ABSTRACT

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Stream restoration projects that attempt to reduce channel incision and bank erosion by reconstructing the channel and grading and armoring stream banks (channel stability restoration projects) are common, particularly in urbanized watersheds. However, integrated assessment of changes in geomorphic processes and ecological properties within the channel and in the surrounding riparian zone induced by stability restoration has rarely been carried out across multiple restored streams. I provide such an assessment by measuring channel complexity, bed sediment dynamics, channel movement rates, riparian soil structure and function, and diatom communities in multiple restored streams located in urbanized watersheds and comparing these measurements to measurements from urban and forested reference streams.

Stability restoration appears to have reduced lateral channel migration and channel incision through channel reshaping. Patterns of bed sediment movement

were altered through the effects of added channel obstructions on flow dynamics and bed sediment size distribution. Channel stability restoration did not alter channel complexity, primarily because channel complexity was not reduced by urbanization as has commonly been assumed. Restoration did not alter diatom communities either, primarily because diatom communities responded more strongly to urbanization-induced changes in water chemistry. Riparian soils were negatively impacted by stability restoration, particularly compared to riparian buffer establishment, which had mostly neutral effects on riparian soils.

Channel stability restoration can provide a minor increase in channel and bed sediment stability. However, changes in bed sediment stability were driven by in-channel restoration structures, which can be placed without grading the banks or reconstructing the channel. Riparian buffer restoration can also stabilize channels and will provide wood to channels, which can provide similar stabilization benefits as restoration structures. Restoration of channel stability using only in-channel structures and riparian vegetation planting would reduce the cost of stability restoration and reduce negative impacts to riparian soils.

Even so, effects of stability restoration were often overwhelmed by processes operating beyond the channel boundaries, suggesting that reach-scale targeting of channel instability needs to be assessed at the watershed scale and may need to be given lower priority to such restoration approaches as stormwater management, which directly address the causes of channel instability.

ECOLOGICAL AND GEOMORPHOLOGICAL IMPACTS OF STABILITY RESTORATION IN URBAN STREAMS

By

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Preface

This dissertation contains an introduction, four research chapters, and a conclusion. Research chapters are presented in manuscript form with abstract, introduction, methods, results, and discussion sections. Tables and figures with legends are embedded in the research chapters. All literature cited throughout the dissertation is listed in a single bibliography at the end.

Dedication

For my cousins, Johnny and Jennifer.

Acknowledgements

I thank my advisor Margaret Palmer for all her guidance and support during the course of this project. She has always pushed me to do my best work and encouraged me to think about how my research fits into the big picture of ecology. I appreciate all her hard work. Thanks also to my committee, Alexa Bely, Don Charles, David Inouye, Brian Needelman, and Peter Wilcock for always lending their time when I asked. Their feedback has greatly improved my research.

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Introduction

The world's human population is estimated to have recently surpassed 7 billion and an increasing proportion of that population is living in urban areas (UNFPA 2011). As urban areas expand, there will be an increasing need to understand the impacts of urban development on ecosystems and how to mitigate these impacts in order to manage natural systems to provide ecosystem services (McDonnell and Pickett 1993, Alberti 2005). The effects of urban development on stream ecosystems have been an intense area of research over the past few decades (see reviews by Paul and Meyer 2001, Walsh et al. 2005), but questions still remain as to the effects of urban development on aquatic communities and ecosystem functions, how best to manage aquatic resources in urban areas, and the most appropriate restoration strategies in degraded systems (Wenger et al. 2009).

With urban development comes an increase in impervious surface cover on the landscape, which reduces infiltration and increases surface runoff during rain events (Dunne and Leopold 1978). As a result, stormwater is routed more directly to stream channels and causes an increase in peak flows and an increase in the rate of the rise and fall of peak flows (Espey et al. 1965, Hirsch et al. 1990). The increased peak flows have greater power to erode and transport sediment from the channel and often accelerate rates of bank erosion and channel incision (Wolman 1967, Hammer 1972, Arnold et al. 1982, Booth 1990, MacRae and Rowney 1992, Booth and Jackson 1997, Hardison et al. 2009). Increased rates of bank erosion are often a concern in urban areas where property alongside streams may be threatened.

Restoration of channel stability is often undertaken in urban streams to try and reduce rates of channel incision and bank erosion (Brown 2000, Shields et al. 2003, Booth 2005, Hassett et al. 2005, Radspinner et al. 2010). Such channel stability restoration projects aim to create a stable geomorphic channel form and do so by grading stream banks and reconstructing the channel to designed cross-section and longitudinal profiles (Shields et al. 2003, Hassett et al. 2005). The channel form is stabilized by different combinations of channel and bank protection features such as riprap, which is simply large boulders placed on the channel banks, log and boulder flow deflectors, which route water away from the banks during high flow, and rock weirs and log vanes, which prevent channel incision at certain points on the channel (Brown 2000, Radspinner et al. 2010). Channel form is typically designed by classifying the stream according to an established system and matching channel dimensions to the identified channel type and measured values at nearby undisturbed reference streams (e.g., Rosgen 1996).

The fact that channel stability restoration targets channel form in an attempt to alter the processes of bank erosion and channel incision has been criticized (Simon et al. 2007), and there are several examples in the literature demonstrating how this approach can fail geomorphologically (Kondolf et al. 2001, Smith and Prestegaard 2005). Yet channel stability restoration projects are common throughout the United States (Bernhardt et al. 2005), and managers are still being trained in the use of channel classification approaches. Therefore, it will be important to understand how stability restoration projects in urban areas are altering geomorphic processes and their impacts on ecological systems both within the channel and in the adjacent riparian zone.

Geomorphological studies of stability restoration projects have focused on how channel morphology changes after restoration (Kondolf et al. 2001, Smith and Prestegaard 2005, Miller and Kochel 2010), and whether individual structures placed within the channel provide the intended benefit, (Brown 2000, Borg et al. 2007, Bhuiyan et al. 2009, Miller and Kochel 2010, Radspinner et al. 2010). Ecological studies of projects that target habitat improvement are numerous (see reviews by Roni et al. 2008, Miller et al. 2010, Palmer et al. 2010), but studies of stability restoration projects are less common, although some work suggests stability restoration may benefit fish habitat and fish populations (Shields et al. 1995, Baldigo et al. 2008, Ernst et al. 2010). Work that examines both the physical and biological impacts of stability restoration are rare (Tullos et al. 2009, Violin et al. 2011), and very few studies have simultaneously examined how geomorphological process and ecological structures (e.g., habitat complexity, biological communities) in urban streams respond to stability restoration at the reach scale across multiple sites. Providing such a study was the primary goal of my dissertation research.

In Chapter 1 I investigated whether stability restoration altered channel form using a measure relevant to stream processes and biological communities – channel complexity. To determine whether restoration altered channel complexity, I compared measurements in restored streams to measurements made in nearby urban and forested reference streams. Previous work has generally found complexity to be low in urban streams (Grimm et al. 2005, Gooseff et al. 2007, Violin et al. 2011), and

I therefore hypothesized that restoration would increase channel complexity, primarily through the addition of large structural elements, e.g., boulders and logs. However, I found that urban stream channels were no less complex than forested reference channels and that restoration did little to alter channel complexity of urban streams. These results suggest: 1) that the common assumption that urbanization will reduce channel complexity (Walsh et al. 2005) is not necessarily true, and 2) that channel stability restoration does not increase habitat heterogeneity in urban streams. I also compared levels of channel complexity in my study streams to values of channel complexity available in the literature and found that the variability in channel complexity I observed across my study streams spanned the variability seen in the literature. This suggests that my study streams are comparable to streams from diverse geographic regions and imply that the results should be broadly applicable beyond my specific study region.

In Chapter 2 I explored how stability restoration altered geomorphological processes – specifically, bed sediment disturbance and channel movement rates (e.g., bank erosion, channel incision). Using research on the effects of bridge piers and other flow obstructions on flow dynamics and bed sediment scour and fill processes, I predicted that the boulders and logs added during restoration would similarly act as flow obstructions and change bed sediment movement patterns through alterations of flow and bed sediment size distributions. I tested these predictions by using scour chains to measure scour and fill patterns in restored streams and one urban and one forested stream during two separate periods in which high flows occurred. I also quantified the influence of flow obstructions on patches (0.36 m^2 areas) of bed sediment where scour chains were installed and explored relationships between scour/fill patterns and these flow obstruction metrics at the patch, unit (individual riffles and runs), and reach (~50-100 m stream length) scales. Flow obstructions were found to influence the probability of sediment movement at the patch scale. At the reach scale, sediment size influenced both the proportion of bed sediment that was mobile during high flow events and the variability in proportion bed sediment mobile between units, with streams with coarser sediments having lower overall proportion mobile bed sediments but higher variability between units in proportion mobile bed sediments. Measures of bed sediment distribution suggested that flow obstructions had altered flow patterns sufficiently to coarsen bed sediments. I propose a mechanism for the observed patterns of scour/fill and bed sediment distribution whereby flow obstructions alter baseflow shear forces, which in turn coarsens the surrounding sediment. The coarser sediment around flow obstructions is predicted to be less prone to movement during high flow than sediment in units without flow obstructions where sediment is more easily entrained.

I investigated channel movement rates in restored streams in comparison to urban and forested streams by taking repeated measures of cross-section profiles at each stream. Rates of channel widening and thalweg scour (incision of the deepest part of the channel) were significantly lower at multiple restored streams compared to the urban stream during one period, suggesting that restoration may have helped reduce bank erosion and channel incision in these streams. Erosion rates on armored and non-armored banks were found to be similar, suggesting that reduction in channel movement rates was attributable to establishment of a stable channel form rather than bank armoring. However, significant aggradation in one channel from high sediment loading upstream indicated that watershed processes can overwhelm any local effects of stability restoration.

In Chapter 3 I assessed the impact of stability restoration procedures on riparian soil structure and function in comparison to impacts from riparian buffer restoration. Channel stability restoration involves the use of heavy machinery to grade channel banks and arrange structures within the channel. Use of heavy machinery is known to compact soil (Merz and Finn 1951, Campbell et al. 2002, Bruland and Richardson 2005) and previous work on restoration projects similar to channel stability restoration have shown reduced variability and total amount of soil organic matter (SOM) in restored areas after restoration (Unghire et al. 2010). I therefore hypothesized that channel stability restoration would negatively impact soils by increasing bulk density (a measure of soil compaction) and reducing SOM, root biomass (vegetation is removed during restoration and root growth is reduced in compacted soils), and denitrification enzyme activity (DEA, which is positively correlated with SOM when other factors are not limiting) in comparison to non-restored control sites. I also hypothesized that the disturbance of stability restoration would increase cover of invasive plant species. Further, I hypothesized that soils would recover over time from the disturbance of stability restoration. In contrast, I hypothesized that riparian buffer restoration would have positive to neutral effects on riparian soils and vegetation. Soils in riparian buffer restoration sites were generally similar to soils in the control reach, which supported the hypothesis that riparian buffer restoration would not negatively impact riparian soils. In contrast, channel stability restoration had generally negative impacts on riparian soils, in particular bulk density was significantly higher, root biomass was significantly lower and invasive species cover somewhat higher in restored sites relative to non-restored control sites. There was also no evidence of recovery over time at stability restoration sites. Although comparison with forested stream soils suggested the magnitude of effects of stability restoration was small, the generally negative impacts suggest that riparian buffer restoration is a more ecologically favorable method of stabilizing channels.

In Chapter 4 I sampled diatom communities in restored streams and compared richness, diversity, and composition of the diatom community to communities sampled in non-restored urban and forested streams. Given that I saw no effect of restoration on channel complexity in Chapter 1 and that studies of habitat restoration projects have generally found no increase in invertebrate and fish diversity (Roni et al. 2008, Palmer et al. 2010), I hypothesized that restoration would have little impact on diatom diversity and community composition. Richness and diversity were similar in forested, urban, and restored streams and community composition was similar in urban and restored streams, in support of the hypothesis. The similar levels of diversity in forested and urban streams was somewhat surprising, given that urbanization has generally reduced diversity of fish and invertebrates (Paul and Meyer 2001, Walsh et al. 2005), but the result agrees with some studies of algal diversity in urban streams (Sonneman et al. 2001, Potapova et al. 2005). I attributed the differences in community composition between forested and urban streams to urbanization-induced differences in water quality and concluded that urbanization may be driving regional homogenization of diatom communities through loss of species typically found in forested streams.

I conclude that the main effect of channel stability restoration has been the alteration of bed sediment disturbance patterns by restoration structures. These structures can be added without bank grading and channel reconstruction, which will reduce the cost of stability restoration and reduce negative impacts to riparian soils. Furthermore, riparian buffer restoration is found to be a more ecologically favorable restoration technique, as it had some positive effects on riparian soils and may mitigate urbanization-induced degradation of channel complexity. I also conclude that effects of reach-scale restoration may be overwhelmed by watershed-scale processes that are better addressed by other restoration techniques, such as stormwater management, that address the cause of channel instability.

Chapter 1: Range of variability of channel complexity in urban, restored, and forested reference streams

ABSTRACT

Channel complexity is an important ecological property of stream systems and is often targeted for restoration in channelized urban streams. However, channel complexity is rarely defined explicitly and little research on channel complexity has been conducted in streams in urban catchments that have not been directly channelized by human activities. Therefore, it remains unclear whether restoration of non-channelized urban streams has improved complexity. I explicitly define channel complexity and use a multi-metric approach to provide a comprehensive assessment of complexity in multiple restored, urban, and forested streams on the Maryland Coastal Plain and two streams of differing land use in Colorado. I also expand on the Maryland and Colorado results with a literature survey of channel complexity from diverse geographic regions. Many streams draining urban catchments in Maryland had relatively high values of some complexity metrics compared to forested reference streams in Maryland and compared to values for pristine streams calculated from the literature. This suggests that streams in urban catchments that are not directly manipulated by human activities (e.g., channelization or piping) may be able to maintain channel structures beneficial for aquatic organisms even when impervious surfaces are the dominant form of land use in the catchment. Restored streams in Maryland had equal or lower values of many complexity metrics compared to streams draining urban catchments in Maryland. This suggests that restoration of streams

draining urban catchments did not improve overall channel complexity. My results highlight the need to define explicitly and measure attributes of channel complexity that are targeted during restoration, to determine if streams in urban catchments are truly degraded with respect to channel complexity. Combined with recent synthesis work suggesting biodiversity may not be improved by increasing channel complexity, these results indicate that targeting catchment processes may prove a more useful approach to restoration than attempting to move channel complexity in streams draining urban catchments toward conditions in forested reference streams.

INTRODUCTION

While stream and river restoration has been dramatically influenced by hydrogeomorphic theory (Palmer and Bernhardt 2006), ecological theory has also played a role particularly in terms of the interactions of physical processes with ecological processes and biotic communities. One prominent example comes from theory on the importance of physical heterogeneity in structuring and sustaining ecological systems (Levin and Paine 1974, Winemiller et al. 2010). In streams, spatial heterogeneity in geomorphology is widely known to interact with flow dynamics to create diverse habitat patches (Palmer et al. 1997, Lake 2000) which in turn may influence species diversity and ecological resilience in the face of disturbances such as floods (Townsend 1989, Hildrew and Giller 1994). Heterogeneity has received much attention in stream management because influencing physical structure seems more tractable than influencing many other factors believed to support productive and diverse ecosystems (Palmer et al. 1997, Palmer et al. 2010). In fact, central to many stream restoration efforts is the assumption that rehabilitation of physical habitat heterogeneity will lead to the restoration of biological communities (Palmer et al. 1997, Spanhoff and Arle 2007).

Various concepts have been used to explore physical heterogeneity in streams including ones that focus on measurements intended to characterize channel complexity based on reach-scale geomorphic attributes. It is widely assumed that channel complexity plays a critical role in maintaining stream ecosystem structure and function, and studies have shown that channel simplification can lead to reduced diversity and abundance of fish and macroinvertebrates, reduced hydraulic retention, and reduced retention of organic matter and nutrients (Jungwirth et al. 1993, Laasonen et al. 1998, Muotka and Laasonen 2002, Grimm et al. 2005, Sheldon and Thoms 2006, Bukaveckas 2007, Gooseff et al. 2007, Muotka and Syrjanen 2007). One of the most commonly cited impacts of urban development is loss of channel complexity due to more frequent erosive floods that can cause channel incision and bank erosion (Walsh et al. 2005). Additionally, streams are often piped, straightened, channelized, or otherwise intentionally simplified for various purposes during catchment urbanization (Arnold et al. 1982, Ramírez et al. 2009).

Given the evidence that physical complexity is ecologically important, increasing channel complexity has often been a goal of stream restoration (Brookes et al. 1996, Bernhardt et al. 2005, Katz et al. 2007). Many restoration projects on channelized streams have involved increasing substrate, depth, and flow variability (Jungwirth et al. 1993, Laasonen et al. 1998, Muotka and Laasonen 2002, Pretty et al. 2003, Harrison et al. 2004, Lepori et al. 2005, Bukaveckas 2007, Lorenz et al. 2009). Only a few studies have evaluated the effects of restoration on channel complexity in streams that have not been deliberately channelized such as those streams in urban catchments assumed to be geomorphically simplified by altered flow regimes. While these studies have used various methods for estimating channel complexity, it is clear that restoration does not always increase physical complexity (Larson et al. 2001, Tompkins and Kondolf 2007, Tullos et al. 2009, Violin et al. 2011).

Even beyond the restoration literature, channel complexity has rarely been defined explicitly and has been measured in different ways depending on the objectives of each study. Some authors have implicitly defined channel complexity as essentially equivalent to hydraulic retention (Grimm et al. 2005, Bernot et al. 2006). Gooseff et al. (2007) found that hydraulic retention was correlated with channel complexity, measured using a multimetric index based on slope, longitudinal roughness, and sinuosity. Sheldon and Thoms (2006) devised measures of complexity based on cross-section profile variability and related these measures to storage of organic matter. Others have used channel complexity to refer to the quality of in-channel habitat, defining streams with large amounts of instream wood, multiple habitat types (pools, riffles, runs, etc.), and large pool volumes as being more complex than channels with flat bed profiles lacking instream wood (Roper and Scarnecchia 1995, Schmetterling and Pierce 1999, Kaufmann et al. 2008, Tullos et al. 2009). The latter definition has also been referred to as habitat heterogeneity, habitat complexity, and habitat diversity (Gorman and Karr 1978, Schlosser 1982, Shields et al. 1998, Milner et al. 2008, Violin et al. 2011). These terms are often used loosely and interchangeably to describe either spatial or temporal variability in channel physical features or structures.

I argue that there is a need to be more explicit about how channel complexity is measured and why different aspects of complexity may be more or less emphasized depending on the ecological attribute of interest. This is particularly important in a restoration context because project designs may target different physical aspects of complexity depending on the goal of the project. My objectives were to: 1) generate a comprehensive measure of channel complexity using a multivariate statistical approach; 2) use this measure to determine whether different components of channel complexity vary across a catchment urbanization gradient; 3) determine whether these complexity components respond similarly to restoration interventions; and 4) assess the range in channel complexity over a broader geographical area than my study sites. Previous research has generally found indicators of complexity to be lower in non-channelized streams in urban catchments compared to streams in natural reference condition (Pizzuto et al. 2000, Reid et al. 2008, Cookson and Schorr 2009, Violin et al. 2011) and many restoration projects focus on enhancing channel complexity (Brookes et al. 1996, Bernhardt et al. 2005, Katz et al. 2007). Thus, my null hypothesis was that streams in more urbanized catchments would exhibit the lowest levels of complexity and that after restoration, each component of complexity would increase relative to non-restored control streams in urban catchments.

METHODS

<u>Study Sites</u>

I gathered geomorphic data on multiple streams in forested and urban catchments and also on restored streams in urban catchments, all located in the

Coastal Plain physiographic province of Maryland (USA). To provide a broader geographic context for the range of variability in complexity observed in Maryland streams, I compared Maryland streams to streams surveyed with the same methodology in the plains and Front Range mountains of northern Colorado. I also expanded the geographic context by collecting channel complexity values available in the literature for streams across diverse regions.

