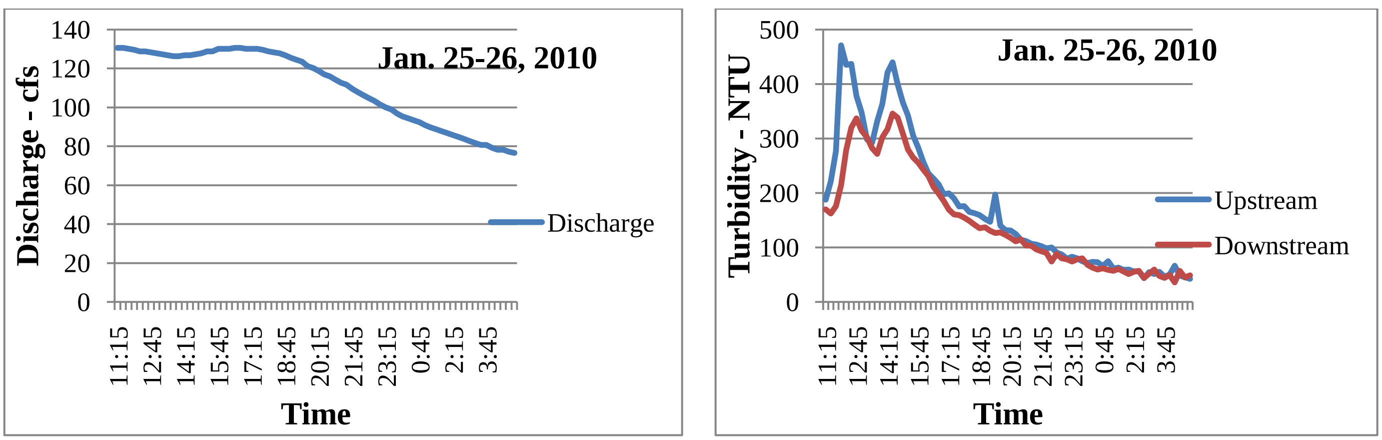
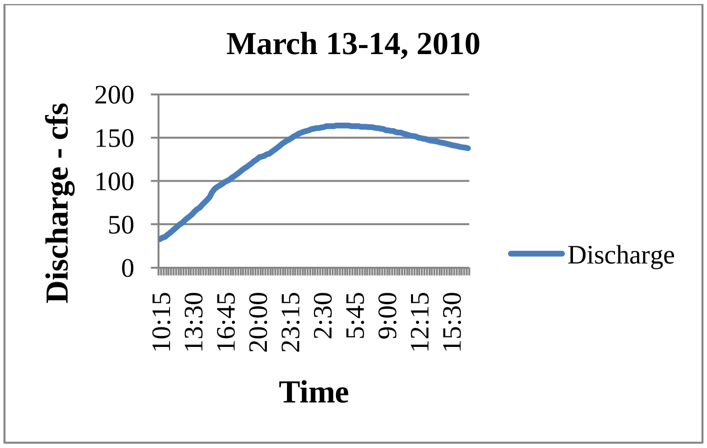


Figure 12. Estimated discharges in Slab Cabin Run and turbidity at monitoring stations upstream and downstream of cross vanes, January 25-26, 2010.

The March 2010 storm produced a maximum discharge of approximately 164 cfs in Slab Cabin Run, about 23% higher than peak flow during the January storm (Figure 13). Peak and median turbidity at the upstream and downstream sites were substantially less than during the January storm (Table 12). As a result, total sediment load during the March storm was nearly

43,000 kg over 31 h at the upstream site and about 5% less than the downstream site. Here again, the cross vanes seem to have reduced sediment load, but at a modest level.



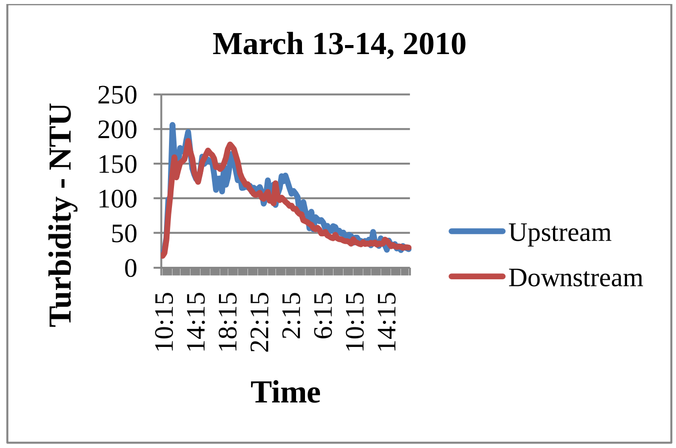


Figure 13. Estimated discharge in Slab Cabin Run and turbidity at monitoring sites upstream and downstream of cross vanes, March 13-14, 2010.

The difference in turbidity between the two storms was largely responsible for the higher sediment load in January. One possible reason for the higher turbidity in January was that there was substantial amount of fine sediment stored in the stream channel, and much of this stored sediment was transported downstream during the January storm. Between January and March it is likely that sediment once again began to accumulate in the stream channel, but the accumulation was much less than that prior to the January storm. Hence, there was less sediment available for transport during the March storm.

Regardless of the reason(s) for differences in sediment loads between storms, we believe these data provide convincing evidence that the cross vanes were indeed functioning as planned. Given that there are a total of eight cross vanes in Millbrook Marsh, it is likely that sediment reduction during these two storms was substantially greater than that which we documented.

Conclusions: Ground-based Monitoring and Ecological Anlayses

Ecological Messages (◊) & Important Findings (•)

◊ Specify which onsite stressors the BMP(s) will address and monitor the BMP with the appropriate indicator:

- Narrow grass buffers, stream crossings, and bank stabilizations were effective in reducing sediment loads in Cedar Run and Slab Cabin Run but did not reduce nutrients; however, these BMPs were directed at sediment reduction, not nutrient reduction.
- Direct measurements of the stressor (% fines in stream substrates) and biological density metrics demonstrated more obvious and interpretable responses than biological composition metrics.

◊ Be aware of the limitations of the BMP:

- Reductions in fine sediment produced a response in the existing community (i.e., increased density) but did not result in strong increases in richness or diversity, most likely because other community stressors were not addressed.

◊ Identify the hydrologic nature of the stream and allow sufficient monitoring time to account for responses to hydrologic fluctuations and other stressors:

- Indicators displayed delayed responses to BMPs due to drought years, especially in Slab Cabin Run, where during dry periods flow percolates through the streambed, often resulting in a dry channel.
- Monitoring may require more than ten years to effectively rule out climate cycles.

◊ Cross vanes are effective in flooding wetlands and reducing pollutant loads in downcut streams and may represent another BMP that can be employed where circumstances permit.

LANDSCAPE CHARACTERIZATION AND COARSE- VS. FINE-GRAINED ASSESSMENT*

* This section is intended for future publication. Additional information will be made available in the following manuscript: *Assessing performance of pasture-based Best Management Practices: coarse- vs. fine-scale analysis*

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Background

Main objectives:

- *Summarize land use within a 100-m buffer on either side of the stream, and locate agriculturally-based heavy use areas using high-resolution aerial photography;*
- *Evaluate the effect of DEM resolution on flow path calculations by comparing coarse- (30 m), medium- (10 m), and fine-grained (1 m) elevation maps with each other and the straight-line distance.*

Impaired streams are typically treated, in part, by placing stream-bank Best Management Practices (BMPs) down slope from any nearby animal heavy use areas. However, current NRCS guidelines for such BMP placements are very general and, if applied primarily through visual inspection, may not insure that the chosen BMPs are optimally sized or selected to account for the flow distance from the heavy use area to the stream or are optimally placed where that flow will enter the stream (USDA-NRCS, 2010).

Procedure

High-resolution aerial photography was used to digitize and identify land use and animal heavy use areas within 100 m of the visible stream for three subwatersheds of Spring Creek watershed in Centre County, PA: Slab Cabin, Cedar Run, and Upper Spring Creek. The distance from each heavy use area to the stream was determined using two metrics: “as the crow flies” to produce the straight path, and the topographically-based flow path as identified by a coarse- (30 m), intermediate- (10 m), and fine-scale (1 m) DEM (Figures 14 and 15). The adjusted difference in distance of the straight path from the topographically-based flow path for each DEM resolution was calculated as [100 * (straight path distance – topographically-based distance) / straight path distance)]. Additionally, for all heavy use points, the stream offset for each pair of straight path and topographically-based stream entry points was calculated as the difference in along-stream distances.

Results

Agricultural land use within the riparian buffer was highest in the watersheds with BMPs (Cedar Run 46%, Slab Cabin Run 37%) and lowest in the Upper Spring Creek watershed (17%) (Table 12). The BMP-treated watersheds had lower percentages of forest land use, but all three watersheds had similar percentages of residential and developed land (Table 12). Across all watersheds, topographically-based flow paths were substantially longer than straight paths with a median value of 19 m longer than that of the straight path for both the 1-m (71% longer) and 10-m (49% longer) DEMs, and 48 m longer for the 30-m DEM (91% longer; Figure 16). The stream offsets were also considerably different, with median differences of 281 m for 1-m DEM, 256 m for 10-m DEM, and 1198 m for the 30-m DEM (Figure 17).

Table 12. Summary of riparian land use for the Spring Creek watershed.

Sub-Watershed	% Stream Treated (BMPs)	Size (total ha)	% Agricultural Land Use	% Forest Land Use	% Residential/ Commercial
Spring Creek	0	653	17	45	38
Cedar Run	98	699	46	19	35
Slab Cabin Run	67	625	37	36	27

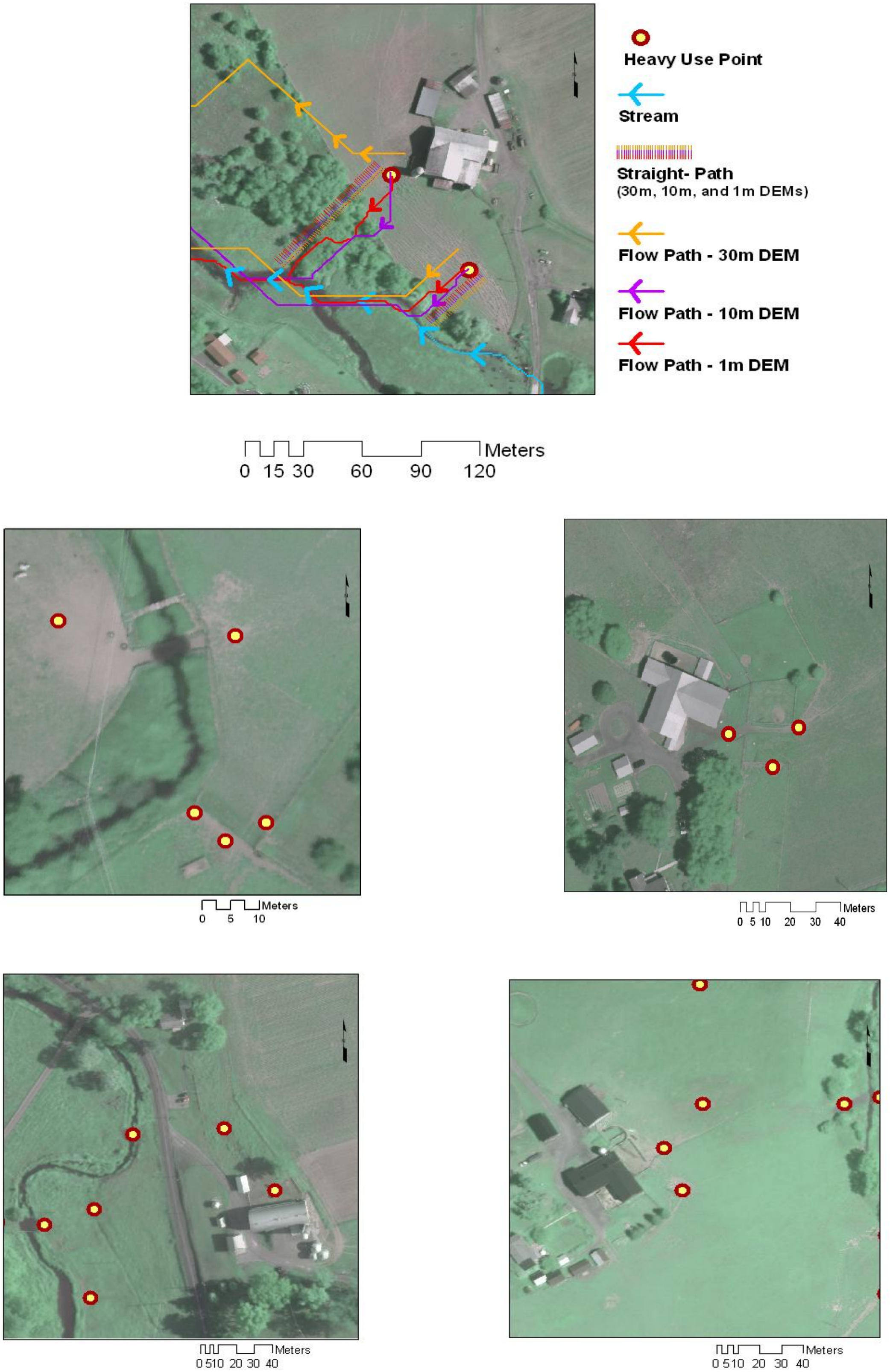

Figure 14. High resolution aerial photography of animal heavy-use areas (points) in the Spring Creek watershed.

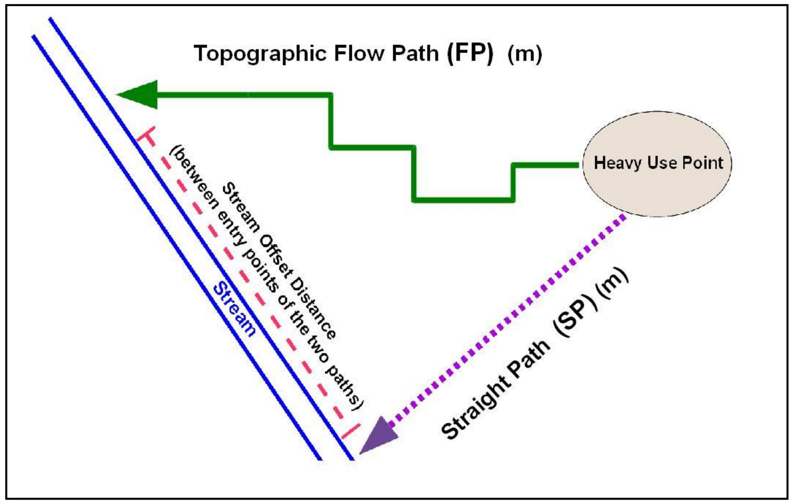

Figure 15. Diagram of straight path, topographically-based flow path and stream offset distance.

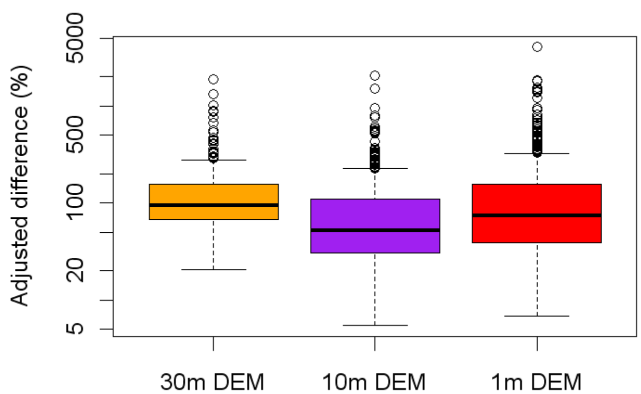

Figure 16. Adjusted difference (%) in length of flow path and straight path from each heavy-use point to the stream.

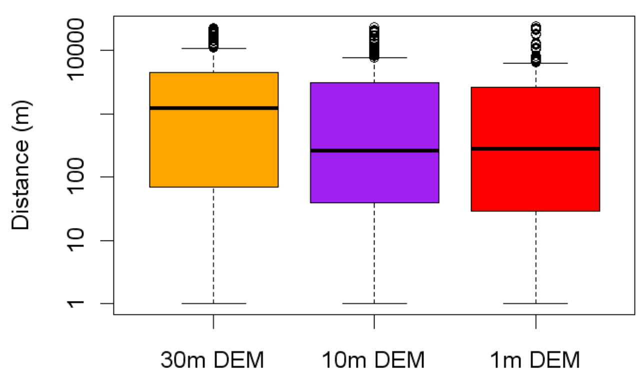

Figure 17. Stream offset (m) between the flow and straight path entry points for each heavy-use point.

Conclusions: Landscape Characterization and Coarse- vs. Fine-grained Assessment

Coarse- vs. Fine-Grained Messages (◊) & Important Findings (•)

◊ Current NRCS standards do not include recommendations to examine actual flow path from the heavy use area to the stream when determining related BMP placements.

◊ Visual assessments and straight-line paths from heavy use areas to streams often give misleading estimates of flow path lengths and stream entry points, resulting in inefficiently placed BMPs.

- Correct flow paths are needed to accurately estimate nutrient and sediment loadings and concentrations into the stream.
- The actual point where water flowing from a heavy use area enters the stream may be nowhere near the straight-line entry point, thus bypassing the BMP entirely.

◊ When determining BMP placement for a particular site, the 10-m DEM appears to be adequate. While 1-m DEM data provides more accurate results, these data are not widely available and require more intensive processing.

HYDROLOGIC AND LANDSCAPE MODELING OF BMP PERFORMANCE*

* This section is intended for journal publication. Further information will be made available in the following manuscripts:
(1) Analyzing high resolution topographic data to infer denitrification potential in riparian zones.
(2) Using the Sednet model with high resolution topographic data to estimate the amount and sources of stream sediment.
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This section is composed of two parts: I) a landscape characterization evaluating the effect of topographic resolution on parameter estimates related to sediment transport and riparian buffers; and II) an examination of the utility of various simulation models for estimating the relative importance of upland, riparian, and in-stream sources on sediment loads.

Part I: Landscape Characterization

Background

Main Objective:

- *Examine the effect of topographic data resolution on (1) estimates of stream corridor characteristics related to sediment transport and (2) watershed metrics related to riparian buffers*

We examined how the resolution of topographic data affects estimates of stream corridor characteristics related to sediment transport. We used digital elevation models at three different resolutions (30 m, 10 m, and 1 m horizontal resolution) to develop stream network maps using standard geographic information system (GIS) algorithms. We also tested the highest resolution data (derived from LiDAR remote sensing with 15 cm vertical resolution) to evaluate its potential to capture stream incision and to identify floodplain areas only slightly elevated above the stream channel. Such areas are likely to receive sediment deposits during high stream flows, and they may also tend to have saturated soils that support nitrogen removal through denitrification.