I surveyed 26 1st or 2nd orders streams in Anne Arundel County Maryland (N 39°03'00", W 76°37'00"), including 9 restored streams (Fig. 1.1). Although Anne Arundel County is contained entirely within the Coastal Plain physiographic province, it is further subdivided into the Glen Burnie rolling upland district in the northern part of the county and the similar but somewhat more dissected Crownsville Upland District in the central and southern parts of the county (Reger and Cleaves 2008; Fig. 1.1). In addition, sediments in the central region of the county are of Tertiary origin and are primarily composed of glauconitic fine to medium sand and silts (Mack 1962, Glaser 1968). Sediments in the north of the county are of Cretaceous origin and are also composed of sand, silts, and clays, but contain more gravel than southern formations (Glaser 1968).

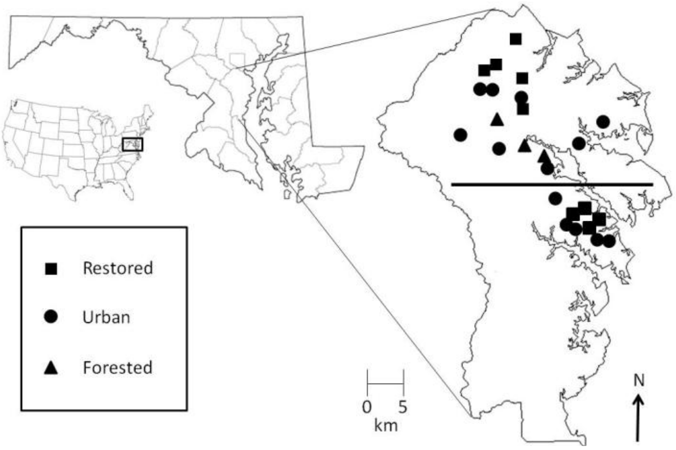


***Figure 1.1**. Map of Anne Arundel County, Maryland and study sites. The line across the map separates northern from southern streams.*

All restoration projects were stability restoration projects, involving a combination of channel manipulation and bank stabilization activities, all performed with heavy machinery (Fig. 1.2). At each restored site, banks were graded and backfilled to achieve designed cross-section profiles and boulders and large logs were added along several banks at each site to help deflect high flow away from the banks and stabilize cross-section morphology. In addition, the channel at each site was reconstructed to achieve a designed slope profile, which was stabilized by different combinations of rock vanes, rock weirs, riprap, and log weirs at the different sites.

***Figure 1.2**. Photograph of a representative restored stream. Boulders have been placed on the channel banks to prevent bank erosion and have been placed in the channel to prevent channel incision.*

For each of the study streams, catchment area (0.06 - 3.8 km^2) and land use were determined using GISHydro2000, an ArcView GIS-based software package developed to aid in hydrological analyses in Maryland (Moglen 2007). The application uses 30 m resolution digital elevation models (DEMs) to delineate catchments, and has land use data current to 2002. The percentage of riparian area occupied by impervious surface along each study reach was measured using Google

Earth (Google Inc., Mountain View, California, USA). Each stream was traced manually on satellite images and all impervious surfaces within 30 m of the stream were delineated and tabulated.

Land use in the catchments was predominantly a mix of forest and urban development. Agricultural land cover was variable but did not exceed 34% in any one catchment. Restored streams were all located in catchments with >30% impervious surface cover. The other 16 streams were located in catchments spanning a range of development, from 5%-75% impervious surface cover (Table 1.1). I divided non-restored streams into forested reference streams and urban streams by classifying all streams with at least 15% impervious surface cover in their catchments as urban. Most streams in catchments with >15% impervious cover show signs of biological impairment (Klein 1979, Jones and Clark 1987, Steedman 1988, Horner et al. 1997, Wang et al. 1997, Wang et al. 2000, Ourso and Frenzel 2003). However, urban streams in this study were not independently assessed as to their level of impairment (e.g., by measuring biotic indices, water quality, or channel stability indices), and therefore urban stream refers to any stream in a catchment with >15% impervious surface cover. The forested streams in this study have some development in their catchments and may have been impacted by agriculture in the last century. As such, they are best classified as least disturbed (Stoddard et al, 2006).

Table 1.1. Land use characteristics for study streams.

Stream (GPS Coordinates)	Type (Year restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Plum Creek (N 39°03’31”, W 76°35’17”)	Restored	1	2.4	35	11	62	2	0	100
Severn Run Tributary 1 (N 39°04’34”, W 76°37’07”)	Forested	2	2.1	31	10	55	14	0	100
Severn Run Tributary 2 (N 39°06’21”, W 76°39’03”)	Forested	1	0.5	14	4	79	0	1	95
South Fork Jabez Branch (N 39°03’60”, W 76°39’06”)	Urban	2	2.4	34	18	28	34	0	100
Broad Creek (N 39°00’05”, W 76°33’36”)	Urban	1	0.4	31	17	51	3	0	100

Table 1.1 cont.

Stream (GPS Coordinates)	Type (Year restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Cockey Creek (N 39°07’06”, W 76°30’48”)	Urban	1	0.6	71	48	26	3	0	100
Cypress Creek (N 39°04’33”, W 76°32’12”)	Urban	2	2.1	88	47	4	0	1	95
Harbor Center East (N 38°58’33”, W 76°32’31”)	Urban	1	0.3	91	76	1	0	11	87
Harbor Center West (N 38°58’29”, W 76°33’02”)	Urban	2	0.8	79	69	2	0	0	100
Herald Harbor (N 39°02’56”, W 76°34’39”)	Urban	1	0.06	54	26	34	0	2	95

Table 1.1 cont.

Stream (GPS Coordinates)	Type (Year restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Marley Creek Tributary (N 39°08’02”, W 76°37’03”)	Urban	1	0.7	73	34	20	2	11	63
Picture Spring Branch (N 39°05’33”, W 76°41’47”)	Urban	1	1.0	42	28	58	0	2	92
Sawmill Queenstown East (N 39°09’06”, W 76°39’39”)	Urban	2	3.8	50	18	29	16	0	100
Sawmill Queenstown West (N 39°09’07”, W 76°39’42”)	Urban	1	0.9	50	24	38	4	0	99
Spa Urban East (N 38°57’47”, W 76°30’29”)	Urban	1	0.2	95	61	0	0	7	77

Table 1.1 cont.

Stream (GPS Coordinates)	Type (Year restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Spa Creek Tributary at Hilltop Lane (N 38°57’49”, W 76°30’31”)	Urban	1	0.1	89	63	6	5	0	96
Elvaton (N 39°07’17”, W 76°37’19”)	Restored (2004)	2	1.4	82	32	13	3	10	67
Harundale (N 39°09’12”, W 76°36’23”)	Restored (2005)	1	1.1	81	42	5	0	0	100
Muddy Bridge Branch (N 39°10’32”, W 76°38’41”)	Restored (1997)	2	2.8	81	70	8	4	0	89
Spa Creek (N 38°58’23”, W 76°31’03”)	Restored (2001)	1	0.9	87	60	7	1	4	77

Table 1.1 cont.

Stream (GPS Coordinates)	Type (Year restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Tributary 9 of Sawmill Creek (N 39°10’54”, W 76°38’09”)	Restored (1994)	2	1.7	81	34	11	1	0	100
Tributary 10 of Sawmill Creek (N 39°10’58”, W 76°37’24”)	Restored (1998)	1	0.8	86	34	11	0	0	95
Weems Creek at Bristol Drive (N 38°59’11”, W 76°31’10”)	Restored (1997)	1	0.5	79	38	10	0	1	95
Cowhide Branch (N 38°59’30”, W 76°32’14”)	Restored (1994)	1	0.4	85	67	15	0	0	100
Weems Creek at Moreland (N 38°59’06”, W 76°31’37”)	Restored (1999)	1	0.2	85	69	10	0	0	99

Quantifying Channel Complexity

I sought a measure of channel complexity that accounted for channel attributes that are assumed to be important to ecological patterns and processes. For example, heterogeneous bed sediments and variation in depth created by irregular bedforms can enhance surface water flux into the hyporheic zone (Cardenas et al. 2004, Mutz et al. 2007, Hester and Doyle 2008). In addition, the presence of both deep, slow-flowing water (i.e., pools) and shallow rapidly-flowing water (i.e., riffles) increases habitat heterogeneity in streams, which is assumed to increase diversity of stream communities (Palmer et al. 2010). To capture these ecological attributes of stream channels, I attempted to design a measure of complexity that assessed overall variability in channel morphology.

I took an approach similar to that used by Bartley and Rutherfurd (2005) and used multiple metrics to assess variability in four aspects of channel morphology: i) cross-section profile, ii) longitudinal profile, iii) planform profile, and iv) bed sediment distribution (Table 1.2). It was important to measure all four aspects to assess overall channel complexity, because each aspect can vary independently in response to disturbance (Bartley and Rutherfurd 2005). For example, knowledge of the longitudinal profile variability (i.e., knowing how thalweg depth varies downstream) provides no information on the heterogeneity of the bed sediments.

The metrics were generated from channel surveys that included cross-section, long profile, planform, and grain size measurements (see below and Baker et al. (2011) for details on the survey protocol). Cross-section profiles were simplified by measuring only wetted width, maximum depth, and maximum velocity in order to increase the number of cross-sections sampled at each stream. Variation in maximum cross-section velocity was used as a measure of cross-sectional variability, because changes in cross-section shape drive changes in velocity through the principle of flow conservation. Variability in planform profile was described by a single metric, sinuosity.

All but two of the metrics I used have been previously described elsewhere (see Table 1.2). I developed two new metrics of variation in wetted width for this study by applying calculations to wetted width profiles (wetted width measured at successive points downstream, see Fig. 1.3) that were originally applied to longitudinal profiles. The first was the fractal dimension of the wetted width profile. Fractal dimension measures the crookedness of a line and is calculated using the program Vfractal (Nams 1996; www.nsac.ns.ca/envsci/staff/vnams/Fractal.htm). The fractal dimension can take a value between 1 and 2, with 1 indicating a straight line and 2 indicating a line with sufficient crookedness to completely fill a plane (Nams 1996). Bartley and Ruterfurd (2005) used fractal dimension as a metric of the variability in longitudinal and cross-section profiles. The second new metric of width variation was average width-profile concavity (AWC), calculated as:

$$AWC = \left(\frac{1}{n}\right)\left(\sum_{i=1}^{n}\left|\frac{d^2 w_i}{dx_i^2}\right|\right),$$

where n = number of cross sections, w = wetted width, and x = distance downstream from the top of the reach (Fig. 1.3). This equation substitutes cross-section wetted width for water surface elevation in the equation for average water surface concavity developed by Anderson et al. (2005) and applied by Gooseff et al. (2007) as a measure of channel complexity. The metric reflects the overall variation of the wetted channel width along the stream reach, i.e., the degree to which the stream channel changes from narrow to wide, and vice versa, between cross-sections throughout the study reach.

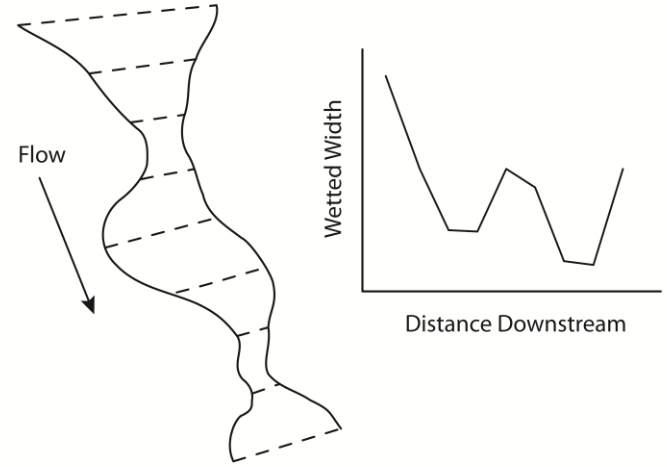


Figure 1.3. Planform view of hypothetical reach showing cross-section measurement points (dotted lines) and resulting wetted width profile. The wetted width profile was used to calculate coefficient of variation (CV) of width, width residual, fractal mean of wetted width, and average width-profile concavity (AWC). Concavity at a measurement point increases as the difference in wetted width between the measurement point and each adjacent measurement point increases and the difference in wetted width between the adjacent measurement points decreases, i.e., the line connecting three sequential measurement points becomes more peaked in the middle. Thus, AWC is higher for wetted width profiles that have more of a zigzag or saw tooth pattern.

Table 1.2. Complexity metrics used in this study, grouped by channel aspect.

Metric	Dimension	Equation	Description
Cross-section profile variation			
Coefficient of variation of width	-	$CVW = \left(\frac{s_w}{\overline{w}}\right)$	Standard deviation of widths scaled by mean width
Width residual[a]	-	$WR = \frac{\sum_{i=1}^{p}\left(\left\lvert w_i - \overline{w}\right\rvert I_p\right)}{\overline{w}}$	Sum of proportionally weighted deviations in width scaled by mean width (Baker et al. 2011)
Average width concavity[a]	$\frac{L}{L^2}$	$AWC = \sum_{i=1}^{n}\left(\left\lvert\frac{d^2 w_i}{dx_i^2}\right\rvert I_p\right)$	Proportionally weighted concavities at successive points along the width profile - modified from Anderson et al. (2005)
Fractal mean of width profile (D_{wwp})	-	Determined by simulation using the program Vfractal, with a window range of 0.25, random seed start of 1.0 and 30 divisions (Nams 1996; http://www.nsac.ns.ca/ envsci/staff/vnams/Fractal.htm)	Crookedness of width profile (see text). Width profile was detrended prior to analysis
Coefficient of variation of maximum cross-section velocity	-	$CVV = \left(\frac{s_v}{\overline{v}}\right)$	Standard deviation of maximum cross-section velocity scaled by mean maximum cross-section velocity

Table 1.2 cont.

Metric	Dimension	Equation	Description
Longitudinal profile variation			
Coefficient of variation of maximum cross-section depths	-	$CVD = \left(\frac{s_z}{\bar{z}}\right)$	Standard deviation of maximum cross-section depths scaled by mean maximum cross-section depth
Longitudinal roughness[a]	L	$LR = \sum_{i=1}^{n}\left[\left(z_{obs,i} - z_{pred,i}\right)I_p\right]$	Proportionally weighted deviations in thalweg elevation (z) from a straight line between the thalweg elevations at the top and bottom of the reach (Gooseff et al. 2007, Baker et al. 2011)
Average water surface concavity[a]	$\frac{L}{L^2}$	$AWSC = \sum_{i=1}^{n}\left(\left\lvert\frac{d^2 z_{w,i}}{dx_i^2}\right\rvert I_p\right)$	Proportionally weighted concavities at successive points along the water-surface profile (modified from Anderson et al. 2005)
Average thalweg concavity[a]	$\frac{L}{L^2}$	$AThC = \sum_{i=1}^{n}\left(\left\lvert\frac{d^2 z_i}{dx_i^2}\right\rvert I_p\right)$	Proportionally weighted concavities at successive points along the thalweg profile (Anderson et al. 2005, Baker et al. 2011)
Fractal mean of longitudinal profile (D_{lp})	-	Determined by simulation using the program Vfractal, with a window range of 0.25, random seed start of 1.0 and 30 divisions (Nams 1996; http://www.nsac.ns.ca/ envsci/staff/vnams/Fractal.htm)	Crookedness of thalweg elevation profile – used by Bartley and Rutherfurd (2005). Longitudinal profile was detrended prior to analysis

Table 1.2 cont.

Metric	Dimension	Equation	Description
Longitudinal profile variation cont.			
Standard deviation	L	$SD = \sqrt{\frac{1}{N}\sum_{i=1}^{n}\left(z_i - z_{\min}\right)^2}$	Standard deviation of thalweg elevations relative to the highest point in the thalweg profile (Bartley and Rutherfurd 2005). Longitudinal profile was detrended prior to analysis
Bed sediment distribution			
Heterogeneity	-	$Het = \frac{d_{84}}{d_{50}}$	84th largest particle relative to median particle size – larger values indicate a greater range of substrate sizes
Sorting	-	$Sort = \frac{\phi 84 - \phi 16}{2}$	Measures the standard deviation of the bed sediment size distribution (Briggs 1977 Bartley and Rutherfurd 2005)
Fredle index	L	$f_i = \frac{\sqrt{d_{16} d_{84}}}{\sqrt{\frac{d_{75}}{d_{25}}}}$	Measures the porosity of bed sediments (Lotspeich and Everest 1981)
Gradation coefficient	-	$s_{grad} = \frac{\left(\frac{d_{84}}{d_{50}} + \frac{d_{50}}{d_{16}}\right)}{2}$	Measures the spread of the bed sediment distribution (Bunte and Abt 2001)

Table 1.2 cont**.**

Metric	Dimension	Equation	Description
Bed sediment distribution cont.			
Sediment coefficient of variation	L^{-1}	$CV_s = \frac{\sqrt{\frac{d_{84}}{d_{16}}}}{d_{50}}$	Geometric standard deviation of bed sediment distribution relative to median particle size (Baker 2009)
Channel planform variability			
Sinuosity	-	$s = \frac{L}{L_s}$	Channel length relative to straight-line distance between top and bottom of reach

[a] Proportional weighting was calculated as half the distance between successive measurement points upstream and downstream of measurement point *i* relative to total reach length using, $I_p = \frac{\frac{(x_{i+1} - x_{i-1})}{2}}{L}$, where x = distance and L = reach length (Baker et al. 2011).

Field Surveys of Channel Complexity

The survey methodology is described by Baker et al. (2011), but is summarized here. Study reaches were established at each stream by measuring a length 15 times the estimated bankfull width. I divided the reach into at least 20 equally spaced sections by running at least 21 transects perpendicular to the stream. Along each transect I measured wetted width, maximum stream depth, and maximum flow velocity during baseflow conditions. The cross-section measurements were used to calculate the coefficient of variation (CV) of width, CV maximum depth, CV maximum velocity, width residual, AWC, and fractal mean of the wetted width profile (see Table 1.2). Grain size distributions within the wetted width of each reach were quantified by measuring between 600 and 1200 particles throughout the reach. Grain size distributions were used to calculate sorting, the gradation coefficient, the Fredle index, the sediment coefficient of variation, and sediment heterogeneity (see Table 1.2). I also surveyed the longitudinal profile of each reach. Measurement points were located at breaks in slope, and the channel bed elevation, water surface elevation, and water depth were recorded at each point. Longitudinal profile surveys were used to calculate longitudinal roughness, average water surface concavity, average thalweg concavity, standard deviation of depths, and fractal mean of the longitudinal profile (see Table 1.2). Sinuosity was calculated by dividing the reach length by the straight-line distance between the upstream and downstream points of the reach, both measured using aerial photographs.

Comparisons with other Study Sites

Data from Colorado study sites were provided by Baker et al. (2011) and came from six reaches on two streams. Sheep Creek (N 40^{o}55’48”, W 105^{o}38’16”) was located at an elevation of 2530 m and had minimal development in the catchment, though it was influenced by a small dam upstream and one reach was actively grazed by livestock. Spring Creek was located in an urbanized catchment in the town of Fort Collins (N 40^{o}30’50”, W 105^{o}4’7”) at an elevation of 1500 m. One reach was located in a municipal park, one reach was deliberately straightened, and one reach had extensively rip-rapped banks and grade control structures.

I also surveyed the literature to find papers that reported values of CV velocity, CV depth, and CV width or that reported enough information to calculate these metrics. Papers were acquired by first examining studies that had evaluated the effects of stream restoration on habitat heterogeneity (reviewed in Palmer et al. 2010). I also collected papers by searching Web of Science^{SM} (Thomson Reuters, New York City, USA) for the keywords: fish, habitat, transect, and stream. These keywords were chosen because I needed papers that measured width, depth, and velocity along multiple transects and so that I could limit the papers examined to those in which transects were established specifically for the purpose of assessing fish habitat in relatively small streams and rivers. I also looked through the citations of papers found on Web of Science^{SM} for additional relevant papers. I only included papers that measured at least five transects in a stream reach and reported either direct CV measures or that reported means with standard deviations or standard errors with sample sizes.

Statistical Analysis

To generate a comprehensive measure of channel complexity (Objective 1), I used principle component analysis (PCA) combined with a correlation table (Table 1.3) of the longitudinal profile, cross-section profile, and bed sediment distribution metrics to reduce the number of metrics used in subsequent analyses (see below). Both Maryland and Colorado data were included in this analysis. Where I had multiple metrics for one aspect of complexity (cross-section profile, longitudinal profile, and sediment distribution) I kept the metric that had the most explanatory power on the first two components (indicated by the magnitude of the vector in a PCA biplot) and eliminated metrics that were significantly correlated to that metric. I ran a second PCA with the reduced set of metrics to simplify the biplot and to visualize graphically how streams from the different regions grouped together. All additional analyses used this reduced set of metrics.