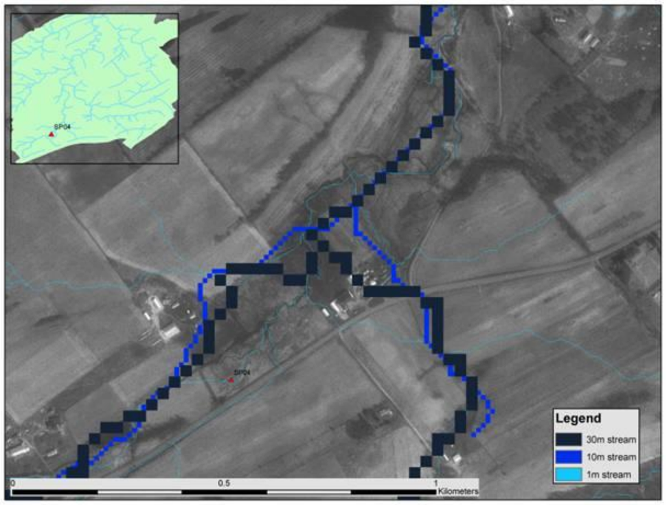
We also tested how the resolution of topographic data affects watershed metrics of the prevalence of riparian buffers. The metric calculation uses topographic data to identify the steepest surface transport pathway from every cropland pixel to a stream. Then, we measure the width of riparian buffer for every crop-to-stream pathway (Baker et al. 2006). For each watershed, we estimate mean buffer width by averaging across all the cropland to stream flow paths in the watershed. We also estimated the frequency of gaps as the percentage of cropland pixels whose flow paths do not pass through a buffer (Weller et al. 1998, Baker et al. 2006a). In a previous analysis, adding unbuffered cropland to a statistical model predicting nitrate concentration from land cover proportions gave an improved model that demonstrated and quantified the effects of riparian buffers on stream nutrient concentrations discharged from watersheds (Weller et al. in review). To explore the effects of data resolution, we calculated the buffer metrics using topographic data from coarse, intermediate, and fine resolution digital elevation models (30-m, 10-m, and 1-m horizontal resolution). At each resolution, the topographic data were used both to map the stream network and to calculate the buffer metrics.

Results

The following results were indicated:

- Finer resolution topographic data yield more accurate stream maps that better define stream channel location, length, and sinuosity than maps constructed from coarser topographic data (Figure 18).
- High-resolution topographic data from LiDAR remote sensing support detailed maps of stream and riparian characteristics, such as the stream cross-section and floodplain area. Coarser resolution topographic data are not precise enough to calculate such measures.
- The LiDAR-based maps of stream and riparian characteristics help identify riparian buffers and provide indicators of characteristics and processes (such as stream incision, hillslope discharge, and floodplain deposition) that control how buffers affect stream water quality.
- Less than 4% of the riparian zone (defined as the area within 1.5 m elevation of the adjacent stream channel) is forest or wetland (Figure 19).

- The effects of topographic resolution on watershed-average buffer potentials remain unclear; but at the scale of individual stream reaches, maps based on coarse resolution data can clearly provide misleading information on the positioning of buffers between oplands and streams.
- LiDAR data provide the topographic detail needed by stream simulation models to estimate gully erosion, streambank erosion, and floodplain deposition (Figure 18).



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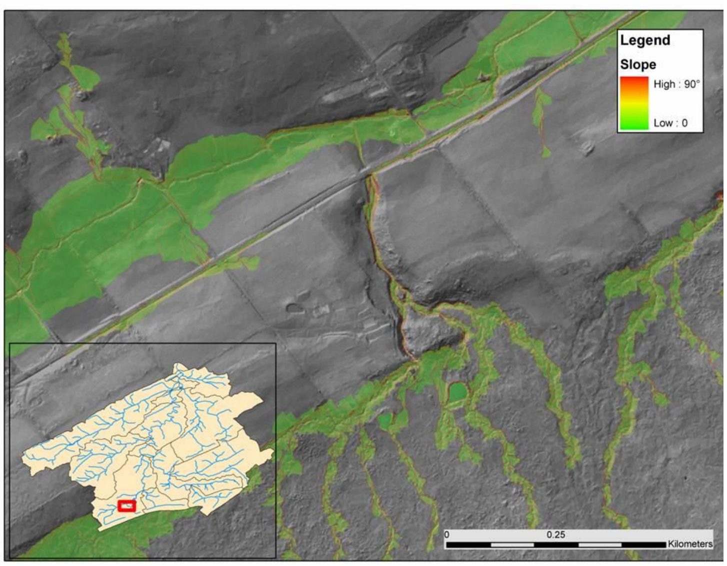


B)

Figure 18. A) Comparison of stream networks derived from 1-m, 10-m, and 30-m DEM data in the upper Spring Creek watershed; B) slope estimates in mapped floodplains derived from 1-m LiDAR data. Results highlight incised channels where stream bank erosion is more likely to occur, and broad floodplain areas where sediment deposition is likely to occur during overbank flow conditions.

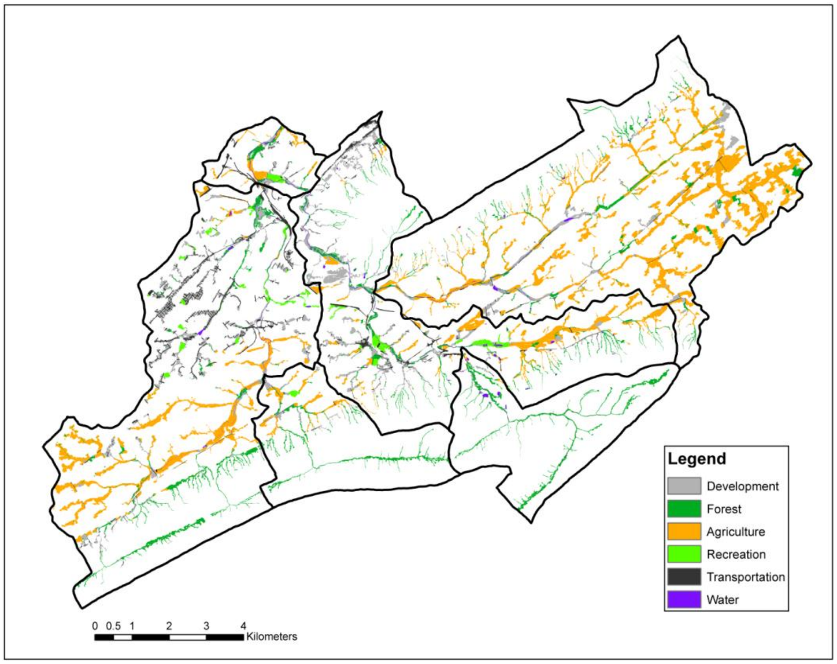


Figure 19. 1993 land cover in riparian floodplain areas along the Upper Spring Creek.

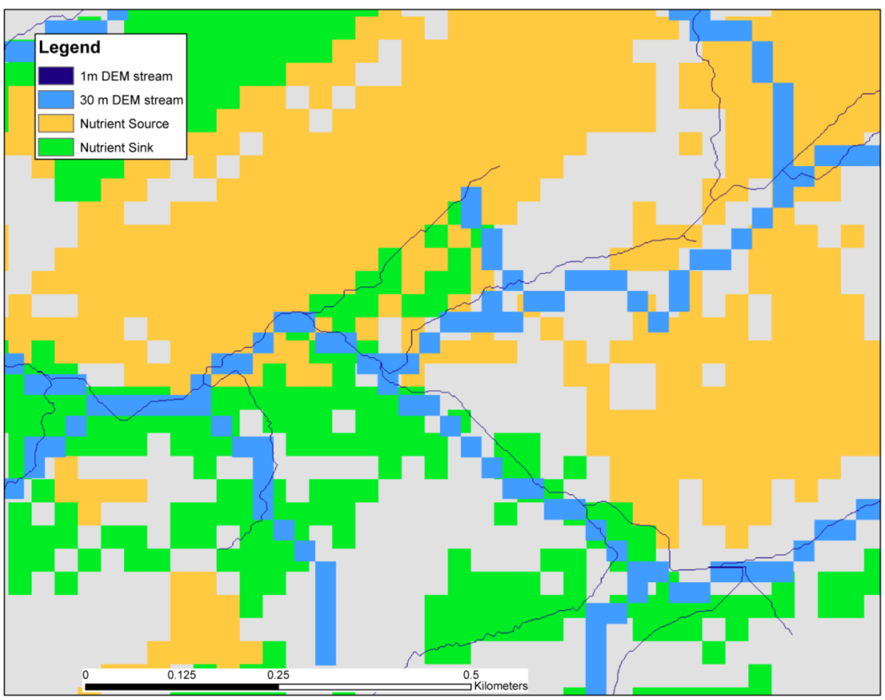


Figure 20. Example of how stream maps influence buffer width estimates. Differences affect the spatial relationship between the stream network and land cover data. Agricultural land use was identified as a nutrient source; forest and wetlands were identified as potential sinks. Differences among the DEM resolutions were more significant where the coarser (30-m) stream network was not field or photo-corrected using a GIS-based stream burning algorithm.

Part II: Hydrologic and Landscape Modeling

Background

Main Objectives:

- *Estimate the relative importance of hillslope, gully, stream bank erosion and floodplain deposition to stream sediment loads;*
- *Compare results from the Sednet model to observed sediment loads and SWAT model predictions.*

We analyzed a set of simulation models to estimate the relative importance of hillslope versus gully and stream bank erosion, and also floodplain deposition to stream sediment loads. We evaluated the capability of each model to capture effects from best management practices on sediment loads. First, we used the Soil Water Assessment Tool (SWAT; Arnold and Allen 1992), which frequently is used to predict flow sediment and nutrient discharge from agricultural watersheds. We developed the SWAT model using land use data derived from 1993 aerial photography, 10 m digital elevation data, and the STATSGO soils database. The model was calibrated with flow observations at the Spring Creek Houserville gauge station (USGS station 01546400).