To determine whether urban development decreased the complexity of Maryland streams (Objective 2), I used canonical correlation analysis (CCA) to examine the relationship between land use variables (% forested, % impervious, and % agriculture) and the reduced set of complexity variables. Wilks' lambda (λ) was used to test the significance of relationships. To assess the effects of restoration on channel complexity in Maryland streams, I compared channel complexity metrics in restored streams to metrics in urban streams (Objective 3). I did not statistically test whether restored streams were different from forested streams due to insufficient statistical power (n = 3 forested streams). I used region (north or south Anne Arundel County) as a blocking variable in my analyses, because geologic differences between the regions (see Study Sites) suggested that channel morphology might differ between streams in these regions, and I indeed found that streams in the north and south had significantly different average geomorphic properties (MANOVA, $F = 3.7054_{(4, 15)}$, $p = 0.027$; Fig. 1.4). I compared all metrics in the reduced set using a fixed-effects MANOVA. I also tested for a difference in each aspect of complexity (cross-section profile, longitudinal profile, sediment distribution, and planform profile) between restored and urban streams using either ANOVA or MANOVA depending on whether each aspect was represented by one or multiple metrics. The Fredle index, the gradation coefficient, sediment standard deviation, sediment heterogeneity, and average water surface concavity were $\log_{10}$-transformed prior to analyses. All other variables met assumptions of homogeneity of variance and normality tested on residual variances calculated within groups. All statistical tests were run using R version 2.11.0 (R Foundation for Statistical Computing, Vienna, Austria).

To gain an understanding of the variability in channel complexity across streams of varying land use and between geographic regions (Objective 4), I compared values from the literature survey to the Maryland and Colorado data with plots of CV velocity vs. CV depth and CV width vs. CV depth.

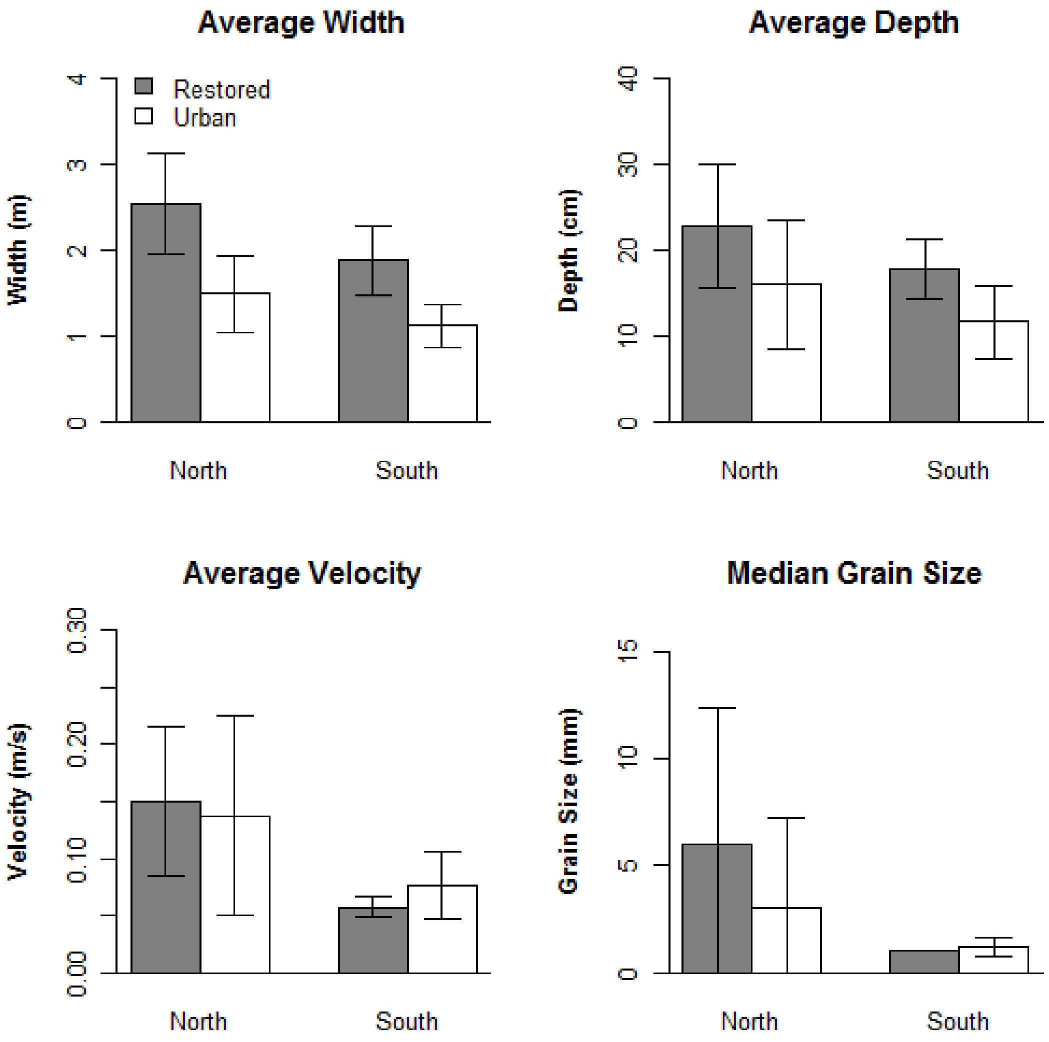


Figure 1.4. Average values of width, maximum depth, maximum velocity, and median grain size at northern restored (*n* = 5), northern urban (*n* = 8), southern restored (*n* = 4), and southern urban streams (*n* = 5) in Anne Arundel County, Maryland. Error bars are standard deviations.

RESULTS

Objective 1: Quantifying Channel Complexity

The first two components of the PCA using all complexity metrics explained 46% of the variance between streams (Appendix I). All of the sediment distribution metrics, except the Fredle index, clustered together on the biplot of the first two components, indicating high correlation among these metrics (see also Table 1.3). The Fredle index was significantly correlated only with the sediment coefficient of variation. Therefore, I chose the Fredle index and sediment sorting as the sediment metrics for further analyses. CV maximum velocity, CV width, and AWC were chosen as the width metrics for further analyses, because they were not significantly correlated with each other. Width residual was excluded, because it was significantly correlated with CV width and fractal mean of the width profile was excluded because it was significantly correlated with both AWC and CV width. Four of the six longitudinal profile metrics (CV depth, fractal mean, average thalweg concavity, and average water surface concavity) grouped together on the biplot and were significantly correlated. Longitudinal roughness and standard deviation of the longitudinal profile were significantly correlated with each other, and neither was significantly correlated with any other longitudinal profile metric. Therefore, I chose CV depth and longitudinal roughness as the longitudinal profile metrics for further analyses, because they had the most explanatory power along components 2 and 3, respectively.

Table 1.3. Correlation table of complexity metrics with both Maryland and Colorado streams included. Only correlations between metrics measuring the same aspect of channel morphologic variability are shown. Numbers in bold are significant at $\alpha = 0.05$. See Table 1.2 for metric names.

	CVW	WR	AWC	D_{wwp}	CVV	CVD	LR	AWSC	AThC	D_{lp}	SD	Het	Sort	f_i	s_{grad}	CV_s
CVW	1	**0.96**	0.17	**0.50**	0.02											
WR		1	0.20	**0.48**	0.05											
AWC			1	**0.71**	-0.28											
D_{wwp}				1	0.07											
CVV					1											
CVD						1	0.14	**0.38**	**0.47**	**0.63**	0.14					
LR							1	0.13	0.22	-0.09	**0.68**					
AWSC								1	**0.85**	**0.42**	0.003					
AThC									1	**0.42**	0.08					
D_{lp}										1	-0.07					
SD											1					
Het												1	**0.71**	-0.26	**0.67**	**0.59**
Sort													1	-0.28	**0.92**	**0.79**
f_i														1	-0.31	**-0.48**
s_{grad}															1	**0.86**
CV_s																1

The first two components of the PCA using the reduced set of metrics explained 48% of the variance between streams. No overall gradient of complexity was apparent in the biplot as different metrics of complexity pointed in opposite directions (Fig. 1.5). All Colorado streams had positive scores on component 1, and the three Spring Creek reaches grouped together closely. The three Sheep Creek reaches were separated along component 2. Two Maryland forested streams and two Maryland urban streams grouped closely with the Spring Creek reaches and two reaches of Sheep Creek. Seven of the nine Maryland restored streams grouped together in the top left quadrant of the biplot. In contrast, all but one of the streams in the lower left quadrant were urban streams.

Objective 2: Channel Complexity along an Urbanization Gradient

The first canonical function from the CCA of the relationship between land use variables and complexity metrics had a relatively high correlation coefficient, but explained only 12% of the shared variance in the complexity metrics and was not significant (Wilks' $\lambda = 0.17_{(24, 38.3)}$, $p = 0.203$; Table 1.4). I had predicted that complexity metrics would decline with increasing impervious cover, since it has been assumed that urban development reduces channel complexity. However, the weak relationship in the CCA showed this was not the case. This did not appear to be an artifact of insufficient statistical power, because many complexity metrics tended to increase with increasing impervious cover, as indicated by the sign of the complexity metric loadings on the first canonical function (Table 1.4).

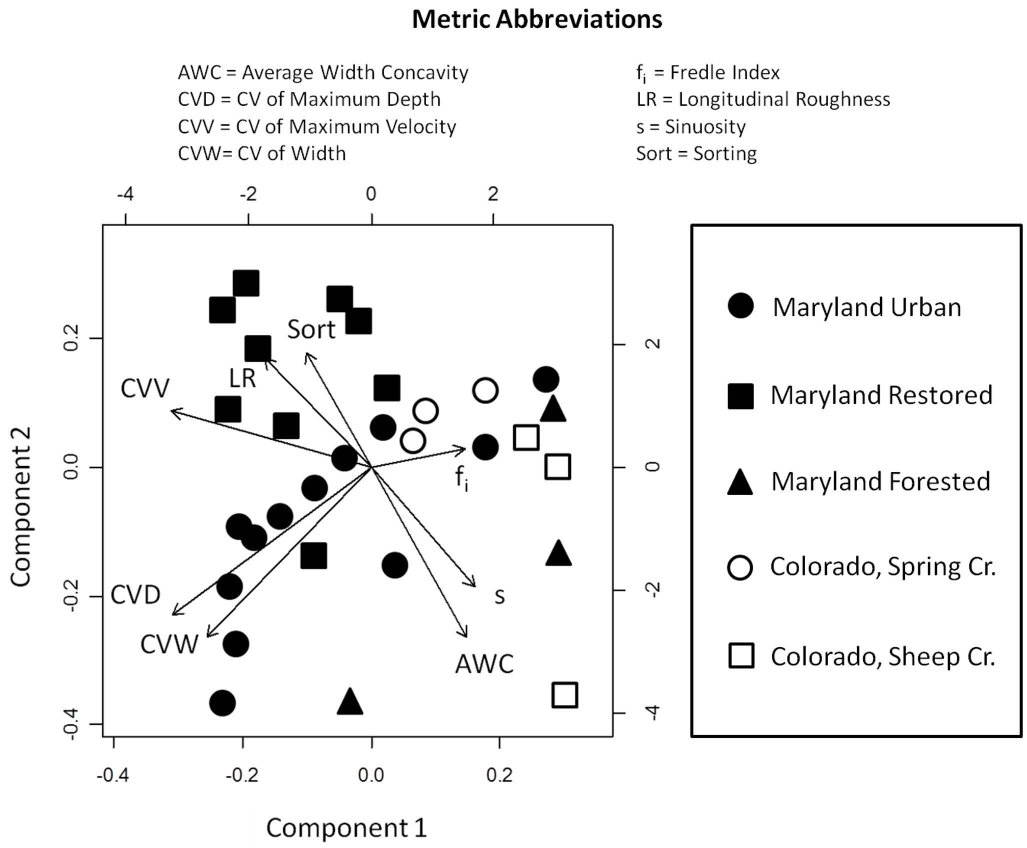


Figure 1.5. Biplot of components 1 and 2 from the PCA with the reduced set of complexity metrics. Arrows and black letters denote complexity metric vectors. Symbols represent different types of streams as specified in the legend.

Table 1.4. Results of the Canonical Correlation Analysis (CCA) between land use variables and complexity metrics using only Maryland streams. Loadings on only the first canonical function are presented. See Table 1.2 for metric abbreviations.

Canonical Function	Df 1	Df 2	Wilks' λ	p-value	R
CF 1	24	38.3	0.16	0.176	0.81
CF 2	14	28.0	0.47	0.556	0.70
CF 3	6	15.0	0.92	0.970	0.27

Variable	Coefficient
% Forest	0.0139
% Impervious	0.0075
% Agriculture	0.0082
CVW	-0.73
AWC	4.65
CVV	-0.41
CVD	0.19
LR	0.13
Sort	0.04
f_i	0.06
S	0.55

Objective 3: Channel Complexity in Restored vs. Urban Streams

The block by treatment interaction effect in the overall MANOVA of complexity data was not statistically significant ($F = 1.4322_{(8, 10)}$, $p = 0.29$), indicating that the complexity metrics in both northern and southern streams were responding similarly to restoration and I could interpret the main effects of block (geologic region) and treatment (restoration or urban) separately. The block effect was significant ($F = 8.0049_{(8, 10)}$, $p = 0.002$) and indicated that blocking in the analysis was useful. Restored streams were significantly different from urban streams overall

($F = 8.5205_{(8, 10)}$, $p = 0.001$). Similar to the overall results, the block by treatment interaction was not significant and the block effect was significant in separate analyses of longitudinal profile, cross-section profile, and bed sediment distribution variability. Restored streams had significantly different cross-section variability compared to urban streams ($F = 6.53_{(3,16)}$, $p = 0.004$), with restored streams having lower measures of CV width and AWC and higher measures of CV maximum velocity (Fig. 1.6). Restored streams were significantly different from urban streams in terms of longitudinal profile variation ($F = 8.6190_{(2,16)}$, $p = 0.003$) and marginally different in terms of sediment distribution ($F = 3.3231_{(2,17)}$, $p = 0.06$), but the direction varied between metrics and between northern and southern streams. The block effect was not significant for sinuosity, but this did not significantly change the interpretation of the main effect of restoration. Restored streams had significantly lower sinuosity than urban streams ($F = 6.0995_{(1, 18)}$, $p = 0.024$).

Values of complexity metrics for the three forested streams generally fell within the range of values seen at the restored and urban streams. However, values at the forested streams were more often on the lower end of the range, with one forested stream having the lowest values of CV velocity and Fredle index of any Maryland stream. Whether overall complexity in forested streams was significantly different from urban or restored streams was not tested due to low sample size of forested streams ($n = 3$).

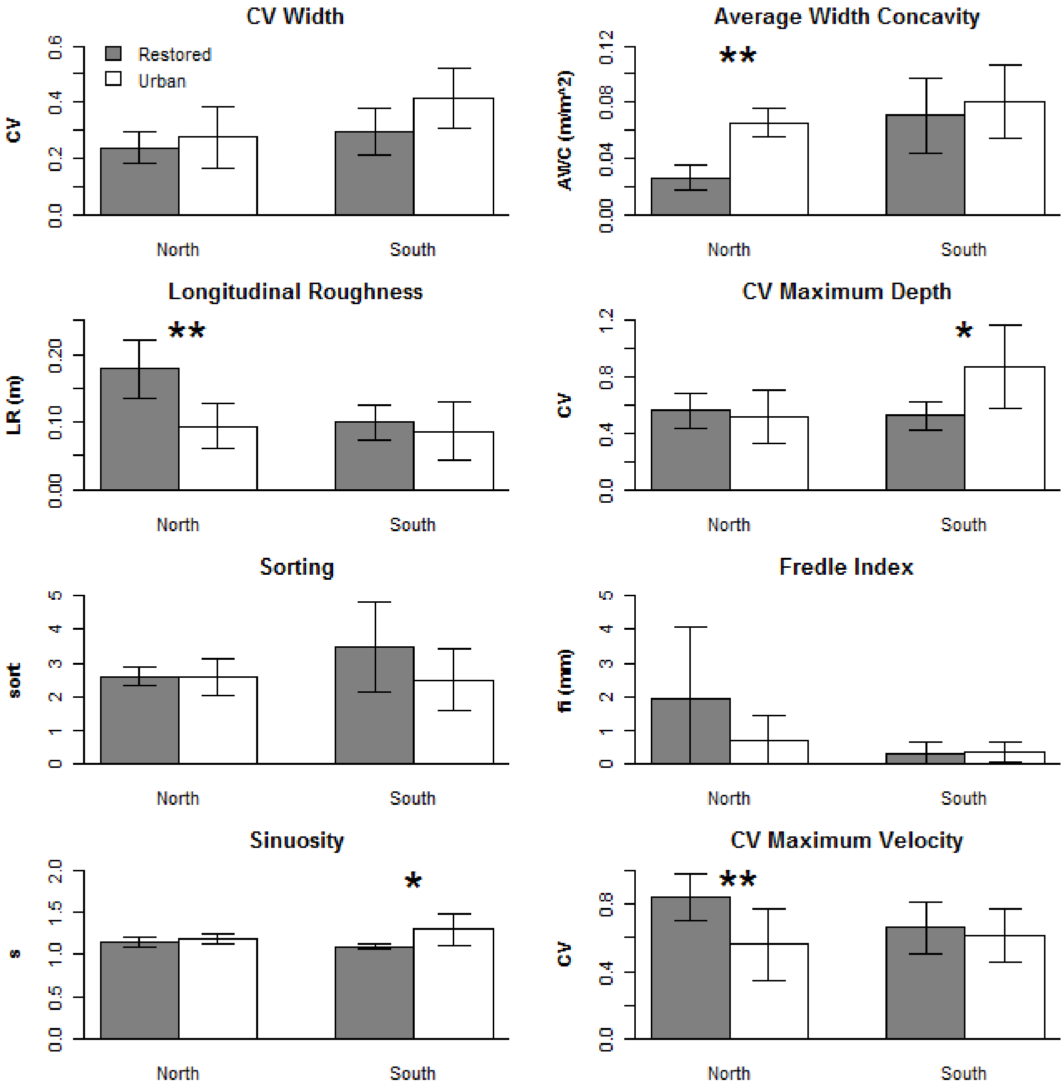


Figure 1.6. Comparison of individual complexity metrics for northern restored (*n* = 5), northern urban (*n* = 8, except longitudinal roughness, *n* = 7), southern restored (*n* = 4), and southern urban streams (*n* = 5) in Anne Arundel County, Maryland. ** above bars indicates a significant difference between restored and urban streams within that region (north or south) at $\alpha = 0.05$, * indicates significance at $\alpha = 0.1$. Error bars are standard deviations.

Objective 4: Geographic Range in Channel Complexity

My literature survey resulted in data on CV velocity and CV depth for 112 individual reaches and data on CV width and CV depth for 98 reaches (Figs. 1.7, 1.8). Catchment area of selected streams ranged from 7.3 to 84,000 km^2 and reach length from 10-2000 m. Most studies were located in temperate zones, but ranged from prairie and coastal streams to mountain streams. Land use was also variable across streams, ranging from nearly pristine to agricultural and urban dominated. Numerous studies also reported data for restored streams.

Streams from the literature mostly fell within the range of values for CV depth and CV width seen in the Maryland study streams (Fig. 1.8). One concrete channel in Florida, USA and numerous streams in urban catchments in Ohio, USA had lower values of both CV depth and CV width than the lowest values seen in the Maryland study streams (Annett 1998, Balanson et al. 2005). Very few streams had higher values of CV width and CV depth than the highest values seen in the Maryland study streams. In contrast, there were numerous reaches, primarily reaches restored with large-wood additions in Germany, that had higher values of CV velocity relative to the highest values seen in Maryland study sites (Gerhard and Reich 2000; Fig. 1.7).

About half of the Maryland urban streams ranked in the top 30% of streams for both CV depth and CV width, though none ranked in the top 30% for CV velocity. One Maryland forested stream and one Maryland restored stream ranked in the top 30% for CV width and the same forested stream and three different Maryland restored streams ranked in the top 30% for CV depth. Colorado streams ranked in the lower 50% for CV velocity and CV depth and generally ranked in the middle 50% for CV width, with the reach containing rip-rap and grade control structures on Spring Creek ranking in the top 30%.