A 20-ha threshold was used to define the upstream ends of the stream network, resulting in 386 SWAT subbasins (Figure 21). The hydrologic response of each SWAT sub-basin was based on the dominant land cover, soil type, and slope class. Impacts of land use change were evaluated by comparing SWAT outputs for 1993, 2000, and 2006 land use data. We also simulated two hypothetical extreme land use conditions: all croplands in the 1993 land use dataset were converted either to forest or developed land. The differences in discharge expected from land use change were compared to the errors between predicted and observed discharges in the model calibration.

We also compared results from the Sednet model to observe sediment loads and SWAT model predictions. The Sednet model explicitly estimates gully and stream bank erosion and floodplain deposition in addition to hillslope erosion (Prosser 2001), whereas the SWAT model assumes hillslope erosion primarily contributes to sediment loads. We used fine-scale topography data to map stream channels, identify channel incision and riparian or floodplain areas, and estimate stream corridor dimensions along the entire stream network. The stream corridor was defined as areas adjacent to streams and within 1.5 m elevation of the stream (Murphy et al. 2008). Stream incised channels, riparian and floodplain areas, and open water were identified based on the topographic slope of cross-sectional flowpaths along the resulting stream corridor (Figure 22). Sediment loads derived from gully and bank erosion were determined from estimates of bankfull cross-section dimensions, reach slope and length, and riparian vegetation cover. Riparian condition was characterized from land use data. We used empirical models developed for Central Pennsylvania to estimate bankfull flow discharge for each stream reach based on watershed area and land cover (Stuckey 2006).

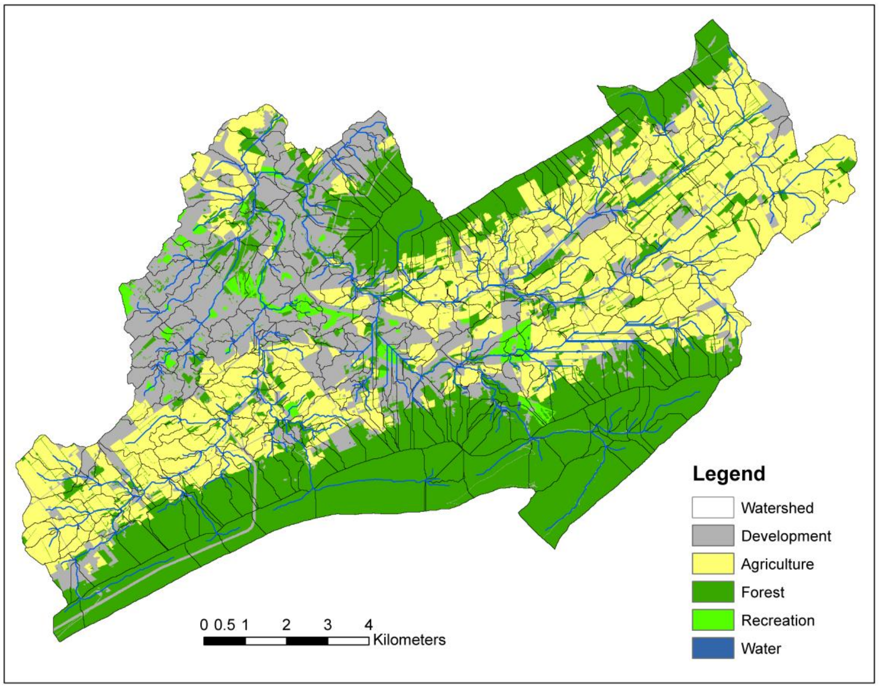
Results

Evaluation and comparison of various simulation models revealed the following results:

- A SWAT model with two slope classes and a low runoff factor for all land uses gave the best calibration to flow measurements from the Spring Creek Houserville gauge (Nash-Sutcliffe (NS) coefficient: 0.63). Adding a third slope class and using distinct runoff potentials for different land uses gave a weak model (NS:-0.7). Replacing the default crops with corn and cold season grasses appropriate for Spring Creek improved that model (NS = 0.46), but not enough to match the initial calibration (NS=0.63).
- Massive hypothetical land use changes (converting all cropland to either forest or developed land) changed stream flow predictions of the SWAT model in the expected directions, but the changes were small relative to the prediction errors of the calibrated model. Land use scenarios restricting changes to the riparian zone (where riparian BMPs could be implemented) produced much smaller changes in predicted discharges (Figure 23).
- Sednet estimates of bankfull discharge, which were based on USGS empirical data, generally exceeded SWAT estimates by more than 100%. Sediment yields, however, were less than 50 percent of the SWAT predictions. At the Spring Creek Houserville gauge, the predicted annual average sediment load is approximately six thousand Mg/yr, whereas only 1,100 Mg were observed during the 1992 adjusted water year (Sept 1, 1991 through August 31, 1992), a relatively dry year.

The model suggested that incised stream reaches, which resulted in higher stream power, and limited floodplain deposition primarily contributed to the predicted loads (Figures 23 and 24).

- The Sednet model predictions were highly sensitive to land management practices in the stream corridor. Throughout the upper Spring Creek watershed, hypothetically converting all land cover in the mapped stream corridor to agricultural lands, and hence removing most of the vegetative cover, increased the predicted annual average load at the Houserville gauge by 26%. Converting the land use/land cover to forest/wetland (i.e., assuming a high density of plants limits bank erosion) resulted in sediment loads almost 90% lower than the sediment load predicted under current land cover conditions (Figure 23).



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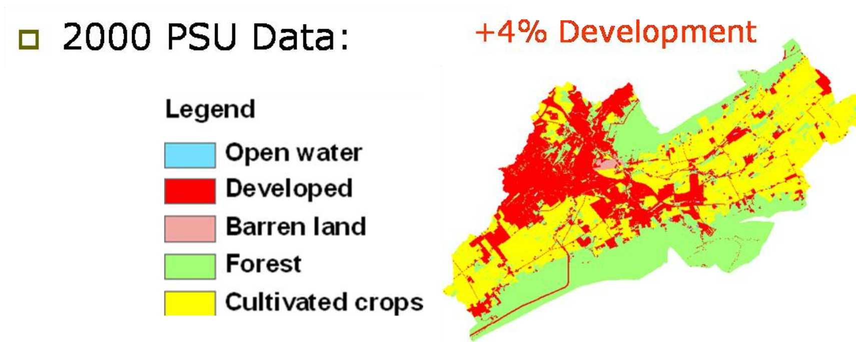

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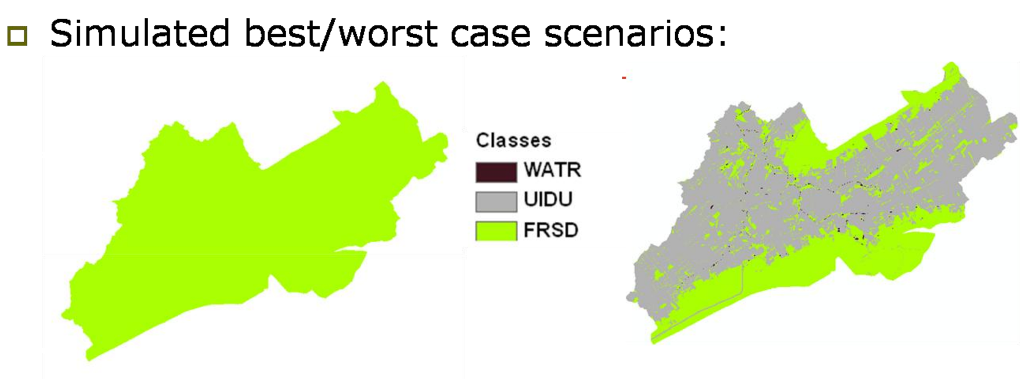

Figure 21. A) 1993 land use conditions; B) land cover change from 1993 to 2000; and C) hypothetical land use conditions in which all 1993 agricultural lands were converted to forest or development.