Figure 1.7. Biplot of CV velocity vs. CV depth showing the range in values for these variables determined from the literature survey relative to the range observed in the current study. CV velocity includes measures of CV maximum velocity and CV average velocity. CV depth includes measures of CV maximum depth, CV bankfull depth, and CV average depth. Different types of streams in the current study are not differentiated on the plot. Benvenuto et al. (2008) report data for one undisturbed stream in the Tuscany region, Italy. Brunke et al. (2003) report data for two reaches on one river, one undisturbed and one levied, and eight tributaries and floodplain streams in the southern Alps, Switzerland. Champion and Tanner (2000) report data for one agricultural stream in New Zealand. Dolinsek et al. (2006) report data for two undisturbed streams in New Brunswick, Canada. Duehr et al. (2006) report data for two buffered and two non-buffered reaches on one agricultural stream in Iowa, USA. Gerhard and Reich (2000) report data for a mix of undisturbed, channelized, and restored reaches on two rivers in Germany. Jähnig et al. (2010) report data for disturbed and restored stream pairs throughout central and southern Europe. Kondolf et al. (2000) report data for one undisturbed stream in Yellowstone National Park,

USA. Negishi and Richardson (2003) report data for two undisturbed reaches, one disturbed reach, and one restored reach on a stream in British Columbia, Canada. Parkyn et al. (2009) report data for three reaches on one forested/pine plantation stream in New Zealand. Pretty et al. (2003) report data for 12 paired channelized and restored agricultural streams in the UK. Rahel and Hubert (1991) report data for two reaches on one undisturbed mountain to prairie stream in Wyoming, USA. Riley and Fausch (1995) report data for six paired restored and control streams in relatively undisturbed streams in northern Colorado, USA. Shields et al. (1994) report data for three agricultural streams and one forested reference stream in Mississippi, USA. Torgersen and Close (2004) report data for one river in northeast Oregon, USA. Van Zyll de Jong et al. (1997) report data for one forested, previously logged stream in Newfoundland, Canada.

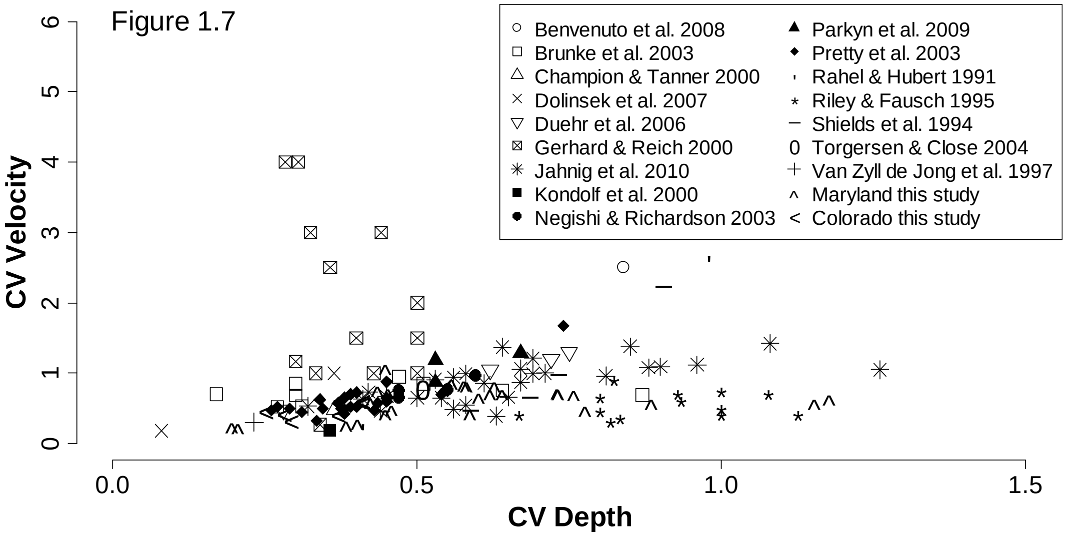


Figure 1.7

Figure 1.8. Biplot of CV width vs. CV depth showing the range in values for these variables determined from the literature survey relative to the range observed in the current study. CV width includes measures of CV bankfull width and CV wetted width. CV depth includes measures of CV maximum depth, CV bankfull depth, and CV average depth. Different types of streams in the current study are not differentiated on the plot. Abes and Agostinho (2001) report data for three reaches on a buffered pasture stream in southern Brazil. Annett (1998) reports data for one canal in Florida, USA. Balanson et al. (2005) report data for streams along a gradient of urbanization in northeast Ohio, USA. Brown (2003) reports data for one undisturbed stream in Vermont, USA. Champion and Tanner (2000) report data for one agricultural stream in New Zealand. Fausch and Northcote (1992) report data for undisturbed and woody debris removal reaches on two streams in British Columbia, Canada. Gerhard and Reich (2000) report data for a mix of undisturbed, channelized, and restored reaches on two rivers in Germany. Hall et al. (2009) report data for one reservoir outlet river in Nevada, USA. Hampton and Berry (1997) report data for nine reaches on one agricultural river in South Dakota, USA. Henderson and Letcher (2003) report data for three streams with a mix of forest and pasture in Massachusetts, USA. Kondolf et al. (2000) report data for one undisturbed stream in Yellowstone National Park, USA. Loomis et al. (1999) report data for 11 reaches on one grazed prairie/agricultural river in South Dakota, USA. Overton et al. (1994) report data for grazed and non-grazed reaches on two streams in the Sierra Nevada Mountains, USA and for two undisturbed streams in central Idaho, USA. Robinson et al. (2005) report data for three burned and three unburned, otherwise undisturbed streams in central

Idaho, USA. Shields et al. (2006) report data for three non-restored and one restored reach on a forested/pasture mix stream in Mississippi. Torgersen and Close (2004) report data for one river in northeast Oregon, USA. Van Zyll de Jong et al. (1997) report data for one forested, previously logged stream in Newfoundland, Canada.

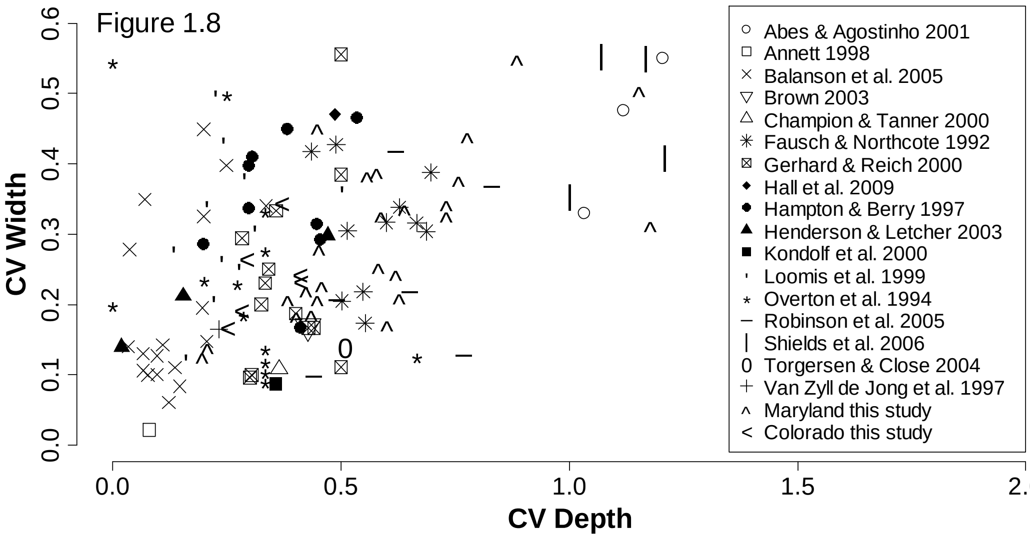


Figure 1.8

DISCUSSION

I defined channel complexity as overall variation in the cross-section profile, longitudinal profile, planform profile, and bed sediment distribution. By using multiple metrics for these four aspects and applying PCA, I assessed how channel complexity differed among streams of varying land use across geographic regions. Assessment of different aspects of channel complexity was used previously to investigate changes in channel geomorphic diversity caused by increased sediment loading (Bartley and Rutherfurd, 2005). My application of the approach led to unexpected results regarding urban land use and the effects of restoration on channel complexity. The metrics I measured did not combine to a single gradient of complexity. Streams with high values of one metric often had low values of other metrics. Thus, my approach demonstrated the limitation in using any single variable as an indicator of overall channel complexity. In addition, because different measurements of the same attribute sometimes yielded conflicting results, the approach highlighted the importance of explicitly defining channel complexity and the methodology used to measure it.

Urbanization and Channel Complexity

Based on the assumption that urbanization leads to a simplification of stream channels (Walsh et al. 2005), I hypothesized that non-channelized streams in urban catchments would have lower channel complexity than forested reference streams. My surveys of streams across a gradient of catchment urbanization (collectively referred to as urban streams, regardless of the state of degradation) did not support this. First, the PCA results showed that there was no single gradient of complexity, and it was not possible to define streams in urban catchments as having lower or higher complexity than forested streams. Second, I did not find a significant relationship between land use and complexity metrics in Maryland streams (Table 1.4), which I would have expected if streams in more urbanized catchments had lower complexity. Multiple metrics, including CV width, CV maximum velocity, and sorting, tended to increase with impervious surface cover (Table 1.4). Third, many urban streams in Maryland ranked in the top third of sites surveyed in the literature for two complexity metrics (CV width and CV depth), which included many near-pristine streams.

Research over the past 40 years has shown that the mean value of many aspects of channel morphology changes predictably in response to urbanization. Channels generally become wider and more deeply incised in response to urban development (Wolman 1967, Hammer 1972, Arnold et al. 1982, Booth 1990, Pizzuto et al. 2000, Hardison et al. 2009). In contrast, my findings suggest that variability in some aspects of channel morphology (e.g., CV depth, CV width) does not respond predictably to increased urban development in the catchment and that overall variability in channel morphology (i.e., channel complexity) is not consistently lower in streams draining urban catchments compared to forested reference streams (Fig. 1.5). Some aspects of channel morphology did vary consistently between urban and forested streams, as Maryland urban and forested streams grouped separately on the PCA biplot (Fig. 1.5). However, there were multiple complexity metrics that were not different in urban and forested streams, suggesting that channel morphology can adjust to urbanization in highly variable ways, i.e., urbanization does not always reduce variability of all aspects of complexity.

My results contrast with previous research reporting lower channel complexity in urban streams compared to reference streams (Davis et al. 2003, Grimm et al. 2005, Walsh et al. 2005, Gooseff et al. 2007). However, much of this work has focused on streams that were concrete-lined or deliberately straightened and studies have typically been limited to 5 urban streams or less. I surveyed a relatively large number of streams in urban catchments (n = 12 in Maryland alone) that were not directly channelized or straightened by human activities and found that complexity was relatively high in these streams relative to forested reference streams.

My results for the urban streams do not appear anomalously high nor results from the forested streams anomalously low, as all streams fell within the range of variability for several complexity metrics seen across diverse geographic regions from the literature survey. Instead, these results indicate that urban land use in a catchment does not necessarily lead to channel simplification when channels are not directly manipulated, and this conclusion should be relevant beyond my focal study region, given the fact that my study streams are within the range of variability in channel complexity seen across streams from diverse regions (Figs. 1.7 and 1.8).

Rather than assuming channels in urban catchments are geomorphically simplified, I found that it is important to measure multiple aspects of channel complexity. One of the Colorado stream reaches has been purposefully straightened, and this reach had relatively low values for many complexity metrics, including the lowest values of sinuosity, sorting, and three longitudinal profile metrics (standard deviation, fractal mean, and average thalweg concavity). However, this reach had relatively high values for some metrics, highlighting the fact that even channelized streams can have high complexity in certain attributes. I also found that metrics of the same aspect of complexity were often uncorrelated, for example CV width and AWC (Table 1.3). AWC measures sequential variation in the width profile, whereas CV width measures average deviation from the mean width. Similarly, longitudinal roughness measures sequential variation in the longitudinal profile, whereas CV depth measures average deviation from the mean depth. The unique information provided by each metric was important in separating streams, as seen in the PCA biplot (Fig. 1.5). Maryland urban streams were separated from each other along a gradient of CV width and CV depth but were separated from Maryland restored streams by a gradient of longitudinal roughness and AWC. By measuring multiple aspects of complexity, I gained a better understanding of how channel morphologic variability responds to urbanization when channels are not constrained.

Factors Influencing Channel Complexity within and across Geographic Regions

There has been a great deal of recent discussion on the difficulties of identifying appropriate reference sites or the appropriate reference condition (Stoddard et al. 2006, Bernhardt and Palmer 2007, Herlihy et al. 2008, Baattrup-Pedersen et al. 2009, Hawkins et al. 2010). Today, almost all ecosystems are impacted by humans to some extent and urbanization is a rapidly growing land use change (Paul and Meyer 2001). The forested streams in my study have some urban development in their catchments and were likely impacted by agriculture in the past. Thus, while I classified these streams as my reference sites, they are in fact the least-disturbed sites, and it is possible that current and past land use has caused a reduction in channel complexity from historical (e.g. 300 ya) levels. While I cannot dismiss this possibility (much of the Mid-Atlantic region was impacted by agriculture in the last two centuries), comparison with non-urbanized streams from other regions suggests that the levels of complexity I found in the Maryland reference streams are not unusually low (i.e., they were within the range of channel complexity seen across diverse geographic regions). Further, the lack of a clear gradient in channel complexity along an urbanization gradient supports my conclusion that urban development does not necessarily lead to overall reductions in channel complexity. Previous research has also provided some evidence that complexity is not always reduced by urbanization. Hammer (1972) observed that small urban streams in Pennsylvania had particularly variable cross-sectional areas. Kang and Marston (2006) found that urbanization only affected sinuosity patterns through direct manipulation such as rip-rap installations and channelization.

The lack of clear and consistent relationships between complexity and urbanization probably reflects the large number of factors that can vary across catchments even if they have comparable levels of urban development. For example, the urban streams I surveyed had a well-forested buffer, even when impervious cover in the catchment exceeded 60% (Table 1.1). Riparian vegetation exerts a strong influence on channel morphology independent of the level of catchment urbanization (Hession et al. 2003), and a forested buffer along an urban stream could maintain or even enhance channel complexity via increased inputs of wood. Increasing impervious surface cover in catchments has been linked to flashier, more powerful floods and increased bank erosion (Hammer 1972, Dunne and Leopold 1978, Arnold et al. 1982, Booth 1990, Hardison et al, 2009), which could increase lateral movement of the channel across the landscape and transport trees, fallen logs, and other debris (e.g., discarded lumber and concrete, shopping carts, tires, etc.) into the channel more rapidly. Instream wood and urban debris in channels can increase channel complexity by creating variations in scour and fill patterns (Robison and Beschta 1990, Abbe and Montgomery 1996, Buffington et al. 2002), however this process requires that stabilization structures, which are common in urban streams, do not prevent bank erosion (Segura and Booth 2010). The urban streams I surveyed in Maryland were not deliberately stabilized and the process of increased inputs of wood and urban debris may explain the trend toward increased complexity, but remains to be tested.

Previous work has shown that channel morphology responds differently to urbanization in different geoclimatic settings (Utz and Hilderbrand 2011), and it is possible that the relatively high complexity of the Maryland streams in this study - all located on the Coastal Plain - reflects a unique response of Coastal Plain streams to urbanization. In comparison to Piedmont streams (the neighboring Physiographic province in Maryland), Coastal Plain streams suffer less geomorphic degradation with increasing urbanization (Utz and Hilderbrand 2011). This differential response has been attributed to the finer sediments and lower topographic relief of Coastal Plain streams, which may buffer changes in sediment supply and hydrologic patterns associated with urban development (Utz and Hilderbrand 2011). Thus, it is possible that channel complexity is also less severely impacted by urbanization in Coastal Plain streams relative to streams from other regions, but this remains to be tested.

Restoration and Channel Complexity

I hypothesized that complexity in restored streams would be higher compared to non-restored streams in urban catchments. Analysis of the Maryland sites showed that restored streams differed significantly from urban streams in terms of overall complexity, but this difference was non-directional; there was no consistent overall complexity gradient. Restored streams did have somewhat higher CV velocity compared to urban streams and northern restored streams had higher longitudinal roughness compared to northern urban streams (Fig. 1.6). Both CV velocity and longitudinal roughness have been used previously in attempts to explain patterns of transient storage and macroinvertebrate diversity (Brooks et al. 2002, Negishi and Richardson 2003, Gooseff et al. 2007, Baker et al. 2011), suggesting that restoration may have had some benefit for hydraulic retention and habitat quality. However, restored streams had similar or lower values of many complexity metrics compared to urban streams, including sediment sorting and sinuosity, which have also been used in attempts to explain patterns of transient storage and biodiversity (DeMarch 1976, Robertson and Milner 2006, Gooseff et al. 2007; Fig. 1.6).

Without pre-restoration data, it is impossible to conclude with certainty that restoration did not improve overall channel complexity of some stream reaches compared to their more degraded state. It is also possible that restoration increased channel complexity during high flows, as I did not measure channel complexity during storm events. Previous work has shown that large boulders and logs such as those added during restoration can increase hydraulic retention during storms, but the effect has been less pronounced during storm events than at baseflow (Webster et al. 1987, Muotka and Laasonen 2002, Dewson et al. 2007). Nonetheless, the high variability in complexity I observed across urban streams makes it unlikely that the geomorphic restoration approach used on my study sites consistently increased overall channel complexity across a variety of flow levels.

Biological Implications

Restoration has often attempted to increase the physical heterogeneity of perceived degraded streams, because theory predicts that species diversity should increase when physical heterogeneity increases. However, recent synthesis suggests that even when indicators of habitat heterogeneity are improved by restoration, macroinvertebrate diversity often does not increase in response (Palmer et al. 2010, Louhi et al. 2011, but see Miller et al. 2010). The lack of response of macroinvertebrate diversity is likely due to processes operating at the catchment scale that alter flow regimes, degrade water quality, and prevent dispersal (Miller et al. 2010, Palmer et al. 2010, Sundermann et al. 2011). I emphasize the importance of measuring multiple aspects of physical complexity in stream channels, to ensure that overall heterogeneity has improved with restoration. The results of my comprehensive measure of channel complexity and previous studies in urban catchments (Tullos et al. 2009, Violin et al. 2011) suggest that restoration may not always result in increased channel complexity, in part because physical heterogeneity may not be a limiting factor for biodiversity in non-channelized urban streams. Therefore, invertebrate recovery may not have the opportunity to be influenced by channel complexity if other catchment-scale factors are limiting.

Chapter 2: Effects of stream restoration on bed sediment disturbance and bank stability

ABSTRACT

Disturbance of streambed substrates and scouring of streambanks during high flow events has received considerable attention by ecologists because such physical disturbance events are believed to influence the structure and dynamics of benthic communities. Ecologists have also focused on the impacts of land use on benthic communities and the role of restoration in moderating those impacts. If restoration projects stabilize the streambed this could confer increased resistance and resilience of benthic communities in the face of flow disturbances. To date however, studies have not determined whether channel stability restoration projects actually decrease streambed and streambank disturbance. I tested the hypothesis that the sediment disturbance regime in urban streams with channel stability restoration is modified compared to the regime in an urban and forested stream. Bed sediment disturbance and changes in streambank cross-section morphology were quantified in four restored streams, one urban stream, and one forested stream in the Coastal Plain physiographic province of Maryland. Sediment disturbance in relation to in-stream flow obstructions was a particular focus because obstructions in the form of large boulders and wood are commonly used in restoration. Each stream was divided into units corresponding to pools, riffles, and runs and grain size distribution was measured in each unit. The influence of flow obstructions in each unit was then quantified using

two metrics, and bed sediment disturbance patterns were quantified by measuring scour/fill depth over the course of two separate high-flow periods (July-August and September 2008) in multiple units within each stream. Bank erosion and channel incision rates were quantified by multiple repeated cross-section measurements between stakes on a floodplain at each stream. Flow obstruction metrics were positively correlated with grain size and negatively correlated with scour/fill magnitude during each flow period. I propose an explanation for these patterns whereby flow obstructions increase local turbulent forces that entrain deposited sand, coarsen bed material, and lead to a reduction in disturbance probability during elevated flow. The proposed mechanism was found to be consistent with experimental data, particularly in streams with a wide grain size distribution, because variability in proportion of mobile bed sediments among units was positively correlated with grain size at the reach scale (~50 m stream length) for each flow period ($r = 0.66$, $p = 0.16$ for July/August; $r = 0.95$, $p < 0.01$ for September). These results suggest that stability restoration has altered bed sediment disturbance patterns within local stream units – an outcome with potential consequences for benthic communities. Restored streams had lower rates of channel widening and scour compared to the urban stream during one sampling period. This suggested that restoration may have achieved the stated goals, but lack of pre-restoration data makes it unclear whether the effects of restoration were ecologically beneficial. Similarity in erosion rates between armored and non-armored cross-sections suggests bank armoring had little effect on bank erosion. One stream was found to have significantly aggraded during the study period due to an input of upstream-derived sediment passing through the reach, suggesting that watershed-scale processes may overwhelm effects of stream restoration on channel stability.

INTRODUCTION

The role of disturbance in structuring benthic communities has been a focus of stream ecology for many years, and the effects of sediment-mobilizing flows have been particularly well studied (Resh et al. 1988, Lake 2000, Stanley et al. 2010). Previous research has shown that disturbance of the streambed can influence stream communities directly by scouring organisms from the bed and significantly reducing biomass (Fisher et al. 1982, Power and Stewart 1987, Grimm and Fisher 1989). This sediment disturbance often creates patches of streambed that are severely disturbed and patches that are relatively stable and can act as refugia (Lancaster and Hildrew 1993, Palmer et al. 1996). Such heterogeneity further influences the distribution and composition of stream benthic communities (Sedell et al. 1990, Townsend et al. 1997, Matthaei et al. 1999, Effenberger et al. 2006).

Despite extensive research on flow-sediment-organism interactions in streams, little work has focused on potential interactions between natural disturbances and land use change or human alteration of stream channels (Stanley et al. 2010). Anthropogenic modifications of landscapes, such as those associated with urbanization and channel restoration, interact with or directly alter a stream's flow disturbance regime which may lead to changes in streambed dynamics. Previous research has clearly demonstrated the impacts of land use on flow variability (Poff et al. 2006). In particular, changes in the discharge regime accompanying urban development have been well quantified (Dunne and Leopold 1978, Paul and Meyer 2001, Walsh et al. 2005).

The influence of stream restoration on flow, sediment, and ecological processes is less well studied but is emerging as an important field (e.g., Fellows et al. 2006, Udy et al. 2006, Bukaveckas 2007**,** Roberts et al. 2007**,** Kaushal et al. 2008, Shields 2009, Tullos et al. 2009). Several recent studies have focused on how channel restoration influences reach-scale patterns and processes such as invertebrate biodiversity (see reviews by Miller et al. 2010, Palmer et al. 2010) and discharge or channel form (e.g., Kondolf et al. 2001, Thompson 2003, Smith and Prestegaard 2005, Simon et al. 2007), but less work has been done at smaller within-reach scales.

The influence of restoration at smaller scales within a reach and among substrate patches within a reach could also affect ecological patterns and processes. If restoration projects influence small-scale spatial heterogeneity in streambed stability, this could have significant implications for benthic communities, particularly those in urban streams that face flow disturbances. Understanding such effects will be critical to the successful management of stream ecosystems in modified landscapes (Collier and Quinn 2003), as well as the design of restoration projects.

The mechanisms by which restoration could alter patterns of bed sediment disturbance are numerous but implementation practices such as the addition of boulders and logs to the stream channel are likely to exert effects through changes in near-bed flow (Miller and Kochel 2010). Boulders and logs are commonly added along the stream banks and to the bed to stabilize channels, protect stream banks from erosive forces, and maintain a desired channel gradient (Shields et al. 2003). However, these structures act as flow obstructions that may alter patterns of scour and fill on the streambed. For example, in-stream restoration structures such as vanes and weirs can create local scour and alter sediment transport (Borg et al. 2007, Bhuiyan et al. 2009). Similar processes can occur around boulders and logs added during restoration and studies of these processes could serve as a guide for predicting patterns of scour and fill following placement of flow obstructions after restoration (Beschta 1986, Shields et al. 1995, Abbe and Montgomery 1996, Hassan and Woodsmith 2004, Shields et al. 2004, Webb and Erskine 2005, Borg et al. 2007).

Reducing channel incision and bank erosion are often stated as goals for stream stabilization restoration projects (Shields et al. 2003, Radspinner et al. 2010), particularly in urban streams where these channel adjustments in response to changing flow patterns are often perceived as threats to infrastructure. However, channel movement through these geomorphic processes may be critical for maintaining aquatic and riparian habitat in natural systems (Florsheim et al. 2008), suggesting that the ecological implications of stabilization projects may extend beyond their potential effects on benthic invertebrates to include habitat and associated ecosystem processes at local and reach scales. Thus, I designed a research project to study patterns of bed sediment disturbance (scour and fill) and channel movement processes (e.g., bank erosion, channel incision) in restored, forested, and urban streams. I tested the hypothesis that both sediment disturbance patterns and channel movement rates in urban streams with channel stability restoration projects are modified compared to a non-restored urban stream and a forested stream.

In this chapter I first use previous research to make and justify predictions regarding how flow obstructions placed in the channel during restoration may alter bed sediment composition, scour and deposition patterns (hereafter termed scour/fill) at the patch scale, and how patch-scale processes may scale up to affect reach-scale sediment disturbance patterns. I tested my predictions using an empirical study of scour/fill patterns during two separate high flow events. I also present results of channel movement (e.g., bank erosion, channel incision) measurements made in stream cross-sections that were stabilized using common restoration techniques and that were not stabilized.

PREDICTIONS

Prediction 1

The bed sediments in geomorphic units (defined as individual riffles and runs) containing flow obstructions should be coarser on average than in units without flow obstructions.

Flow pattern and resulting sediment scouring around bridge piers have been well studied for multiple decades (Laursen and Toch 1956, Melville and Coleman 2000). Obstructions such as bridge piers cause flow separation and acceleration and generate turbulent eddies that scour sediments upstream of the obstruction, on the sides of the obstruction, and downstream of the obstruction, where a scour hole is created whose depth is dependent on the sediment size among other factors (Melville and Raudkivi 1977, Chiew and Melville 1987, Dargahi 1990, Kirkil and Constantinescu 2010). Similarly, an obstruction extending into the flow from the channel bank creates a flow convergence and scour hole on the side of the obstruction (Thompson 2006, Thompson and McCarrick 2010). Logs and other in-stream structures projecting above the streambed, including restoration structures, similarly alter flow patterns that can cause scouring of bed sediments (Shields et al. 1995, Mutz 2000, Shields et al. 2004). These objects act as flow obstructions, and should have similar effects as bridge piers, though exact prediction of flow patterns around these obstructions is difficult due to flow complexities related to larger-scale geomorphic features and the non-uniform shapes and spatial arrangement of the obstructions within the channel. Qualitatively, obstructions should produce local flow constrictions and separations and generate turbulent eddies that will entrain bed material. The entrainment of bed material could create scour holes around the obstructions or could cause a coarsening of the bed material if there are larger grains on the bed that are not entrained by the local turbulent forces (Melville and Sutherland 1988, Kothyari et al. 2007). Bed sediment coarsening (i.e., development of an armor layer) has been observed around flume-simulated bridge piers where there is a wide distribution of sediment sizes and has been shown to decrease scour depths during flows in which sediment transport does not exceed excavation rate of the scour hole (Melville and Sutherland 1988, Dey and Raikar 2007).

Prediction 2

Bed sediments are more likely to remain stable during floods in patches with nearby obstructions than in areas without obstructions.

Basic sediment transport theory holds that coarser sediments require greater shear stress to be mobilized. Therefore, if prediction 1 holds and sediments in units with obstructions are coarser than areas without obstructions, the sediments in units with obstructions will not be mobilized over a broader range of flows and will more likely be stable during low to moderate flows. This prediction will not hold at flows sufficient to mobilize the largest grains found on the bed.

Prediction 3

Spatial variability in bed sediment disturbance at the patch scale should be higher in units with flow obstructions.

As discussed in prediction 1, flow obstructions can cause local flow accelerations that create scour holes or coarsen bed sediments. However, in the case of bridge piers, a depositional area also forms downstream of the scour hole, as turbulent eddies lose energy and can no longer transport entrained sediment from the scour hole (Kirkil and Constantinescu 2010). Similarly, obstructions extending into the flow from channel banks create a wake zone and depositional area immediately downstream of the obstruction (Thompson 2006, Thompson and McCarrick 2010). These studies suggest that complicated flow patterns will be established during high flow in units containing flow obstructions, creating variability in the magnitude of scour/fill between patches.

Prediction 4

At the reach scale, the area of bed sediment disturbance will be lower and spatial variability higher in streams with greater channel area affected by flow obstructions.

As discussed in prediction 2, I expected units with flow obstructions to have more stable patches than units without obstructions and as discussed in prediction 3, I expected variability at the patch scale to be higher in units with obstructions. Therefore, I predicted that the total area of bed sediment disturbance within the reach (i.e., amount of sediment that did not remain stable during a high flow event) would be lower and variability in bed sediment disturbance between units higher in streams with increased area affected by flow obstructions.

Prediction 5

a) Channel movement rates (measured as bank erosion, cross-sectional widening, and thalweg scour) will be reduced in restored streams relative to a forested and urban stream and b) armored banks (banks lined with boulders) within restored streams will show reduced bank erosion relative to unarmored banks.

The restoration design aims of channel stability restoration projects in urban systems are to create a stable channel form and reduce bank erosion and channel incision (Shields et al. 2003, Radspinner et al. 2010). Therefore, if projects are effective, they should show reduced channel movement rates, particularly in comparison to non-restored urban streams. Stability restoration is also predicted to reduce channel movement rates relative to forested systems, because bank erosion and channel migration are natural processes and help maintain habitat diversity (Florsheim et al. 2008). In particular, lining banks with boulders is predicted to reduce bank erosion because this is the purpose of armoring the banks.

METHODS

Study Sites

The study was conducted from 2007 to 2011 in 1st and 2nd order streams in the Coastal Plain region of Maryland (N 39°03’00”, W 76°37’00”). Measurements of bed sediment disturbance and bank erosion were made in six streams, including four restored streams, one forested stream, and one urban stream (Table 2.1). Although the forested and urban stream had similar urban cover (31% and 34%, respectively) the urban stream had higher impervious cover (18% compared to 10% for the forested stream) and received urban runoff from multiple direct stormwater drains upstream of the study site, whereas the forested stream did not receive direct urban stormwater runoff. These differences distinguished the forested and urban stream, but it is recognized that the urban stream was only mildly impacted by urbanization. Despite large variation in the amount of impervious cover between watersheds, all of the study streams had well developed riparian corridors (% forested within 30 m of the stream > 75%, see Table 2.1). All four restoration projects were stability restoration projects, involving a combination of channel manipulation and bank stabilization activities. At each restored site, banks were graded and backfilled to achieve designed cross-section profiles and boulders and logs were added along several banks at each site to help stabilize cross-section morphology. In addition, the channel at each site was manipulated to achieve a designed slope profile, which was stabilized by different combinations of rock vanes, rock weirs, riprap, and log vanes at the different sites. At the forested and urban sites, the only flow obstructions in the study reaches were logs that had fallen into the channel.

The length of the study **reach** in each stream was approximately 15 times the bankfull width (50 – 100 m). For sub-reach scale measurements, each reach was divided into sub-reach **units** based on three criteria: flow type (riffle, run, scour pool, dammed pool, or cascade), dominant sediment size (sand, gravel, cobble, or bedrock), and a visual estimate of the amount of flow within the unit that was influenced by flow obstructions (0-25%, 25-50%, 50-75%, or 75-100%). Thus, there were 80 possible unique sub-reach unit types in each reach (5 flow types x 4 sediment sizes x 4 obstruction categories), but there were no more than 11 unit types in any one reach. Unit length averaged 2 m and always spanned the entire wetted width of the channel.

Bed Sediment Size Composition

To test prediction 1, that bed sediments should be coarser in units with flow obstructions compared to units without flow obstructions, I measured grain size distributions in all units in which bed sediment disturbance was measured (see below). Grain size distributions were determined by measuring the diameters of 100 randomly chosen grains in each unit surveyed.

Table 2.1. Land use characteristics for study streams. This table is a subset of the sites listed in Table 1.1

Stream (GPS Coordinates)	Type (Year restored if restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Severn Run Trib. 1 (SRT) (N 39°04’34”, W 76°37’07”)	Forested	2	2.1	31	10	55	14	0	100
S. Fork Jabez Br. (JAB) (N 39°03’60”, W 76°39’06”)	Urban	2	2.4	34	18	28	34	0	100
Spa Crk. (SPA) (N 38°58’23”, W 76°31’03”)	Restored (2001)	1	0.9	87	60	7	1	4	77
Trib. 9 (T9) (N 39°10’54”, W 76°38’09”)	Restored (1994)	2	1.7	81	34	11	1	0	100
Trib. 10 (T10) (N 39°10’58”, W 76°37’24”)	Restored (1998)	1	0.8	86	34	11	0	0	95
Cowhide (CB) (N 38°59’30”, W 76°32’14”)	Restored (1994)	1	0.4	85	67	15	0	0	100

Bed Sediment Disturbance Measurement Using Scour Chains

Prediction 2, that bed sediments are more likely to be stable in units with obstructions, was tested by using scour chains to measure the amount of scour/fill that occurred in localized **patches** of the streambed (0.36 m^2 square segments) during two periods of high flow. Metal-link scour chains (60 cm long, 3 cm links) were installed vertically in the streambed and surveyed before and after high-flow events (Colby 1964, Laronne et al. 1994, Matthaei et al. 1999). If scour occurs during high flow, fewer links will be buried vertically in the sediment and thus the length of newly exposed chain served as a measure of scour depth. Conversely, if deposition occurs, the chains will be buried under sediment and the sediment depth above chains served as an indicator of deposition depth. Scour chains also indicated the process of scour followed by deposition if they are buried and fewer links are buried vertically in the sediment (Laronne et al. 1994).

I installed scour chains in a grid pattern (1 chain length apart and at least 20 cm away from streambanks) in each unique riffle and run sub-reach unit in all streams; 5-8 sub-reach units were surveyed with scour chains in each study stream. Chains were installed to a bed depth of approximately 50 cm. Chain length above the bed was measured before and after a series of high flows in July-August 2008 (i.e., one measurement of scour/fill depth across multiple high flow events) and a single high flow event in September 2008 (Fig. 2.1). I quantified the magnitude of scour/fill at each relocated chain by measuring the depth of sediment covering chains or the length of newly exposed chain. Each chain was given a single value for magnitude of scour/fill, with positive values indicating depth of deposition, negative values indicating depth of scour, and 0 indicating no net change in bed elevation. Discharge during the high flow events at the urban stream was taken from a USGS gaging station (01589795). At the remaining sites, pressure was monitored continuously with a HOBO U20 Water Level Data Logger (Onset Computer Corp., Pocasset, MA, USA) and discharge was estimated using pressure-transducer rating curves.

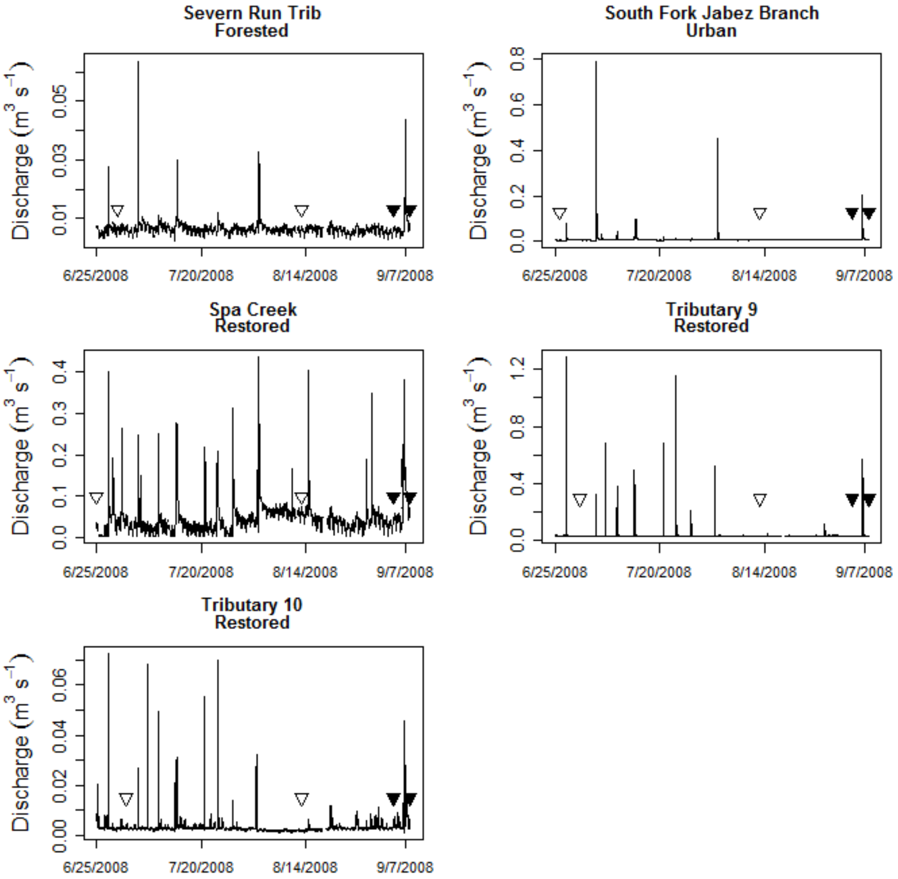


Figure 2.1. Hydrographs over the July-August and September sampling periods at five of the six sites. South Fork Jabez Branch data are from the USGS. Open triangles mark the beginning and end of scour chain deployment during the July-August period and closed triangles mark the beginning and end of scour chain deployment during the September sampling period.

Flow Obstruction Metrics

Predictions 1-4 deal with how flow obstructions are expected to alter bed sediment distribution and movement patterns from the patch to the reach scale. Therefore, it was necessary to define flow obstructions and quantify the degree to which bed sediment patches around each scour chain were influenced by flow obstructions. Flow obstructions were defined as coarse woody debris and cobble and boulder-sized rocks that projected from the streambed above the water surface at baseflow and that persisted in the channel through the duration of the experiment. I used two different metrics to quantify the influence of flow obstructions. The first was the number of flow obstructions that were directly upstream, downstream, and cross-stream from each chain and within 60 cm of each chain (60 cm was the spacing between chains). This metric was termed the **number of flow obstructions** and ranged from 0-4 for each chain. The second metric was the proportion of a 60x120 cm rectangle centered on each chain that was occupied by flow obstructions. This metric was termed the **flow obstruction area**. I chose these metrics because the greater the number of flow obstructions around a chain, the more likely the chain is to be located in an area influenced by the turbulence generated by a flow obstruction.

Channel Movement Rates

Prediction 5, that channel movement rates would be reduced in restored streams relative to the forested and urban stream and that armored banks would have lower bank erosion rates relative to non-armored banks, was tested using repeated measurements of permanent cross-sections at each unique unit type in each stream.

Cross-sections were marked by stakes on a floodplain - one on each side of the stream. I surveyed cross-section profiles in 2007, 2009, and 2011 at all sites, except at Tributary 9 where cross-sections were surveyed in 2007, 2008, and 2011. Total cross-section area and bank area were calculated from survey data using the computer program XSPRO (Hardy et al. 2005). Stream bank area was defined as the area between the stream bank at baseflow and the bankfull elevation, bounded by vertical lines drawn at the intersection of the bank with the baseflow and bankfull elevations. Area of bank erosion was calculated as the difference in bank area between successive surveys and was expressed as a daily rate of cross sectional area loss or gain. I also calculated total cross-section area changes (channel widening) and the amount of incision or aggradation of the maximum cross-section channel depth (thalweg scour) between successive surveys. Within two of the restored sites (Spa Creek and Tributary 10 of Sawmill Creek), I surveyed multiple cross-sections in areas where stream banks were lined with boulders (armored banks) and multiple cross-sections in areas where stream banks were not lined with boulders (non-armored banks). This allowed for comparison of bank erosion and channel widening rates between armored and non-armored sections of the same channel.