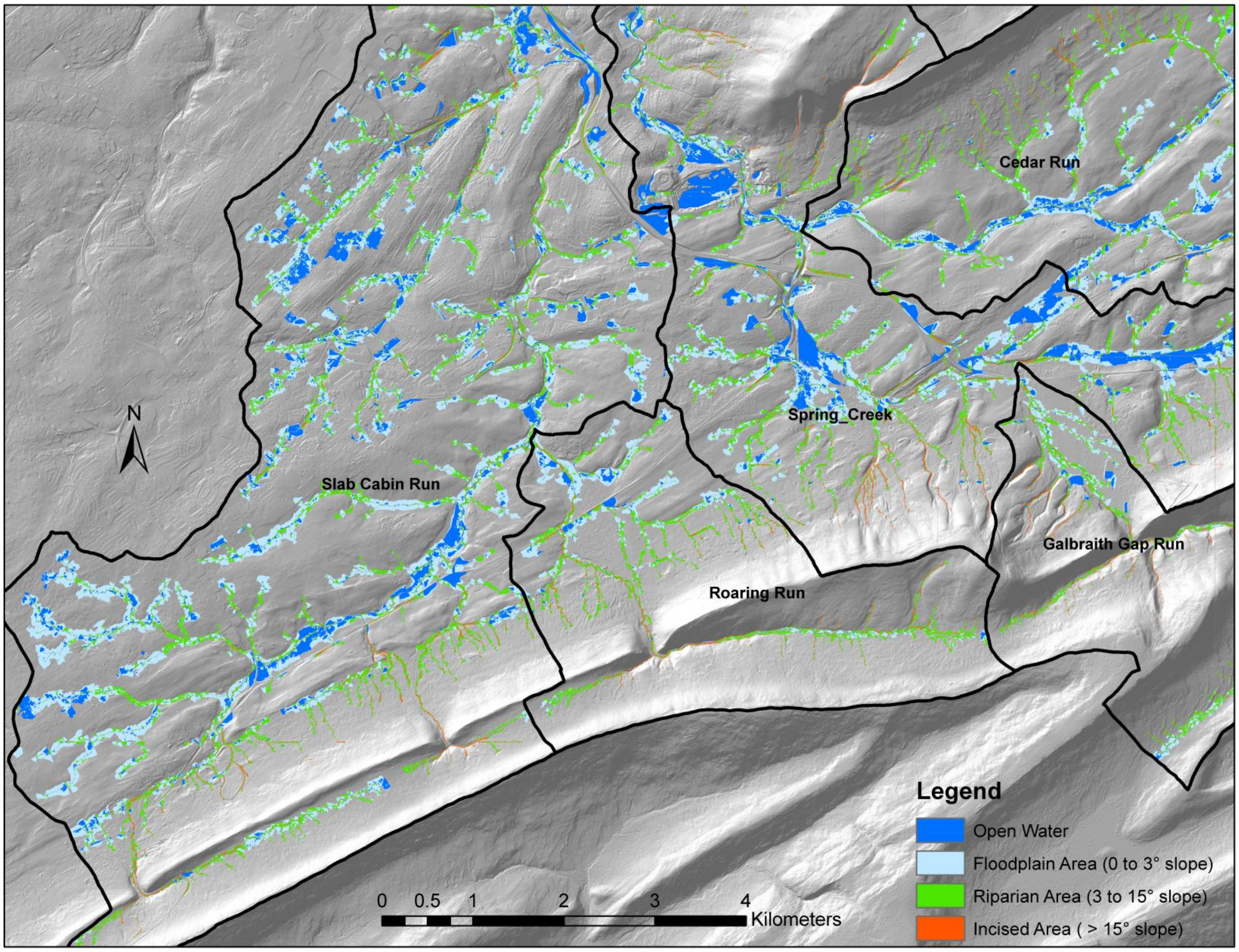

Figure 22. Stream corridor derived from 1-m LiDAR based digital elevation model. The stream corridor was identified based on the elevation difference between land pixels and the adjacent surface water body. Stream corridor functions were based on the topographic slope across the mapped stream corridor.

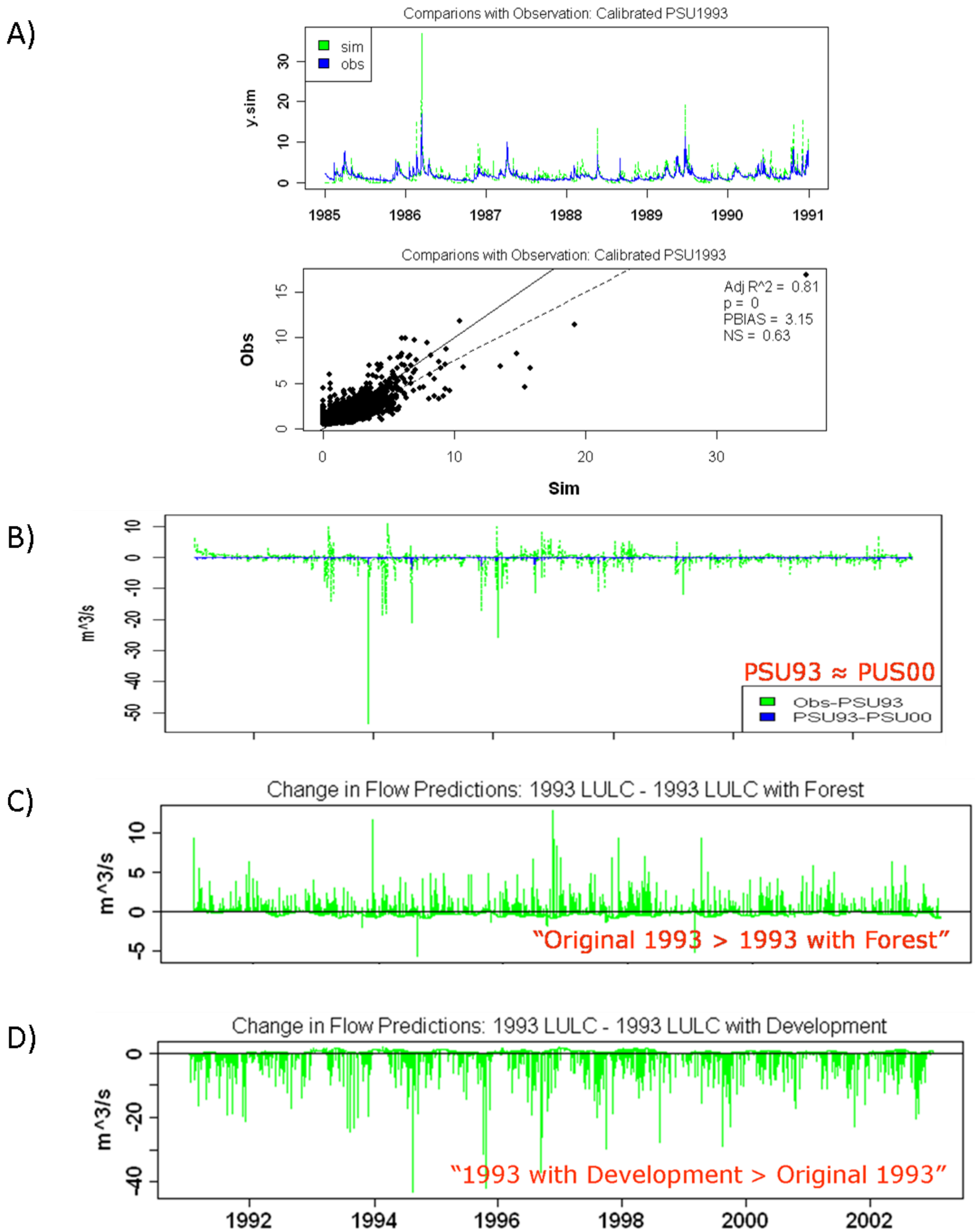

Figure 23. A) Results of SWAT model application calibrated with data from the Spring Creek Houserville gauge station and using the 1993 land cover data. Change in model predictions when B) 1993 land use data are replaced with 2000 land use data; C) 1993 agricultural areas are converted to forest; and D) 1993 agricultural areas are converted to development.

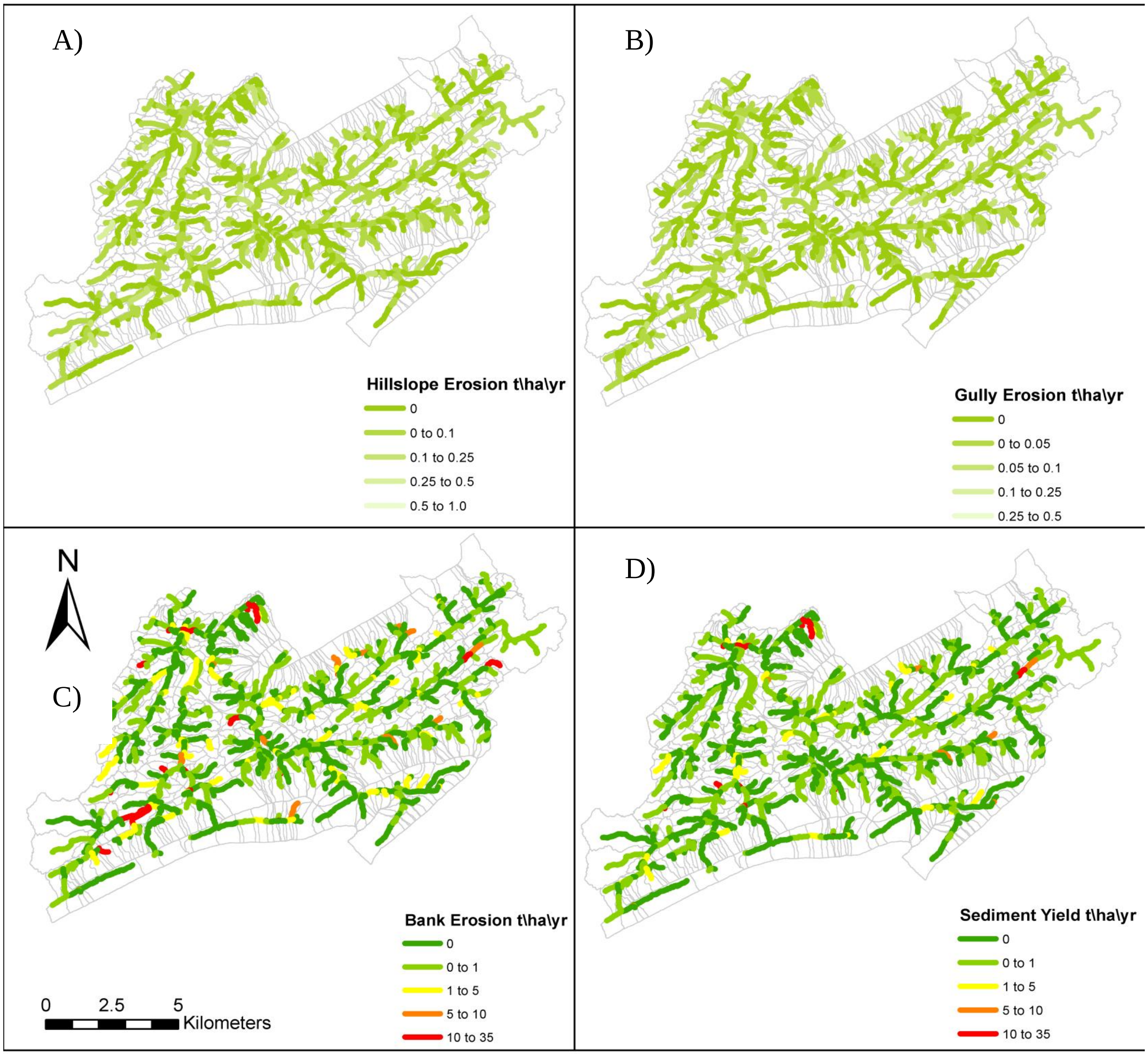

Figure 24. Sednet model predictions for the upper Spring Creek watershed. A) Sediment derived from hillslope erosion. B) Sediment derived from gully erosion. C) Sediment derived from streambank erosion. D) Total sediment yield.

Conclusions: Hydrologic and Landscape Modeling

Hydrologic and Landscape Modeling Messages (◊)
& Important Findings (•)

◊ Fine-scale LiDAR-based topography data were essential to modeling stream processes affecting sediment transport at the watershed scale. The detailed data enabled us to locate drainage features more accurately, identify the stream corridor, and evaluate the interactions between the drainage features and the surrounding landscape. Coarser-scaled elevation data did not provide enough information to identify incised channels, estimate channel dimensions, or identify riparian and floodplain areas.

◊ The combined results of our modeling effort suggest that hillslope processes affect stream sediment loads more by effects on channelized flow connections than by direct contributions of sediment delivery.

- Both the SWAT and Sednet models had a limited change in sediment delivery derived from hillslope erosion due to broad-scale changes in land use.
- Riparian BMPs represented in the Sednet model, however, showed significant improvements in water quality for the targeted reach, which were consistent with trends in the field-based monitoring and ecological analysis.
- The model also showed the difficulty of capturing these effects at down-gradient stream monitoring locations when the bank stabilization treatments are limited to a small proportion of the entire stream network.

SOCIO-ECONOMIC ANALYSES

Background

The purposes of the social science research component were to 1) assess current riparian BMP practices in the Spring Creek watershed; and 2) evaluate opportunities and obstacles for more widespread BMP adoption. In the fall of 2009, we began a two-stage analysis of riparian landowners throughout all sub-basins of the watershed.

Main objectives:

- *Explore the factors that affect effective farmer adoption of BMPs and citizen perception of water quality*

Methods

Qualitative Stage: The first stage included semi-structured interviews of landowners (n=16) and organization representatives (n=4) to gain a preliminary understanding of the following:

1) Underlying attitudes and social norms within the study watershed;
2) Landowners' experiences with riparian BMPs;
3) Existing riparian conservation programs; and
4) Factors that affect—positively or negatively—riparian BMP adoption.

Interviews were conducted in three rounds. First, organization representatives were interviewed for baseline information on existing programs and recommendations of potential landowner participants. Next, agricultural landowners (including commercial and part-time farmers) were interviewed. Lastly, non-farmers (i.e., residential landowners) were interviewed for comparison with farmer participants. Interviews were recorded and transcribed for thematic analysis.

Quantitative Stage: In the second stage, we implemented a mail-back questionnaire to riparian landowners. Riparian landowners were identified using an overlay analysis of 2007 Centre County tax parcel records and 1:24,000 high resolution USGS National Hydrologic Database surface hydrology maps (US Geologic Survey 2010). We used county land use codes (agricultural, residential, or vacant lot) to identify private landowners (n=706), from which we randomly sampled 500 recipients. We used a multistage mailing procedure consistent with Dillman (2000). This yielded 175 useable surveys, for an adjusted response rate of 39%.

The survey included questions addressing current land uses and management practices; knowledge and concern about water quality at three different scales: in the adjacent stream, the Spring Creek watershed, and in the Chesapeake Bay; baseline knowledge of riparian buffers; attitudes towards buffers; potential adoption obstacles; and sociodemographic characteristics. We conducted a small non-response telephone survey follow-up (n=11). The non-respondent landowners we were able to reach demonstrated less concern for water quality than did survey respondents but did not differ in other ways (Armstrong 2010).

To compare landowner types, we constructed a landowner typology that sorted landowners into three categories: traditional farmers, part-time farmers, and non-farmers. Landowners that had more than 10 acres, indicated they owned livestock or harvested crops on their property were categorized as "traditional farmers" (17%, n=26). Part-time farmers (15%, n=23) were differentiated by owning less than or equal to 10 acres of having less than 25% pasture and 25% fields. All remaining respondents were categorized as non-farmers (69%, n=109).

Landowner attitudes were assessed across seven dimensions: (1) baseline adoption willingness; (2) marginal willingness, or the increase in buffer adoption willingness under specified incentives (measured on a five-point scale from 'no change' to 'much more willing'; (3) buffer constraints, or limiting characteristics of the innovation; (4) outcome expectations (perceptions of potential outcomes from riparian buffers); (5) environmental attitudes; (6) innovation attitudes; and (7) private property rights attitudes.

Buffer constraints and outcome expectation items underwent a maximum likelihood factor analysis with varimax rotation and Kaiser normalization. Prominent items from each factor analysis were used to create composite variables. Marginal willingness, environmental attitudes, and innovation attitude items were also combined into three composite scales. All composite variables were tested for reliability. Private property rights attitudes and baseline willingness were both measured with single item indicators.

Results

Qualitative—Participant Characteristics: Twenty semi-structured interviews were conducted, including 16 riparian landowners and four institutional actors. Landowner participants were comprised of ten active farmers, six of which were hobby farmers and four were traditional farmers. Six non-farming riparian landowners were also interviewed as well (Table 13). Half of the 16 riparian landowners owned more than 50 acres, and four landowners had ten or fewer acres. Four interview participants were early adopters, having installed streambank fencing under a 1990's streambank fencing initiative led by Penn State extension. Two agricultural landowners and two non-farmers did not have riparian fencing or buffers. Most other agricultural landowners in this study had adopted riparian fencing, while two non-agricultural adopters had re-vegetated their streambanks. Re-vegetation generally included native tree and shrub plantings. Two non-farming landowners who owned retired farm properties were enrolled in CREP. Three landowners collaborated with multiple organizations for their riparian project. One landowner created a riparian buffer under a conservation easement agreement, while another household implemented a buffer on its own.

Four institutional actors representing the local watershed organization, the county soil and water conservation district, the municipal water authority, and Penn State Extension were also interviewed (Table 14). All participating institutional actors were directly involved with riparian buffers in the Spring Creek watershed. All of the four participating institutions administer conservation programs in addition to buffer implementation (Armstrong 2010).

<u>Qualitative—Private Landowners (Emerging Themes):</u>

(1) Land use determines information sources.

Agricultural landowners and hobby or non-farmers learned of riparian buffers through different diffusion pathways. Farmers typically learned of riparian buffers directly from within the agricultural community (i.e., farming organization, fellow farmer) or from within their agricultural professional network or Extension. Many of the farmers interviewed implemented streambank fencing in an early-1990's livestock exclusion initiative led by Penn State extension in cooperation with Trout Unlimited. Riparian BMPs such as streambank fencing were not widely adopted prior to this initiative, yet one farmer reported familiarity before Penn State approached him: *"When I was reading about [riparian fencing] in the [farming] magazine, I thought, 'Oh, I don't want that,' but then after I thought about it for a while, I thought, 'Yeah, I guess it would be alright.'"* (LO 7) Another commercial farmer learned of riparian buffers through a professional group: *"I went to a young farmer's meeting one night. The lady was there talkin' about it and I said, I could use some ideas, and maybe some help."* (LO 13)

Table 13. Landowner participants in qualitative phase.

Landowner	Land-owner Pseudonym	Land-owner Type	Parcel size (acres)	Adoption	How adopted
LO1	Clark Smith	non-farm	175	riparian buffer	conservation easement
LO 2	Fred and Lindsay Williams	non-farm	1	non-adopter	-
LO 3	Brian and Betty Reed	non-farm	2	non-adopter	-
LO 4	Tim and Megan Card	non-farm	2	riparian buffer	self-implementation
LO 5	David and Sara Hunter	non-farm	197	riparian buffer; CREP	CREP agreement
LO 6	George and Cheryl Hoyer	hobby farm	116	riparian buffer; fencing	multiple organizations
LO 7	Dan and Jo Kelley	hobby farm	20	fencing	1990s initiative
LO 8	Wade and Jane Rider	hobby farm	197	riparian buffer; CREP	CREP agreement
LO 9	Bart and Amy Greene	hobby farm	15	riparian buffer; fencing	multiple organizations
LO 10	David Miller	hobby farm	27	non-adopter	-
LO 11	Larry and Lydia Martin	hobby farm	10	fencing	1990s initiative
LO 12	Charles and Abby Long	agricultural	152	fencing	1990s initiative
LO 13	Jim Ford	agricultural	92	non-adopter	-
LO 14	James Harris	agricultural	150	fencing	1990s initiative
LO 15	Steve and Sue Welch	agricultural	47	fencing	multiple organizations
LO 16	Mark Johnson	agricultural	313	fencing	1990s initiative

Table 14. Institutional actor characteristics.

Institut-ion No.	Institution represented	General activities	Riparian activities
INST 17	Local watershed organization	Riparian buffer implementation, environmental advocacy conservation easements	Installation, project coordination, education, include buffers in conservation easements
INST 18	County conservation district	Soil and water conservation on farms and impaired waterways	CREP and other installation, project coordination, education
INST 19	Penn State extension	Education and outreach for soil and water conservation	1990s streambank fencing initiative
INST 20	Municipal water authority	Water supply and service; property management of riparian and well areas	Landowner outreach and recruitment for project installation, funding, project maintenance

In contrast, hobby farmers learned about riparian fencing and stream buffers through social networks.