Statistical Analysis

I used multiple linear correlation to test whether the median grain size (D_{50}) was correlated with flow obstruction metrics as per prediction 1. I also used multiple linear correlation to test whether flow obstruction metrics could explain patterns of scour/fill across all sites for the two different flow periods (July-August and September 2008). This analysis was run for individual chain data and for unit averages of scour/fill and flow obstruction metrics and tested prediction 2. To examine prediction 3, that variability in scour/fill was greater in units with more obstructions, I calculated the average absolute difference in magnitude of scour/fill between adjacent chains within units. This served as a measure of the spatial heterogeneity of scour/fill magnitude within an individual unit. Multiple linear correlation was also used to test whether flow obstruction metrics were related to patterns in the spatial heterogeneity metric.

To test prediction 4, I estimated bed sediment disturbance on the reach scale by calculating the proportion of chains in the stream that did not remain stable (proportion mobile). As an estimate of variability on the reach scale, I calculated the standard deviation (SD) of the proportion mobile between units within a stream. I tested the null hypothesis that the proportion mobile and SD proportion mobile were not different between restored sites and non-restored sites (forested and urban grouped together) using one-way ANOVA. I also ran simple correlation of both proportion mobile and SD proportion mobile with reach-scale average flow obstruction metrics. These analyses indicated that grain size might control the proportion mobile at this scale, and I therefore examined correlations between an estimate of grain size (D_{84}) and proportion mobile and SD proportion mobile as well.

To test Prediction 5, I compared estimates of bank erosion, channel widening, and thalweg scour rates between the urban stream and restored streams and between armored and non-armored cross-sections within two restored streams (Spa Creek and Trib 10). Comparisons were made for both the periods 2007-2009 and 2009-2011. I tested the null hypothesis that there was no difference in bank erosion, channel widening, and thalweg scour rates rates between streams using separate one-way ANOVAs with streams as treatments. Experimental units for bank erosion were individual stream banks that eroded during the study period and were individual cross-sections for channel widening and thalweg scour. When there was a significant treatment effect, I computed contrasts between the urban stream and each restored stream to determine whether individual restored streams differed from the urban stream. The null hypothesis that bank erosion and channel widening rates were not different in armored compared to non-armored cross-sections was also tested using separate one-way ANOVAs. Bank erosion data were log-transformed prior to analysis to meet assumptions of normality and homogeneity of variance. All statistical tests were run in R version 2.11.0.

RESULTS

Discharge Patterns

The July-August and September scour chain measurement periods had different flow patterns (Fig. 2.1). At least three discharge peaks occurred in each stream during the July-August flow period and chains were measured at least 72 hours after the recession of the last discharge peak. Only one flow event occurred during the September sampling period and chains were sampled within 48 hours of peak discharge and in most cases flow was still elevated above baseflow. Therefore, the July-August period represents a condition of multiple high flow events followed by a period of baseflow, whereas the September period represents the direct effects of one high flow event.

Prediction 1: Bed Sediment Distribution in Units with and without Obstructions

Flow obstruction metrics explained a significant amount of the variability in median grain size between habitat units ($r^2 = 0.489$, $F = 17.3_{(2, 32)}$, $p < 0.01$). Number of obstructions and flow obstruction area were both positively correlated with D_{50} across all habitat units at all streams. In addition, most correlations between flow obstruction metrics and D_{50} within streams were positive with two exceptions (number of flow obstructions at the urban stream and flow obstruction area at Cowhide Branch).

Prediction 2: Scour/fill Patterns in Units with and without Obstructions

Recovery of chains was relatively high – approximately 80% for each sampling period. Unrecovered chains were most likely buried rather than scoured completely, as the maximum scour depth seen during the experiment was 10 cm, much shorter than the chain length installed in the bed (50 cm). The proportion of missing chains in each number of flow obstructions category (0, 1, 2, 3, and 4) was approximately similar to the proportion of chains in each flow obstructions category, suggesting no flow obstruction bias in missing scour chains.

The magnitude of scour/fill was highly variable between chains within individual units (Tables 2.2 and 2.3). The flow obstruction metrics explained a small but significant amount of this variability between individual chains for both flow events, but the proportion of variance explained for the July-August period was higher than for September ($r^2 = 0.071$, $F = 7.495_{(2, 168)}$, $p < 0.01$ for July-August; $r^2 = 0.023$, $F = 3.019_{(2, 169)}$, $p = 0.05$ for September). The number of flow obstructions and flow obstruction area were negatively correlated with scour/fill magnitude. Plots of the average scour/fill magnitude (± 95% CI) across all chains for each value of number of flow obstructions (0-4) showed that the negative relationship was due to chains with more obstructions near them being more likely to be stable and to have less deposition. The confidence interval for average scour/fill magnitude only overlapped 0 for the 3 and 4 obstruction categories in July-August and only for the 4 obstructions category in September, although the slope of the trendline was greater in July-August (Fig. 2.2). The significant negative trend between flow obstruction metrics and scour/fill magnitude held at the unit scale only for the July-August period ($r^2 = 0.122$, $F = 3.369_{(2, 32)}$, $p = 0.05$ for July/August; $r^2 = 0.054$, $F = 1.981_{(2, 32)}$, $p = 0.15$ for September).

Prediction 3: Variability in Bed Sediment Disturbance

The spatial heterogeneity in scour/fill magnitude within units showed no significant relationship with the flow obstruction metrics ($r^2 = 0.0$, $F = 0.3859_{(2, 29)}$, $p = 0.68$ for July/August; $r^2 = 0.032$, $F = 1.509_{(2, 29)}$, $p = 0.24$ for September).

Prediction 4: Reach Scale Analyses

There was no general trend for restored streams to be different from the forested or suburban streams in terms of the proportion mobile on the reach scale for either period ($F = 2.4839_{(1,4)}$, $p = 0.19$ for July-August; $F = 1.731_{(1,4)}$, $p = 0.26$ for September), nor variability in proportion mobile between units for either period ($F = 1.2877_{(1,4)}$, $p = 0.32$ for July-August; $F = 1.1238_{(1,4)}$, $p = 0.35$ for September). The flow obstruction metrics were highly correlated on the reach scale ($r = 0.97$), therefore I only used the number of obstructions when examining relationships between flow obstructions and reach-scale bed sediment disturbance patterns.

Spatial variability in proportion bed sediment mobile (measured as standard deviation in proportion mobile between units) on the reach scale was not significantly correlated with the number of flow obstructions for either period ($r = 0.56$, $p = 0.25$ for July-August; $r = 0.64$, $p = 0.17$ for September). The correlation between average proportion mobile and the average number of obstructions on the reach scale was also not significant for the July-August period ($r = -0.58$, $p = 0.22$; Fig. 2.3). In contrast, there was a significant negative relationship between average proportion mobile and the average number of obstructions during the September high flow event ($r = -0.81$, $p = 0.05$). However, this negative trend appeared to be driven by differences in grain size between streams, because the streams with larger grain size (measured as D_{84}) tended to have a higher value of number of obstructions, and a partial correlation test indicated the relationship between the reach-scale proportion mobile during the September high flow event was not significant when accounting for the effect of grain size ($p = 0.26$). In addition, grain size (measured as D_{84}) was significantly negatively correlated with proportion mobile on the reach scale for both periods ($r = -0.82$, $p = 0.04$ for July-August; $r = -0.86$, $p = 0.03$ for September; Fig. 2.4). In contrast, the variability in proportion mobile between units within a stream showed a positive relationship with grain size and was significant for the September high flow event ($r = 0.66$, $p = 0.16$ for July/August; $r = 0.95$, $p < 0.01$ for September; Fig. 2.4).

Table 2.2. Average and standard deviation (SD) in scour/fill magnitude within habitat units and across all habitat units within streams for the July-August period. Adjacent variability is the average difference between adjacent chains within a habitat unit.

		Unit Scour/Fill (cm)			Stream Averages (cm)		
Stream	Unit	Average	SD	Adjacent Variability	Average	SD	Adjacent Variability
South Fork Jabez Branch (Urban)	1	4	5.3	6	3	3.0	2
	2	5	3.0	4			
	3	1	-	-			
	4	2	0.5	0			
	5	3	2.2	3			
	6	7	1.4	2			
	7	1	4.2	5			
	8	4	2.8	4			
Severn Run Tributary 1 (Forested)	1	7	4.2	6	3	3.9	4
	2	3	1.8	2			
	3	5	5.7	8			
	4	0	-	-			
	5	-1	4.2	6			
	6	4	4.4	4			
Spa Creek (Restored)	1	2	1.3	2	1	2.4	3
	2	0	2.5	3			
	3	1	2.3	5			
	4	-1	2.5	2			
	5	2	2.5	4			
Tributary 9 (Restored)	1	2	1.2	2	2	2.3	2
	2	1	0	0			
	3	4	0.6	1			
	4	2	2.5	2			
	5	2	1.7	1			
	6	2	4.2	3			
Tributary 10 (Restored)	1	4	2.8	4	3	4.0	3
	2	4	4.8	3			
	3	2	3.9	5			
	4	3	5.6	3			
	5	3	1.5	-			
	6	1	2.4	2			
Cowhide Branch (Restored)	1	6	2.9	3	4	3.0	3
	2	5	2.5	4			
	3	7	2.9	5			
	4	3	1.7	2			
	5	3	4.4	7			

Table 2.3. Average and standard deviation (SD) in scour/fill magnitude within habitat units and across all habitat units within streams for the September period. Adjacent variability is the average difference between adjacent chains within a habitat unit.

		Unit Scour/Fill (cm)			Stream Averages (cm)		
Stream	Unit	Average	SD	Adjacent Variability	Average	SD	Adjacent Variability
South Fork Jabez Branch (Urban)	1	-2	7.1	10	2	2.6	2
	2	4	1.6	2			
	3	2	1.6	2			
	4	1	1.6	1			
	5	3	1.0	1			
	6	4	0	0			
	7	3	3.9	5			
	8	3	-	-			
Severn Run Tributary 1 (Forested)	1	3	4.9	7	2	3.2	3
	2	1	2.1	3			
	3	5	0.7	1			
	4	0	-	-			
	5	-1	0.7	1			
	6	6	2.8	-			
Spa Creek (Restored)	1	2	3.0	3	1	2.8	2
	2	1	1.3	1			
	3	1	1.6	2			
	4	0	0.7	0			
	5	3	4.3	3			
Tributary 9 (Restored)	1	-1	1.0	1	0	2.0	1
	2	0	0.0	0			
	3	5	5.1	5			
	4	1	0.7	1			
	5	0	1.2	1			
	6	1	1.1	1			
Tributary 10 (Restored)	1	1	1.4	2	1	2.1	2
	2	1	0.8	1			
	3	0	2.4	3			
	4	0	1.3	2			
	5	4	4.1	3			
	6	1	0.8	1			
Cowhide Branch (Restored)	1	7	1.0	1	3	4.8	3
	2	8	1.4	2			
	3	0	10.1	11			
	4	3	2.6	2			
	5	1	3.5	3			

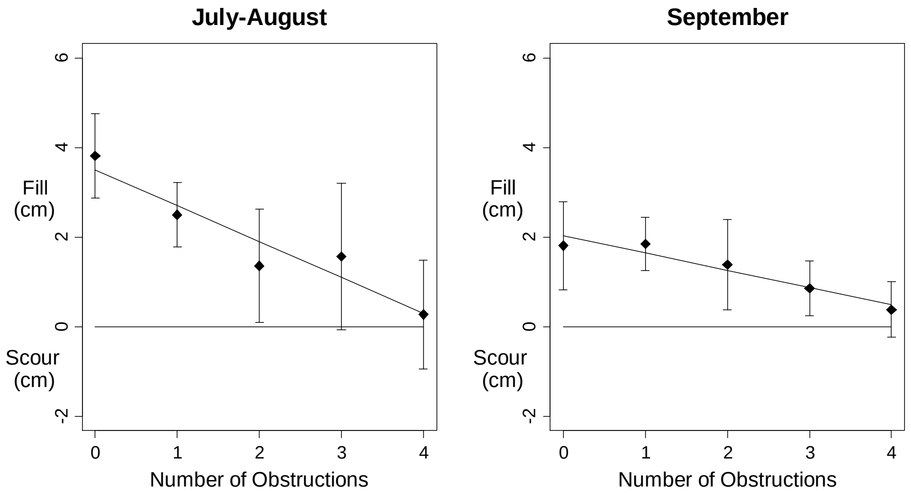


Figure 2.2. Average scour/fill magnitude vs. number of obstructions for all chains in the July-August and September periods. Positive values indicate fill, negative values indicate scour, with 0 indicating stability. Error bars are 95% confidence intervals.

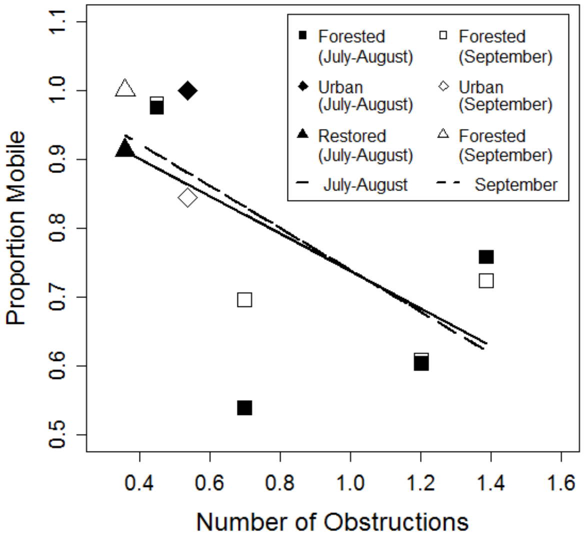


Figure 2.3. Proportion of chains recording scour or fill within a reach vs. reach-averaged number of obstructions per scour chain. Data are for all six study streams in both the July-August and September sampling periods. Lines are best-fit linear regressions through all six streams, fitted separately for the July-August (r = -0.58, p = 0.22) and September (r = -0.81, p = 0.05) sampling periods.

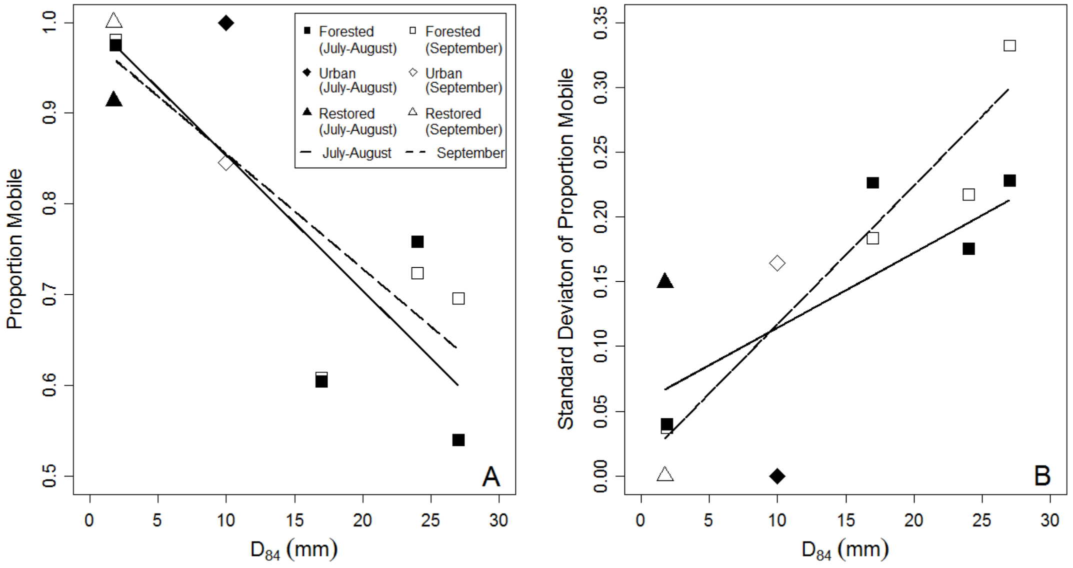


Figure 2.4. Relationships between grain size and A) reach-scale proportion mobile and B) variability in proportion mobile for all six study streams in both the July-August and September sampling periods. Lines are best-fit linear regressions for all six streams, fitted separately for the July-August (r = -0.82, p = 0.04 for proportion mobile; r = 0.66, p = 0.16 for standard deviation of proportion mobile) and September (r = -0.86, p = 0.03 for proportion mobile; r = 0.95, p < 0.01 for standard deviation of proportion mobile) sampling periods.

Prediction 5: Channel Movement Rates

Bank erosion did not differ significantly between streams in either the 2007-2009 or 2009-2011 sampling periods ($F = 1.538_{(4, 50)}$, p = 0.21 for 2007-2009; $F = 1.144_{(4, 47)}$, p = 0.35 for 2009-2001; Fig. 2.5). There was a significant effect of stream on channel widening rates and thalweg scour in the 2007-2009 period ($F = 4.653_{(4, 42)}$, p < 0.01 for channel widening; $F = 6.634_{(4, 42)}$, p < 0.001 for thalweg scour). The effect of stream on channel widening was also significant in the 2009-2011 period and was marginally significant for thalweg scour during this period ($F = 2.784_{(4, 37)}$, $p = 0.04$ for channel widening; $F = 2.38_{(4, 37)}$, $p = 0.07$ for thalweg scour).

Contrasts between the urban stream and restored streams indicated that the significant stream effect in the 2007-2009 period was primarily driven by Cowhide Branch, which showed aggradation through most of the channel during this period and had significantly lower channel widening rates and thalweg scour compared to the urban stream during this period ($p < 0.001$ for both channel widening and thalweg scour; Fig. 2.5). Tributary 9 also had significantly lower channel widening compared to the urban stream during 2007-2009, but did not differ significantly in terms of thalweg scour ($p = 0.01$ for channel widening; $p = 0.41$ for thalweg scour; Fig. 2.5).

Cowhide Branch also had significantly lower channel widening and thalweg scour than the urban stream in the 2009-2011 period, though the magnitude of the difference was much smaller compared to the 2007-2009 period ($p < 0.01$ for channel widening; $p = 0.01$ for thalweg scour; Fig. 2.5). Spa Creek also had significantly lower channel widening and thalweg scour compared to the urban stream during this period ($p = 0.04$ for channel widening; $p = 0.01$ for thalweg scour; Fig. 2.5).

Bank erosion did not differ significantly between armored and non-armored stream banks at Tributary 10 during either sampling period ($F = 0.197_{(1, 20)}$, $p = 0.66$ for 2007-2009; $F = 0.1947_{(1, 18)}$, $p = 0.66$ for 2009-2011; Fig. 2.6). Bank erosion was significantly different between armored and non-armored stream banks at Spa Creek during the 2009-2011 period, but armored banks showed higher erosion rates than non-armored banks ($F = 8.384_{(1, 16)}$, $p = 0.01$; Fig. 2.6). Bank erosion at Spa Creek did not differ significantly between armored and non-armored banks in the 2007-2009 period ($F = 0.0377_{(1, 16)}$, $p = 0.85$; Fig. 2.6).