> One of my soccer mom friends works for Soil Conservation… I'm on her mailing list for some reason. She's the person who notified people about the CREP program. She sent me an email--I'm on a big list serve probably about this email, for her and come to this meeting to learn about the CREP program or whatever. And a lot of times I get stuff from her that doesn't really apply to us because we're such a small-time farmer. But I saw this email from her, and I thought, 'She might know someone to help us to fix our [eroding] pond.' (LO 16)

On another hobby farm, the landowners learned of riparian buffers through personal observation and follow-up with a friend:

> While [my golf buddy] was the superintendent [at a local country club]—they were looking to do some stream rehabilitation thing. And when we bought this property, we called him up and said, 'Hey, where do we get the ball rolling with [our riparian buffer]. And he said, 'Well, the contact person is so-and-so at [the local watershed organization]'. (LO 9)

Here, landowners drew from a non-agricultural riparian project as evidence that they, too, could initiate a riparian restoration project on their small horse farm. In general, residential landowners were not familiar with riparian buffers and attributed them to agricultural properties. These participants typically knew of riparian projects on agricultural properties, and often mentioned farmers by name who installed fencing or riparian vegetation. However, both residential landowners interviewed who did not have a riparian buffer were not aware that buffers were applicable to their residential property. Instead, they followed typical residential lawn care procedures:

> Brian: It's all in grass. We mow it down there. I've planted some trees down there over the years.
>
> Betty: Trees don't grow too well, because it's really too wet down there.
>
> Brian: Yeah, quite a few of them died.
>
> Betty: It's awful soggy down there. But there is, the way the ground is, there isn't too much you can do with it. If it’s' a swamp, I guess it will always be a swamp. (LO 3)

(2) Perceived buffer improvements do not match the policy-based targets.

Adopters and non-adopters noted a variety of improvements associated with riparian buffers, whether these buffers were on their property, or more generally as a concept. The most frequently mentioned benefit was terrestrial or aquatic wildlife habitat.
“I liked the whole idea of the chain of wildlife and nature being as healthy as possible. I've always felt that way. So if you have a stream that's all full of mud, or erode, or if it doesn't have trees around it then it can’t do that.” (LO 1) Some people specified improved fish habitat from their riparian buffer: *“Apparently, this area is great fishing. We don't have people come here and fish, but we send a lot down from what I understand…I’m glad to have the banks preserved, you know....that’s', that's good to have that done.”* (LO 8) This is not surprising given the area’s history as a prime self-reproducing trout fishery.

Many landowners, particularly traditional famers, identified streambank stabilization as stand-alone improvement from riparian fencing. This is most likely because landowners saw the erosion mitigation aspect of stabilization rather than water quality improvements: *“Obviously the big benefit is to keep the livestock out of the stream, and not erode the bank.”* (LO 7) However, preventing property damage through erosion was not enough for one farmer to install streambank fences: *“There are two streams that come down off the hill. And those aren't fenced* *off--they are part of the pasture. I can show you lots of soil erosion. If somebody wants to dig in their pocket book and help financing, I'll be [interested]. I got ideas."* (LO 13)

Water quality was also an important perceived improvement of riparian buffers. Landowners with all types of land uses and from many parcel sizes associated riparian buffers with water quality. As one non-farm landowner said of streambank fencing: *"It helps the whole stream, really, because if somebody muddies it up here and the cows get in it, the problem doesn't just stay there."* (LO 3)

Participants identified three scales where buffers could make improvements: their property (parcel-level), locally (their stream reach or the Spring Creek watershed), or the Chesapeake Bay. As discussed in Chapter Two, there are many policy-based programs that provide farmers financial incentives to improve water quality in the Chesapeake Bay. However, the vast majority of participants did not identify the Chesapeake Bay as connected to their buffer project. Rather, almost all participants expressed that local environmental quality was more important than the Chesapeake Bay or other far-downstream regions: *"I've certainly heard people from the Bay talk. But my mindset would be to make what's best for our immediate watershed because we're the headwaters. And if we don't take care if it right here, how can we possibly take care of it down there?"* (LO 5) One landowner, who buffered a first order stream on his property, spoke with great pride of his contribution to the region: *"This is considered the finest natural brown trout spawning stream in the state, if not the country. So that inspired Trout Unlimited to institute a stream rehabilitation program [here]…and apparently it has been very successful in its protection of the fish, and the spawning has increased."* (LO 8) This project may have met the goals of Trout Unlimited; however, the landowner's perception of success was also limited to the regional improvements, and did not extend farther downstream.

Parcel-level improvements were the most commonly mentioned reason for adoption, within which erosion mitigation and terrestrial wildlife habitat were the most popular. Even a non-adopting landowner who perceived his stream as "background and atmosphere" (LO 10) was concerned about erosion. Streambank stabilization and wildlife can be seen by the untrained eye, which may make these benefits more recognizable to landowners, where as non-point source water quality pollution is more conceptual observable, unless the contaminants are obvious. Two additional parcel-level buffer benefits were frequently expressed—a place for recreation and property enhancement:

> [My wife] always says "I want to see our kids down there fishing and playing." At the end of the day, this was mismanaged for how many years. And it feels good when you do the right thing, regardless of it s this or something else. There's also, I don't know if it adds any equity or value to the home by redoing that, but the kids are part of it, too. We can enjoy the stream. It's not just a stream that's choked with reed canary grass and algae. (LO 9)

Most landowners believed that buffer-related improvements extended downstream. *"I would think [our streambank fence has made a difference]. I would truly think so. I know what the banks look like before it was done and what they do now. The more you can keep the soil in place the better off you are. Everybody benefited from it."* (LO 14) There was a general sense from BMP or buffer adopters that their project extended beyond their property lines. For some, this was expressed by a sense of care specifically for the stream. For others, the stream was part of a larger, but still local entity: *"It's nice being in a little community where you know the neighbors…And the stream, I really do think is our common ground. I mean, we picnic [by the stream] all the time."* (LO 2) Many farmers expressed that they have corrected past behaviors, which suggests there is something socially rewarding in exhibiting behavioral change:

> "When I was a kid, the young cows were in the meadow, they'd just kinda have free roam, and they'd go down [in the stream] wherever they wanted to.…We all know better these days, we're aware of what we were doing wrong." (LO 13)

Many landowners described a stewardship ethic that influenced how they managed their property in general: *"We take this stewardship concept very seriously. Because it's not just about farming practices, it's about the buildings and the apple trees, and, everything. We were just only one step in all the people who are going to live here before us or after us."* (LO 6) Unfortunately for riparian areas, some landowners based what was right upon a traditional aesthetic of "shored up banks" and clean streambanks free of tall, "messy" vegetation. The few landowners who associated the Bay with their riparian projects tended to express environmental or stewardship values: "*As long as I have cattle, boy, I would [maintain my streambank fence]…I think we all benefit from it. The farmer benefits, and I think the neighbors and environmentalists, and the Chesapeake Bay, probably, it would help that, too…It is important to all of us, I figure.*" (LO 16) This farmer and landowners who expressed values were generally involved in national-level environmental or agricultural organizations, which were sources of conservation information, and came from all property types. The landowners who attributed their riparian conservation adoption to improving the Chesapeake Bay stood apart from landowners who expressed little knowledge about riparian buffers or the Chesapeake Bay. This is not to suggest that if landowners knew more about the Bay, they would necessarily be willing to adopt riparian buffers. Rather, this suggests that for landowners who live nearly 200 miles away and are not environmentally oriented, the Bay does not resonate as something more worthy of protection than their backyard (Armstrong 2010).

<u>Quantitative Results—Descriptives:</u> Respondents averaged 62 years old (std. dev. = 13.3 years) and were generally long-time residents of Centre County (mean = 40 years, std. dev. = 21.5). The average property ownership length was 25 years (std. dev. = 16.6), with one quarter of respondents owning their property less than 14 years. Eighty percent of respondents were male. Respondents were highly educated, with 41% having at least some graduate education. The political views of respondents were normally distributed on a scale from 1, or "very conservative" (15%) to 5, or "very liberal" (13%). Respondents were categorized into four parcel size groups: less than one acre (n=57, 33%), one to less than four acres (n=42, 24%), four to less than thirteen acres (n=24, 19%), and thirteen or more acres (n=42, 24%).

Respondents rated the water quality in their own stream (mean = 2.55 on a scale of 1 = "excellent" to 5 = "poor") significantly higher than in the Spring Creek watershed (mean = 2.73, $p = .007$). Water quality in their own stream and in the Spring Creek watershed was also rated significantly higher than the Chesapeake Bay (mean = 4.00; $p = .000$ and $p = .000$, respectively). About one-third of riparian landowners mow their lawns within three feet of the stream. In general, crop harvest took place farther from the stream, with only one landowner indicating that he or she generally harvested crops within five feet of the waterway.

Two-thirds of respondents indicated that they interact with neighborhood members daily or weekly. A majority (59%) of respondents responded that they were close friends with one-quarter of the people in their neighborhood, while 26% of respondents indicated they were not close friends with anyone in their neighborhood.

Respondents were evenly distributed across five categories of riparian buffer adoption willingness: not at all willing (n=33, 21%), not very willing (n=23, 15%), somewhat willing (n=41, 26%), willing (n=33, 21%), and very willing (n=27, 17%). Six of the fourteen tested incentives made 50% or more of respondents “more willing” or “much more willing” to implement a buffer, while two of items elicited no change in willingness from over 50% of respondents (Table 15). The items that encouraged the greatest willingness increase tended to involve on-parcel benefits, such as reduced streambank erosion and free trees and shrubs. On the other hand, the items that received the highest “no change” responses (e.g., “most of your neighbors installed stream buffers”) measured indirect, social incentives. All items were compiled into the “marginal willingness” composite scale (alpha = .949).