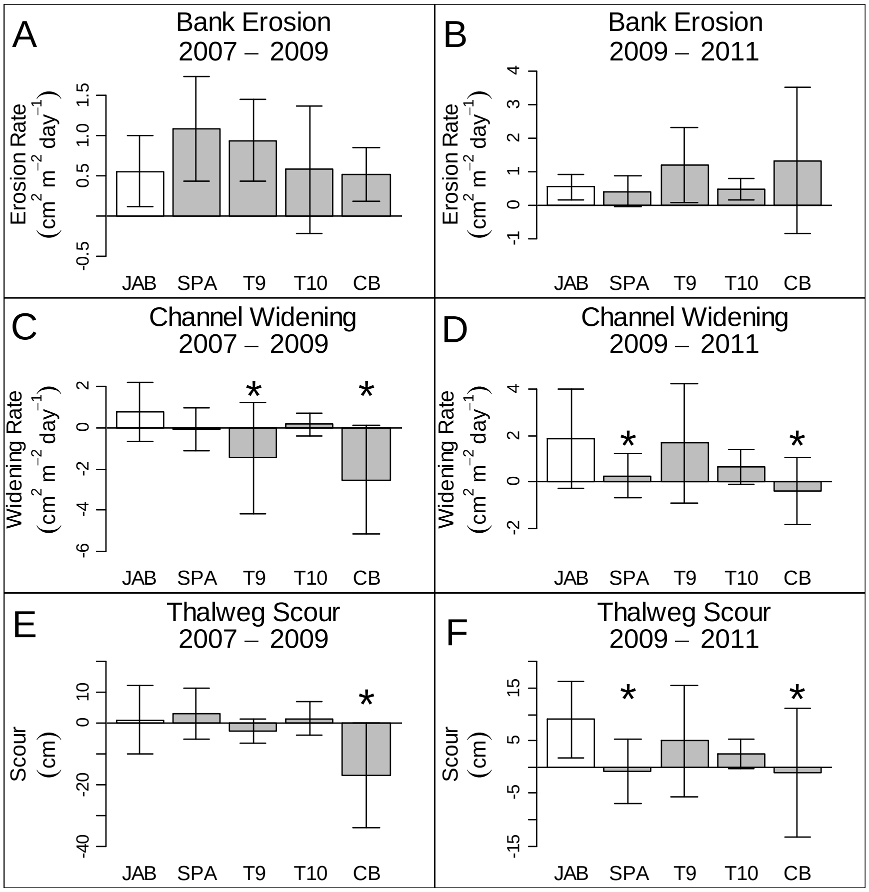


Figure 2.5. Cross-sectional channel changes at study sites from 2007-2009. JAB = South Fork Jabez Branch (urban – open bars), SPA = Spa Creek (restored), T9 = Tributary 9 (restored), T10 = Tributary 10 (restored), CB = Cowhide Branch (restored). * indicates that the restored stream was significantly different from the urban stream at $\alpha = 0.05$. Error bars are standard deviations. A, B) Average bank erosion rate for banks that showed erosion. C, D) Average cross-section widening rates. Positive values indicate cross-section area increased, negative values that cross-section area decreased. E, F) Average thalweg scour. Positive values indicate that channel depth increased, negative values that channel depth decreased.

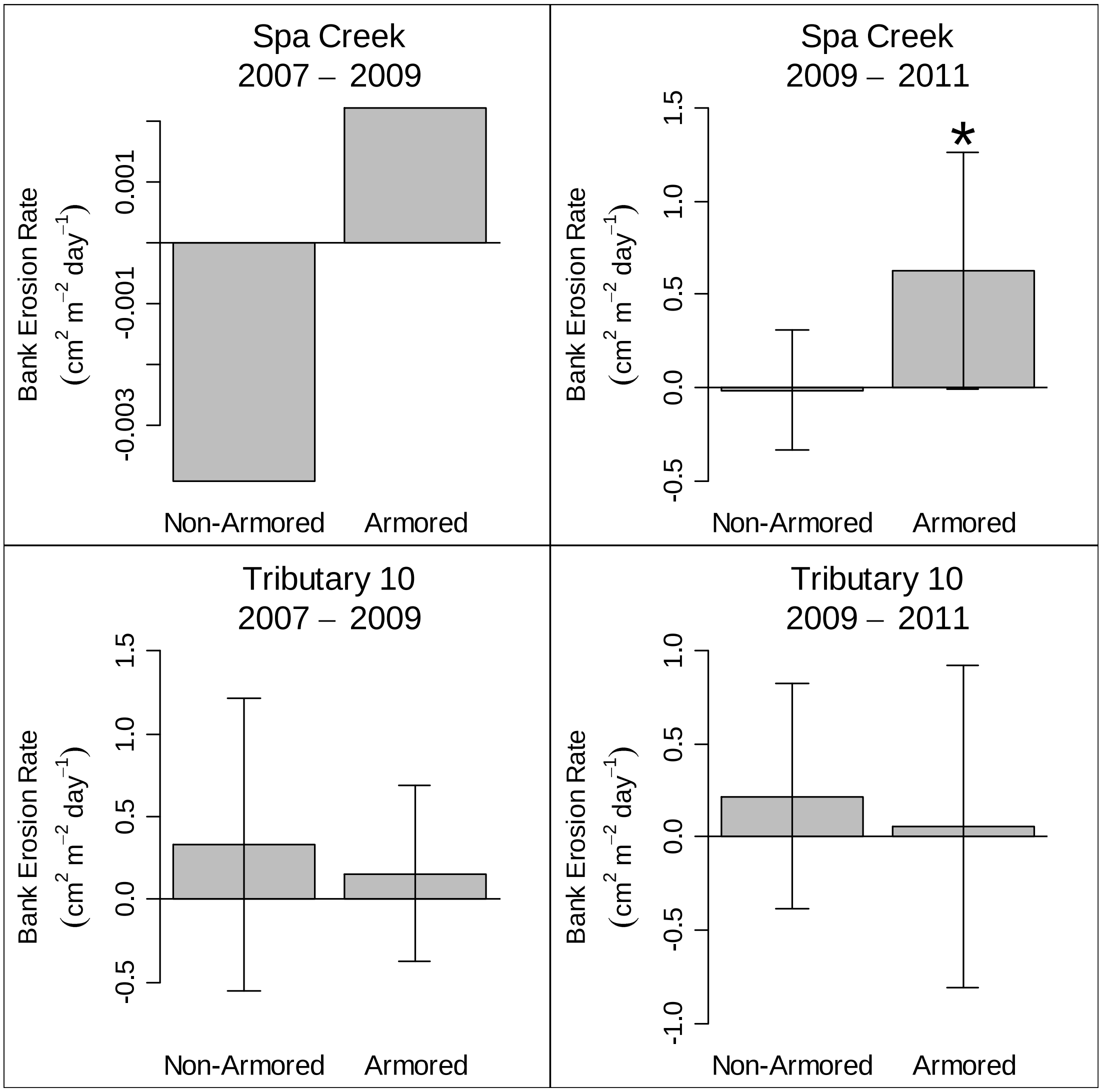


Figure 2.6. Bank erosion rates at armored and non-armored banks in two restored streams – Spa Creek and Tributary 10. Positive values indicate a reduction in bank area (erosion), negative values an increase in bank area (aggradation). Error bars are standard deviations. Error bars are not shown on the Spa Creek 2007-2009 plot, because they are an order of magnitude larger than the average values. * indicates that armored cross-sections were significantly different from non-armored cross-sections at $\alpha = 0.05$.

DISCUSSION

Disturbance of streambed sediments is well known to influence benthic organisms (Fisher et al. 1982, Power and Stewart 1987, Grimm and Fisher 1989) but the relationship between streambed disturbance and channel modification at scales relevant to the biota has received little study (Stanley et al. 2010). The results presented here indicate that stream restoration can modify the impact of high flow events on bed sediment disturbance patterns and do so in complex ways that are likely to impact ecological structure and function. While inferences are constrained by the lack of pre-restoration data and lack of multiple non-restored urban streams for comparison, the results suggest that the addition of flow obstructions and modification of the channel slope profile via restoration may lead to selective removal of fine sediments during baseflow and coarsening of bed sediments that increase resistance to entrainment during high flow events. The increased resistance to entrainment suggests that bed sediment patches near flow obstructions are more likely to remain stable during high flow events (Fig. 2.2) and could serve as refugia for benthic invertebrates. This increase in the stability of bed patches as a direct result of the addition of flow obstructions suggests that the restoration projects have modified the bed sediment disturbance regime on a scale relevant to biota.

In contrast to the effects of restoration on bed sediment disturbance, I did not find evidence that bank armoring substantially reduced bank erosion rates. Bank grading and channel reconstruction may have had a beneficial impact on channel movement rates, but this result must be interpreted with caution due to the lack of pre-restoration data (see Bank Erosion section).

Bed Sediment Disturbance

The predictions regarding how channel stability restoration was likely to influence bed sediment movement patterns were supported by the results of the scour chain experiment. There was a significant positive relationship between flow obstruction metrics and median grain size within habitat units (Prediction 1), indicating that bed sediments were on average coarser in areas with more flow obstructions.

Bed sediments were also more likely to remain stable during flow events in areas with flow obstructions (Fig. 2.2), potentially for two reasons. First, because bed sediments in habitat units with flow obstructions were coarser on average, they were less likely to be mobilized and thus less likely to have high scour depths (Prediction 2). Second, bed sediments near more flow obstructions were also less likely to have high deposition, and this effect was much stronger than the reduction in scour magnitude, as seen in the significant negative relationship between scour/fill magnitude and flow obstruction metrics (Fig. 2.2).

The different flow patterns in the July-August and September study periods also highlight the importance of reduction in deposition depth as an influential effect of flow obstructions. In the July-August study period, bed sediments were reworked by a period of baseflow following the most recent high flow event and prior to scour chain recovery. In September, scour chains were recovered immediately after return to baseflow. The stronger negative relationship between scour-fill magnitude and flow obstruction metrics in the July-August period may indicate that during baseflow and prior to scour chain recovery, sediments deposited during the high flow events were removed preferentially in areas with flow obstructions, thereby reducing deposition. The greater influence of flow obstructions during the July-August sampling period is further supported by the fact that the relationship between unit-scale average scour/fill and flow obstructions was significant in the July-August sampling period but not in September.

I did not find a significant relationship between variability in scour/fill magnitude and flow obstructions at the unit scale (Prediction 3), suggesting that the predictability of scour/fill magnitude at individual scour chains was low in units with and without flow obstructions. Despite this high variability between scour chains, flow obstructions had a detectable effect on bed sediment disturbance patterns by decreasing the probability of a large scour or fill magnitude at individual scour chains.

I predicted that this effect of flow obstructions at the patch scale would lead to altered patterns of bed mobility at the reach scale (Prediction 4). The prediction was supported, but only for streams with a wide grain size distribution (i.e., streams with both sand and large gravels). Streams with a wide grain size distribution had the lowest reach-scale proportion bed sediment mobile but the highest spatial variability in proportion mobile (Fig. 2.4).

A possible explanation for these patterns is that streams with a wide grain size distribution had units with coarse bed sediments and units with fine sediments. Units with coarse bed sediments would tend to have a low proportion of bed sediments mobilized, reducing reach-scale proportion mobile; but units with fine bed sediments would have a high proportion of bed sediments mobilized, which would increase variability in proportion mobile between units. Flow obstructions – along with slope differences – provide a mechanism to create a pattern of units with coarse grains and units with fine grains because, as discussed above, flow obstructions tended to coarsen the grains in units where they were placed. Correlations between proportion of sand in the bed sediments in a unit and unit-scale proportion mobile during both flow periods at streams with the widest grain size distributions were generally negative, offering some support for this hypothesis, but no correlations were significant, indicating that other factors such as channel slope may obscure this pattern. Thus, the validity of this prediction needs to be tested further by manipulating the grain size distribution and number of flow obstructions in individual units while holding other variables such as slope and channel geometry constant.

I did not detect a difference between restored and non-restored streams in either reach-scale proportion mobile or variability in proportion mobile. The lack of an effect may have been due to the strong influence of grain size, which was highly variable between streams. In addition, there was low statistical power to detect a difference, given that there was only one non-restored urban stream, which was not severely impacted by urbanization. Indeed, the influence of flow obstructions on scour/fill magnitude as described suggests that restoration practices did in fact alter reach-scale mobility patterns by reducing the likelihood of sediment entrainment and deposition in localized areas of the streambed. Comparison of restored streams with additional urban streams will be necessary to determine with more certainty whether restoration altered reach-scale patterns of sediment disturbance.

The predictions regarding how channel stability restoration is likely to impact bed sediment movement patterns were derived by considering how flow obstructions were likely to alter flow patterns and in turn, bed sediment size distribution and bed sediment movement patterns. It was hypothesized that flow obstructions would cause local flow accelerations during baseflow, leading to selective removal of small grains and coarsening of the bed material around obstructions (Prediction 1). The coarser material around obstructions was predicted to be more stable during subsequent high flow (Prediction 2), though also more variable (Prediction 3). The increased stability of bed patches in areas with obstructions was predicted to decrease the total amount of bed sediment disturbance on the reach scale and increase the variability between units in amount of bed sediment disturbance (Prediction 4). Experimental results were consistent with these predictions, however, there is no direct experimental evidence that the mechanism as proposed was in fact driving the observed patterns.

Research in flumes has shown that obstructions cause local flow accelerations and generate turbulent eddies that can entrain and transport small grains (Melville and Sutherland 1988, Dey and Raikar 2007, Kothyari et al. 2007), but turbulence intensity was not directly measured in this study, making it unclear whether flow obstructions did in fact increase turbulence intensity during baseflow. In addition, there is no direct evidence that transport of smaller grains was occurring during baseflow in areas around obstructions. It is possible that the observed coarsening of grains around flow obstructions was driven by increased channel slope and obstructions were simply placed in areas with higher channel slope and did not directly increase turbulent forces and transport of small sediment grains. Future work that couples measurement of bed disturbance patterns with direct measures of turbulent forces and sediment transport rates in restored streams will be required to determine the validity of the predicted mechanism for how stability restoration impacts bed sediment movement patterns as described here.

Channel Movement

The practice of bank armoring as part of stability restoration appears to have done little to alter channel movement rates since measures of channel erosion were similar or higher in armored compared to non-armored sections of restored streams. Although a few case studies have found that bank armoring reduces erosion (Chen et al. 2005, NRCS 2007), previous work has also shown that bank armoring and other restoration structures can negatively impact bank stability. Segura and Booth (2010) showed that bank armoring disconnects floodplains from stream channels, which prevents dissipation of flood energy and can actually enhance channel incision. Similarly, restoration structures may sometimes enhance erosion (Thompson 2002, Miller and Kochel 2010).

Although armoring had little effect on bank erosion, the bank grading and channel manipulation performed in the restored streams may have led to reduced bank erosion relative to pre-restoration levels. This is possible because erosion rates, channel widening rates, and thalweg scour were similar and in a few cases lower in restored streams compared to a reference urban stream. Whereas restoration often targets severely incised streams, the urban reference stream sampled in this study has a well-vegetated buffer and was not noticeably incised or degraded.

Previous work has shown that creation of an appropriate morphologic form, not bank armoring per se, creates channel stability (Ernst et al. 2010). Furthermore, failure of restoration to achieve channel stability has been attributed to disequilibrium between the designed channel form and river processes, so that morphologic adjustment occurs after restoration (Frissell and Nawa 1992, Kondolf et al. 2001, Thompson 2003, Price and Birge 2005, Smith and Prestegaard 2005). The urban stream sampled in this study did show a trend of channel widening and scour during the 2009-2011 sampling period, and all restored streams had lower average values of these metrics, with two streams having significantly lower values. Thus, it is possible that restoration practices reduced the amount of widening and scour during this period.

Channel movement rates in the studied restored streams were within the range of the single forested site surveyed, suggesting that channel movement rates in the restored streams were not abnormally low. Since the urban stream had relatively high values of some channel movement rates, there is some evidence that channel movement rates did move toward a more natural state as a result of the restorations. However, without pre-restoration data, the efficacy of stability restoration cannot be determined with certainty.

One restored stream (Cowhide Branch) showed significant aggradation relative to the other study streams during the 2007-2009 sampling period and provides a case study highlighting the importance of watershed processes in controlling channel morphologic change. Cowhide Branch likely aggraded during this period due to the release of a large amount of sediment from the collapse of an upstream stormwater pond in 1993 (Morris 1993, Siegel 1994). There was clearly a significant sediment load that was passing through the reach because nearly all cross sections at this stream showed thalweg aggradation between 2007 and 2009 (Fig. 2.5). In particular, the deepest scour pool that was surveyed at this stream aggraded more than 50 cm during this period (Fig. 2.7). Even if the stormwater pond collapse was not the source of sediment that filled in the channel, it is clear that a large amount of upstream-derived material was moving through the reach, given the consistent aggradation response throughout the channel. Thus, this case study shows that watershed-scale processes can exert controls on channel changes that overwhelm any effect of stability restoration, and supports the argument made elsewhere that restoration should target watershed processes before attempting in-

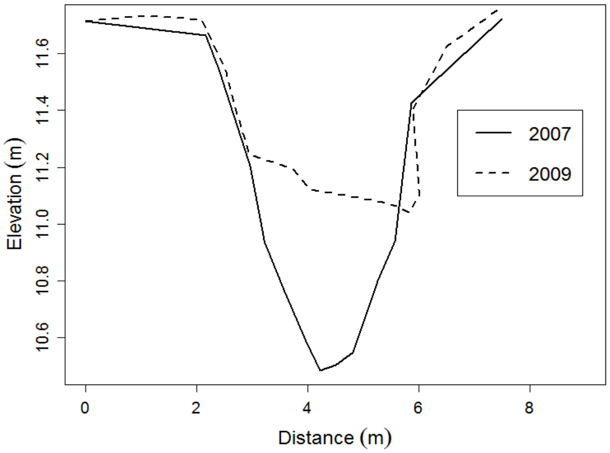


***Figure 2.7**. Cross section of a scour pool at Cowhide Branch that filled in with sediment between 2007 and 2009.*

channel rehabilitation (Roni et al. 2002, Wohl et al. 2005, Roni et al. 2008, Miller et al. 2010, Palmer et al. 2010).

Conclusions and Implications

I found evidence that stability restoration has altered patterns of bed sediment mobility at both the local and reach scales, primarily through the addition of flow obstructions, which act to coarsen the bed sediments and reduce the probability of sediment mobility in surrounding patches of bed sediment. The reduction in sediment mobility may be considered beneficial if the goal of restoration is to reduce channel incision or aggradation. However, the forested stream surveyed in this study had very few sediment patches that remained stable during the measured flow events, and therefore the manipulation of these stream channels has led to an alteration of the disturbance regime, which may impact benthic communities.

The increased number of stable sediment patches could increase heterogeneity on the patch scale by providing refugia during high flow events (Lancaster and Hildrew 1993), and recent work has shown shifts in invertebrate abundance patterns and declines in invertebrate taxonomic richness with increased sediment movement (Albertson et al. 2011, Schwendel et al. 2011). Alternatively, alteration of the natural state may negatively impact biota adapted to high levels of bed disturbance. In either case, recent work has shown that invertebrate communities are weakly related to proportion mobile on the reach scale (Schwendel et al. 2011), and it remains to be tested whether the effect of restoration seen here is substantial enough to alter benthic community dynamics.

With regard to channel movement rates, there was some evidence that restoration reduced channel erosion and scour relative to an urban stream during the sampling period. Reduction of channel movement rates would be viewed as beneficial in terms of the restoration goals, but research has shown that movement of stream channels maintains beneficial habitat for both aquatic and riparian biota (Florsheim et al. 2008). However, channel movement rates in restored streams were not significantly lower compared to a forested reference stream, suggesting that restoration has not overly stabilized stream channels.

It appears that bank grading and channel reconstruction were primarily responsible for changes in channel movement rates as opposed to bank armoring, which did not alter bank erosion rates relative to non-armored banks in restored channels. These results suggest that stability restoration created a more stable channel relative to a reference urban stream but with similar channel movement rates to a forested reference channel. However, whether the restoration projects actually reduced bank erosion and channel incision is uncertain without pre-restoration assessments.

Chapter 3: Temporal effects of channel stability restoration and riparian buffer restoration on riparian soils and vegetation

ABSTRACT

Stream restoration projects are often employed in efforts to stabilize channel banks and prevent bank erosion. Banks are commonly stabilized through grading and armoring with heavy machinery, a hard engineering approach known as stability restoration, or alternatively through riparian buffer restoration. How these different restoration techniques impact riparian soils and vegetation is not well known. I compared how stability restoration and riparian buffer restoration projects impacted riparian soils and vegetation relative to control and forested sites and how impacts changed over time. Each project type was represented by at least 10 paired restored and control reaches and forested sites were represented by five paired upstream-downstream reaches. All streams were located in the Piedmont and Coastal Plain physiographic provinces in Maryland, USA. Stability restoration project ages ranged from 2-16 years and riparian buffer restoration projects ranged from 2-13 years. I measured moisture content, bulk density, soil organic matter (SOM), root biomass, C:N ratio, percent fine sediments, and denitrification enzyme activity (DEA) on bulk soil samples collected at three depths (0-10 cm, 10-20 cm, and 20-30 cm) from each reach. I measured percent cover and percent invasive species at each reach using a quadrat sampling method. Stability restoration sites showed generally negative impacts on riparian soils, particularly at the 10-20 and 20-30 cm soil layer where bulk density was higher and SOM and root biomass were lower in restored relative to

control sites. Percent invasive species was also higher in restored relative to control reaches at stability restoration sites. In contrast, riparian buffer restoration sites had generally positive to neutral effects on soils and vegetation, including an apparent increase in DEA in Piedmont sites. Restored and control reaches of both types of restoration had similar soil properties to forested reference reaches, indicating that the magnitude of restoration effects was relatively small. However, the fact that stability restoration projects showed mostly negative impacts and riparian buffer creation projects showed mostly positive or neutral effects on riparian soils and vegetation suggests that riparian buffer creation is a more ecologically favorable method for bank stabilization.