Table 15. Respondents' marginal adoption willingness.

Would you be more willing if…	Mean	Std. Dev.	More and much more willing	No change
A buffer reduced streambank erosion	3.43	1.630	58%	24%
You had a say in designing your buffer	3.40	1.555	56%	20%
Invasive or noxious weeds were removed for you	3.32	1.602	57%	26%
Your buffer included wildflowers	3.27	1.612	55%	25%
A buffer made water runoff from your property cleaner	3.25	1.587	51%	25%
The trees and shrubs were free	3.10	1.656	51%	31%
You received you received yearly payments for your buffer costs	3.01	1.686	49%	34%
Volunteers planted the buffer	3.01	1.626	47%	31%
You received a one-time payment for your buffer installation	2.71	1.548	36%	34%
You were given guidance how to build a buffer	2.66	1.615	35%	40%
Most of your neighbors installed stream buffers	2.45	1.528	30%	43%
Someone in your neighborhood installed a buffer	2.15	1.384	19%	50%
A good friend installed a stream buffer	2.01	1.361	19%	58%

Scale: 1= no change, 2 = slightly more willing, 3 = somewhat more willing, 4 = more willing, 5 = much more willing

Our factor analyses of attitudinal items yielded two composite variables: "buffer constraints", which represented potential obstacles to riparian buffer adoption (total variance explained = 60.2%; Eigen value 3.612; alpha = .866); and "outcome expectations" or the amount that landowners agree that riparian buffer on their property would improve various conditions (total variance explained = 69.9%, Eigen value = 8.391, alpha = .957) (Table 16). The outcome expectation scale was constructed using all of the tested items, meaning that our factor analysis did not find multiple dimensions to landowners' anticipated outcomes. Five items emerged from the buffer constraints factor analysis (total variance explained = 60.2%, Eigen value = 3.612), which underwent reliability analysis (alpha = .866) and a log transformation. All scaled variables underwent a means substitution for missing data.

We created an environmental attitudes composite scale from four Likert-scale items of environmental attitude measures (alpha = .797), an innovation attitudes scale consisting of two items (alpha = .588), and one Likert-scale item to measure private property attitudes (Table 17).

Table 16. Factor analysis and composite scales.

Buffer Constraints∞ (α = .866)		**Outcome Expectation∞** (α = .957)	
Item	Factor Loading	Item	Factor Loading
doesn't make sense for the size of my property	.720	wildlife habitat	.753
would take up too much land	.788	Water quality downstream	.956
takes too much time to maintain	.650	Water quality in the Chesapeake Bay	.920
plants look messy	.741	Water quality in my stream	.933
doesn't fit appearance of neighborhood	.778	Character of my property	.817
would bother my neighbors	.654	Water quality in local groundwater	.939
		children's exposure to nature	.781
		Fish habitat	.761
		flood protection downstream	.733
		local drinking water	.856
		Property values	.679
		access to buffer program payments	.556

∞ five-point Likert scale

Table 17. Environmental and innovation attitude scales.

Environmental Attitudes∞	**Innovation Attitudes∞**
Items (α = .797)	Items (α = .588)
I have a moral obligation to maintain water quality	I am always looking for ways to improve my property
I would be upset if my activities harmed my stream	I am the kind of person who is willing to take a few more risks than others
Protecting the environment is important to me	
I want to conserve the stream for future generations	

∞ five-point Likert scale

<u>Quantitative Results—Bivariate Relationships (Landowner type and riparian buffer adoption)</u>:

We sought to compare landowner types (traditional farmer, part-time farmer, and non-farmers) across the range of variables linked to riparian buffer adoption: knowledge and concern about water quality; familiarity with riparian buffers; attitudes towards buffers; outcome expectations; and potential adoption obstacles.

Six variables exhibited significant differences between landowner types, signaling that there are far more similarities among landowners than hypothesized on a diverse landscape. Traditional farmers have heard significantly more about riparian buffers than non-farmers (p = .023), and perceive they know significantly more about water quality in the stream on their property than do part-time farmers (p = .050) (Table 17). Non-farmers rate water quality of the stream on their property significantly higher than traditional farmers (p = .001). Concerning attitudes towards buffers, traditional farmers more strongly agree than non-farmers (p = .046) that buffers take too much time to maintain. There were no significant differences in baseline or marginal adoption willingness to increase buffer size among respondents; however, part-time farmers were significantly more willing to increase the size of their riparian buffer than non-farming respondents if a good friend adopted a buffer (p = .029). Traditional farmers are significantly more likely to agree with pro-private property rights statements than non-farmers ($p = .012$); however, differences between landowner types do not exist for environmental and innovation attitudes.

Despite common rhetoric that emphasize differences in characteristics between traditional farmers and ‘newcomers’ to the rural landscape, we found no differences among landowner types in age, length of residence, length of ownership, primary residence, education, political views, gender, neighborhood engagement variables, or outcome expectations. Overall, the bivariate analysis thus reveals that these landowner types are more similar than different in terms of their attitudes towards riparian buffer adoption, environmental attitudes, and socio-demographic attributes (Armstrong 2010).

Conclusions: Socio-Economic Analyses

Socio-Economic Messages & Important Findings (•)

- Non-farmers are a bottleneck for riparian buffer adoption across the watershed. Non-farmers are less willing to adopt riparian buffers than agricultural landowners. We believe this is because non-farmers have heard significantly less about riparian buffers than farmers.
- The amount a landowner heard about riparian buffers is also positively related to the amount heard about Chesapeake Bay water quality. This reflects the general approach to educate people (particularly farmers) about Bay-related water quality improvements from riparian buffers and not local water quality outcomes.
- Willingness to adopt riparian buffers increases with perceived knowledge about stream water quality.
- Baseline willingness will increase with more positive attitudes towards riparian constraints. This suggests that there is a group of landowners who strongly support riparian buffers.
- Riparian buffers are socially desirable based on the proportion of neighbors considered close friends. While this may encourage buffer adoption at the neighborhood scale, it may also discourage buffer adoption in areas where normative behaviors (e.g., lawn mowing) disapprove of riparian vegetation.
- Stream flow regularity is positively related to landowners' perceptions of water quality, attitudes of stream importance, and perceptions of how buffers may improve environmental outcomes. This has important implications for riparian management and water quality in ephemeral stream reaches.
- If landowners believe that buffers produce results, their willingness to adopt buffers will increase. This suggests a need for more education on local and downstream ecosystem services provided by riparian buffers.

OUTREACH

Members of the Spring Creek research team attended the annual meetings of the USDA-CSREES National Water Conference in 2008, 2009, and 2010. Technical presentations were made during these meetings and at other scientific venues.

In June 2010 Canaan Valley Institute (CVI) held a workshop for the public and community officials within the Spring Creek watershed. The goal of the workshop was twofold: to use the results of the socio-economic survey to educate those least likely to adopt riparian buffer practices on their importance and to disseminate the results of the USDA Best Management Practices in the Spring Creek Watershed Project to the residents of the watershed. Invitations were mailed to each of the hobby farmers in the watershed with riparian areas on their properties. Invitations were also sent to elected officials and staff of each of the townships and boroughs within the watershed as well as the members of the Spring Creek Watershed Association. Mailings were followed-up one week prior to the workshop with an email reminder.

Poster presentations were developed to display the Landscape Characterization & Fine-grained Assessment and Hydrologic and Landscape Modeling of BMP Performance portions of the project. Oral presentations were delivered on the overall project's methods and results, the Ground Based Monitoring and Ecological Analysis, and the results of the Socio-Economic Study. The oral presentations were followed by a question/answer session. ClearWater Conservancy provided educational materials on the importance of riparian buffers and information on programs available to assist homeowners in riparian buffer establishment.

Despite the mailing of individual invitations followed by an email invitation, attendance at the workshop was lower than we anticipated. However, the homeowners in attendance were engaged in the conversation and made inquiries into how buffers may be established on a non-agricultural land. Due to feedback from attendees, we conclude that this type of public workshop is an effective means of communicating scientific research projects to the public. However it is our recommendation that additional marketing is necessary to increase attendance at a voluntary workshop. Including a brochure detailing the scope of the project with the invitation along with articles in the local news media likely would have generated interest in homeowners who did not attend.

In May 2010, the Spring Creek research team hosted CEAP's National Synthesis Team for a two-day workshop and field tour to inform them of our findings.

In December 2010, Brooks presented the Spring Creek BMP findings to the Pennsylvania State Technical Committee monthly meeting for the Natural Resources Conservation Service. Following the presentation, there were questions and discussion from the members on how to incorporate these findings into future guidance.

SUMMARY

This study illustrates the synergistic nature of watershed management, of which BMPs play an integral part. These results show the success of a BMP depends on a variety of factors including: (1) proper identification of the locations and sources of pollution, including the actual pathways by which it enters the stream; (2) proper alignment of the BMP with those locations; (3) effective monitoring techniques that target the stressor(s), match the BMP with the appropriate indicator, and allow sufficient time periods for capturing responses; and (4) capitalizing on proven factors that encourage landowner adoptions of BMPs, while effectively addressing impediments to BMP adoption. In addition, BMP success or failure is often dependent on external factors within the watershed, especially land use change and multiple-year weather patterns (e.g., drought, flooding). Thus, it is important to consider these additional impacts to water quality when attempting best management practices.

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