INTRODUCTION

Increased stormflow runoff in urban watersheds commonly leads to increased rates of incision and bank erosion (Wolman 1967, Hammer 1972, Arnold et al. 1982, Booth 1990, MacRae and Rowney 1992, Booth and Jackson 1997, Hardison et al. 2009) and often prompts stream restoration with the intention of stabilizing banks and creating a stable channel form (Bernhardt et al. 2005). Many of these projects use a hard engineering approach that involves bank grading, channel reconfiguration, riparian vegetation planting, and installation of bed and bank protection features such as rip rap and cross vanes (Brookes and Shields 1996, Brown 2000**,** Niezgoda and Johnson 2006).   This type of stability restoration is common throughout the United States and particularly in the Chesapeake Bay watershed (Hassett et al. 2005).

Restoration for channel stability often involves substantial disturbance of the channel and adjacent riparian zones, as trees are often removed and heavy machinery is used to grade channel banks, position boulders, install grade control structures, or reconfigure channels (Rosgen 1996, Miller and Kochel 2010). Previous studies have found that restoration activities similar to those involved in channel stability restoration tend to compact soils and increase bulk density (Merz and Finn 1951, Campbell et al. 2002, Bruland and Richardson 2005). Root biomass has also been shown to be lower in soils where stability restoration occurred relative to undisturbed soils (Gift et al. 2010). Restoration actions similar to those involved in stability restoration typically cause homogenization and loss of soil organic matter (SOM) in riparian and wetland soils (Clewell and Lea 1989, Shaffer and Ernst 1999, Bruland and Richardson 2005, Bruland et al. 2006, Unghire et al. 2010). Loss of SOM may fundamentally alter soil ecosystem processes, in particular, denitrification, which can be dependent on the presence of SOM if other factors (e.g., soil nitrate) are not limiting (Duncan and Groffman 1994, Pavel et al. 1996, Ullah and Faulkner 2006, Hernandez and Mitsch 2007, Balestrini et al., Gift et al. 2010, Sutton-Grier et al. 2010). Reduction in root biomass may further impact denitrification when SOM is a limiting factor, because roots help deliver SOM to deeper soil layers (Gift et al. 2010).

The disturbance to riparian soils and existing vegetation may also promote establishment of non-native invasive species (Burke and Grime 1996, Fennessy and Roehrs 1997, Davis et al. 2000, Spieles 2005). Changes in plant composition may have cascading effects on both soil and aquatic food webs (Belnap and Phillips 2001, Korthals et al. 2001, Wardle et al. 2003, De Deyn et al. 2004, Greenwood et al. 2004, Swan and Palmer 2004, Belnap et al. 2005, Lecerf et al. 2005, Lecerf et al. 2007, Ball et al. 2009).

In contrast to these hard-engineering channel stability restoration projects (hereafter stability restoration), efforts to stabilize stream banks also include restoration projects that focus on enhancing the riparian buffer along streams (hereafter riparian buffer restoration) by planting and tending native vegetation (Parkyn et al. 2003, Beeson and Doyle 2007). Riparian buffer restoration is a less expensive restoration technique than stability restoration (Hassett et al. 2005), and may provide additional ecological benefits to stream ecosystems such as the provision of shade, improved water quality, and the restoration of wood inputs to streams (Naiman et al. 2005, Orzetti et al. 2010). However, the impact of riparian buffer restoration on soil structure and function has received less attention.

Despite previous work that has found generally negative impacts on riparian soils from stability restoration and generally positive impacts on stream ecosystems from riparian buffer restoration, there have been no direct comparisons of the impacts of these restoration techniques on riparian soils and vegetation, and therefore, in this chapter, I provide such a comparison. Specifically, I tested the hypothesis that stability restoration would promote invasive species, increase soil bulk density, reduce SOM, reduce root biomass, and reduce soil dentirification potential relative to riparian buffer restoration. I also tested the hypothesis that riparian soils and vegetation would demonstrate some recovery over time from the one-time impact of stability restoration, while soils at riparian buffer restoration sites would show little change over time.

Since my restoration sites were all located in urbanized watersheds that had once been forested, riparian soils along forested reference streams were included for comparison. In addition, I sampled a riparian soil control site adjacent to each restoration site. Therefore, I had four categories of sample sites (stability restoration, riparian buffer restoration, non-restored urban, and forested), which allowed me to determine whether urban soils were degraded relative to forested soils and to determine whether soil properties in restored streams compared more similarly to urban or forested soils.

METHODS

Study Sites

I sampled soils and vegetation along 28 streams, of which 13 were classified as hard stability restoration, 10 as riparian buffer, and five as forested control sites (Table 3.1; Fig. 3.1). At each stability restoration site, banks had been graded and backfilled to achieve designed cross-section profiles and boulders or logs had been added along several banks to help stabilize cross-section morphology. In addition, the channel at each site was manipulated to achieve a designed slope profile, and channels were further stabilized by different combinations of instream structures, including rock vanes, vortex rock weirs, and log vanes. Riparian vegetation was reseeded or replanted on disturbed soils at all stability restoration sites. Riparian buffer sites had vegetation planting as the only stream restoration activity. Forested reference sites were not subject to any restoration and had a well-forested buffer. All study sites were located in Maryland, with eight in the Piedmont physiographic province (two stability restoration, five riparian buffer restoration, and one forested control) and 20 in the Coastal Plain (11 stability restoration, five riparian buffer restoration, and four forested control). Further details of the soils at each site including upland vs. floodplain distinction, soil texture, and drainage class are provided in the soil maps in Appendix II.

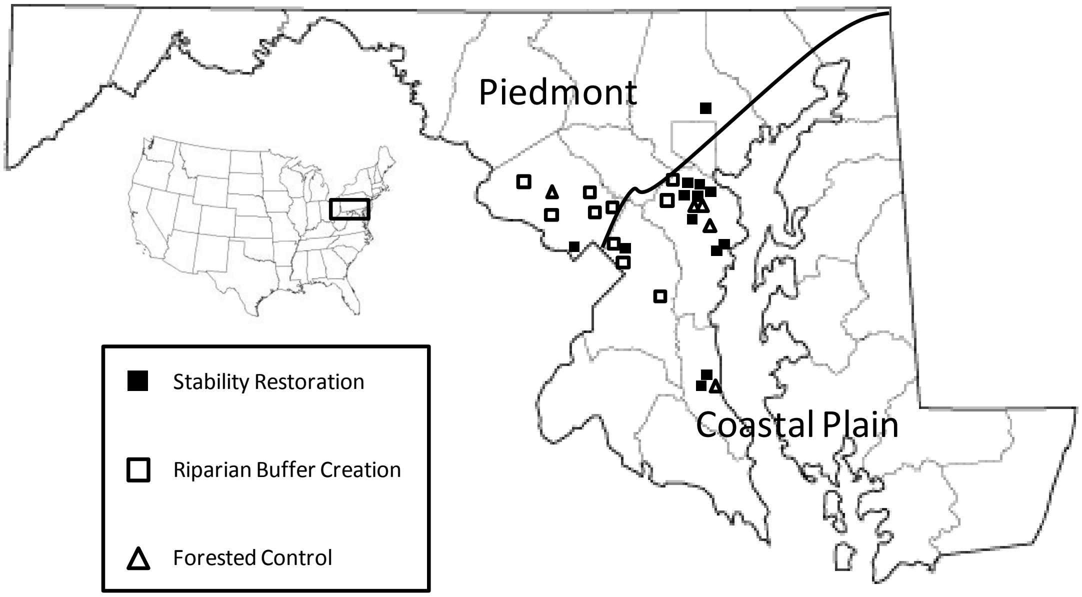


Figure 3.1. Distribution of study sites within Maryland. The dark line shows the approximate boundary between the Piedmont and Coastal Plain physiographic provinces.

Table 3.1. Names, location, type of restoration, physiographic province, and year restoration was completed for all study sites.

Site (GPS Coordinates)	Restoration Type	Physiographic Province	Year Completed
Severn Run Tributary 1 (N 39°04’34”, W 76°37’07”)	Forested	Coastal Plain	-
Severn Run Tributary 2 (N 39°06’21”, W 76°39’03”)	Forested	Coastal Plain	-
Severn Run Mainstem (N 39°06’30”, W 76°39’05”)	Forested	Coastal Plain	-
Parkers Creek Tributary (N 38°31’05”, W 76°31’33”)	Forested	Coastal Plain	-
Gunners Branch (N 39°09’37”, W 77°15’26”)	Forested	Piedmont	-
Elvaton Towne Center (N 39°07’17”, W 76°37’19”)	Hard Stability	Coastal Plain	2004
Harundale Town Center (N 39°09’12”, W 76°36’23”)	Hard Stability	Coastal Plain	2005
Hunting Creek (N 38°33’38”, W 76°35’41”)	Hard Stability	Coastal Plain	2004
Muddy Bridge Branch (N 39°10’32”, W 76°38’41”)	Hard Stability	Coastal Plain	1997
South Fork Jabez Branch (N 39°03’42”, W 76°39’14”)	Hard Stability	Coastal Plain	1997
Spa Creek (N 38°58’23”, W 76°31’03”)	Hard Stability	Coastal Plain	2001
Sullivans Branch (N 38°32’02”, W 76°35’14”)	Hard Stability	Coastal Plain	2008
Tributary 9 of Sawmill Creek (N 39°10’54”, W 76°38’09”)	Hard Stability	Coastal Plain	1994
Tributary 10 of Sawmill Creek (N 39°10’58”, W 76°37’24”)	Hard Stability	Coastal Plain	1998

Table 3.1 cont.

Site (GPS Coordinates)	Restoration Type	Physiographic Province	Year Completed
University Park Stream (N 38°58’14”, W 76°56’53”)	Hard Stability	Coastal Plain	2002
Weems Creek at Bristol Drive (N 38°59’11”, W 76°31’10”)	Hard Stability	Coastal Plain	1997
Little Falls Branch (N 38°57’20”, W 77°06’34”)	Hard Stability	Piedmont	2007
Minebank Run (N 39°24’36”, W 76°33’24”)	Hard Stability	Piedmont	1998
Anacostia River (N 38°56’32”, W 76°56’41”)	Riparian Buffer	Coastal Plain	2009
Bollack Farm Stream (N 39°12’13”, W 76°42’35”)	Riparian Buffer	Coastal Plain	2002
Northwest Branch Lower (N 38°59’11”, W 76°57’51”)	Riparian Buffer	Coastal Plain	1997
Severn Run Mainstem (N 39°07’43”, W 76°42’57”)	Riparian Buffer	Coastal Plain	2001
Western Branch Patuxent River (N 38°48’57”, W 76°44’49”)	Riparian Buffer	Coastal Plain	2002
Hawlings R. Trib. at St. Alb. Dr. (N 39°10’00”, W 77°03’14”)	Riparian Buffer	Piedmont	1997
Leaman Farm Stream (N 39°09’47”, W 77°17’26”)	Riparian Buffer	Piedmont	1997
Northeast Branch headwater (N 39°06’53”, W 76°57’45”)	Riparian Buffer	Piedmont	1998
Northwest Branch Upper (N 39°06’01”, W 77°02’12”)	Riparian Buffer	Piedmont	2003
Watts Branch Trib. at Valley Dr. (N 39°04’39”, W 77°12’39”)	Riparian Buffer	Piedmont	1998

Field Sampling

In summer 2010, at each restored site I sampled both the restored reach and an upstream or downstream non-restored control reach. Downstream reaches were used when the restoration project was located at the headwaters of the stream. Reach length varied from 50 – 100 m depending on the width of the stream. At forested sites I sampled two reaches that were separated by at least one reach length.

I collected one bulk soil sample from each reach at each of three soil depths by compositing 15 individual soil cores taken along a transect parallel to the stream. Three successive 10 cm sub-cores (0-10 cm, 10-20 cm, and 20-30 cm) were extracted one at a time from each sampling location. Each sub-core had a volume of 39 cm^3. Cores were spaced 1-5 m apart along the transect depending on the width of the stream and were not taken from seeps or other anomalous bank features. Each core was located 1-3 m from the edge of permanent vegetation up the stream bank to ensure that bank soils rather than alluvial sediments were sampled. Cores were stored at 4°C in plastic bags until laboratory processing to reduce enzymatic activity and prevent evaporation.

Vegetation along the stream bank was surveyed to determine percent cover and percent cover of common invasive species. Using a 10 x 10 cm grid, one quadrat was surveyed at every other soil core location, starting with the first soil core, for a total of eight quadrats. Within each quadrat, the number of grid points that intercepted live vegetation and the number of vegetation grid points identified as a common invasive plant species were tallied. Percent cover and percent of vegetation composed of invasive species (percent invasive) were calculated for each quadrat and numbers for each quadrat were averaged for each reach.

Laboratory Processing

Upon return to the laboratory, bulk soil samples were thoroughly homogenized, weighed, and analyzed for moisture content, bulk density, soil organic matter (SOM), total carbon, total nitrogen, root biomass, soil size composition, and denitrification potential. Three subsamples measuring about 10% of total wet soil mass were weighed, dried at 70°C for at least 115 hours, and reweighed to determine moisture content. Moisture content of the three subsamples was averaged and used to determine total dry soil mass of each bulk sample. Bulk density was calculated by dividing the total dry mass by the total sample volume.

After bulk density was measured, the dried triplicate subsamples were combined and ground with a mortar and pestle. Three ~3 g subsamples from this dried, ground material were placed in crucibles and combusted in a muffle furnace at 450°C for 16 hours. SOM was then calculated as the percent mass loss on ignition (%LOI) relative to subsample dry mass (Nelson and Sommers 1996). Three ~15 mg subsamples from the dried, ground material were analyzed for total soil carbon and nitrogen using a CHNSO elemental analyzer (ECS 4010 CHNSO Analyzer, Costech Analytical Technologies, Valencia, CA). Total soil carbon and nitrogen masses were used to calculate the C:N ratio of each sample. The three subsamples for SOM and C:N ratio were then averaged for each site. The remainder of the dried, ground material that was not used for SOM and C:N determination was passed through a series of sieves to separate gravel, sand, and fine (<63 μm) soil size classes and roots were picked from the sieved material. The roots and each of the sediment size fractions were then weighed, combusted at 450°C for 16 hours, and reweighed. Mass of soil remaining after combustion was used as the mass of each soil size fraction and each size fraction was expressed as a percent of total dry soil mass. Note that percent fines as used here is defined operationally as soil grains small enough to pass through a 63 μm sieve during a dry sieving process. This likely underestimates the actual percent fines, as fine particle aggregates would not have passed through the sieve. Root biomass was calculated as %LOI relative to dry soil mass. Large pieces of root weighing more than 1% of dry soil mass were excluded from analysis, as these were rare in samples and were not representative of the average root biomass.

Denitrification potential was measured using the assay for denitrification enzyme activity (DEA) developed by Smith and Tiedje (1979) and described by Groffman et al. (1999). The assay was run by placing 10 g of moist soil in a 125 ml Erlenmeyer flask with a tapered ground glass neck fitted with rubber septa to allow extraction of N_2O gas samples via syringe. A solution of glucose and potassium nitrate (KNO_3^-) was added to the soil to ensure non-limiting substrate conditions, and chloramphenicol was added to prevent microbial reproduction. The flasks were flushed three times with N_2 gas to create anaerobic conditions and were then injected with 5 ml of acetylene gas (C_2H_2) to prevent reduction of N_2O to N_2. The flasks were placed on a shaker table for 90 minutes at 100 rpm, at which point an N_2O gas sample was taken from each flask and injected into a 9 ml pre-evacuated septum-sealed vial. Concentrations of N_2O were measured on a GC-14B equipped with an electron capture detector with a Porapak Q column (Shimadzu Corporation, Kyoto, Japan). Soil N_2O flux was calculated as the time-linear rate of N_2O production after correcting for dissolved N_2O in the amendment solution using the Bunsen equation (Groffman et al. 1999). DEA was expressed as the rate of N_2O production per kilogram of moist soil.

Statistical Analysis

To test the hypothesis that stability restoration negatively impacts riparian soils, I performed a repeated-measures analysis using reach type (control vs. restored) within each stream as the repeated measure. To test the effect of stability restoration on soil characteristics I performed a MANOVA with bulk density, SOM, and root biomass included as response variables and with soil depth class (0-10 cm, 10-20 cm, and 20-30 cm) and physiographic province (Piedmont and Coastal Plain) included as blocking variables. This analysis tested whether the difference in soil metrics between the control and restored reach at each stream was significantly different from 0 and whether soil variables were responding differently across depth classes and physiographic province. To determine whether soil variables recovered over time from any negative impacts associated with stability restoration, I included restoration age class (<10 and >10 years old) in the MANOVA and looked for a significant interaction between age class and reach type, which should be present if the difference between restored and control reaches changed over time. Ten years was chosen as the dividing age because previous research has shown that 10 years is sufficient for restored riparian buffers to provide water quality benefits at a similar level as native riparian forest (Orzetti et al. 2010).

I tested whether DEA (representative of a soil function – denitrification) was affected by stability restoration by performing a repeated-measures ANOVA, with depth class, physiographic province, and age class included as blocking variables, as was done with the MANOVA of bulk density, SOM, and root biomass. Repeated-measures ANOVA was also used to test whether percent cover and percent invasive were affected by stability restoration across depth classes, physiographic provinces, and age classes. Separate ANOVAs were performed for these vegetation metrics because they were not significantly correlated and I was interested in examining their responses separately.

To test the effects of riparian buffer restoration on riparian soils and vegetation, I performed the same analyses with the riparian buffer restoration sites as was done for the stability restoration sites (MANOVA of bulk density, SOM, and root biomass and separate ANOVAs for DEA and the vegetation metrics). I also performed these same analyses on the upstream vs. downstream reaches in forested streams, with the exception that age class was not included as a variable. The purpose of performing the analyses with forested streams was to determine whether soils in upstream and downstream reaches differed significantly in the absence of restoration impacts.

Percent cover and all soil variables except bulk density were $\log_{10}$-transformed and percent invasive was logit transformed ($y = \ln(x/1-x)$) prior to analysis to meet assumptions of normality and homogeneity of variance. All MANOVAs were run using the *Anova* function in the car package (Fox and Weisberg

2011) and ANOVAs were run using the *aov* function in the stats package of R, version 2.13.0 (R Development Core Team 2011).

To determine whether soil properties in restored streams were more similar to urban or forested soils, I used Principal Component Analysis (PCA) to identify the underlying gradients that separated sites based on the suite of soil metrics measured. The C:N ratio and percent fine soil (less than sand size) were included in the PCA. I performed a separate PCA for each soil depth and visualized site groupings on two-dimensional biplots. The analysis was performed using the *princomp* function in the *stats* package of R, version 2.13.0 (R Development Core Team 2011).

RESULTS

Forested Sites

Root biomass, SOM, and DEA were on average similar in forested sites compared to a similar study of restoration projects that sampled forested control sites in Maryland (Gift et al. 2010; Table 3.2). Bulk density was relatively low on average in forested sites, usually <1.0 g cm^{-3} and never exceeded values typical of compacted soils that restrict root growth (1.5-1.8 g cm^{-3}; Daddow and Warrington 1983). However, soil properties were variable among forested sites, ranging from relatively low to very high (see Table 3.2). Bulk density increased with soil depth and SOM, root biomass, and DEA all decreased with soil depth.

Most interaction terms in all analyses of differences between control and restored reaches (upstream and downstream reaches at forested streams) were not significant and I report interaction effects where they are significant and otherwise report only main effects throughout. Neither bulk density, SOM, and root biomass collectively nor DEA differed significantly between upstream and downstream reaches at forested sites ($F = 3.019_{(1, 11)}$, $p = 0.11$ and $F = 0.6772_{(1, 9)}$, $p = 0.43$, respectively; Fig. 3.2). Percent cover, and percent invasive were also similar between upstream and downstream reaches ($F = 0.0994_{(1, 3)}$, $p = 0.77$ and $F = 0.0004_{(1, 3)}$, $p = 0.98$, respectively; Fig. 3.3).

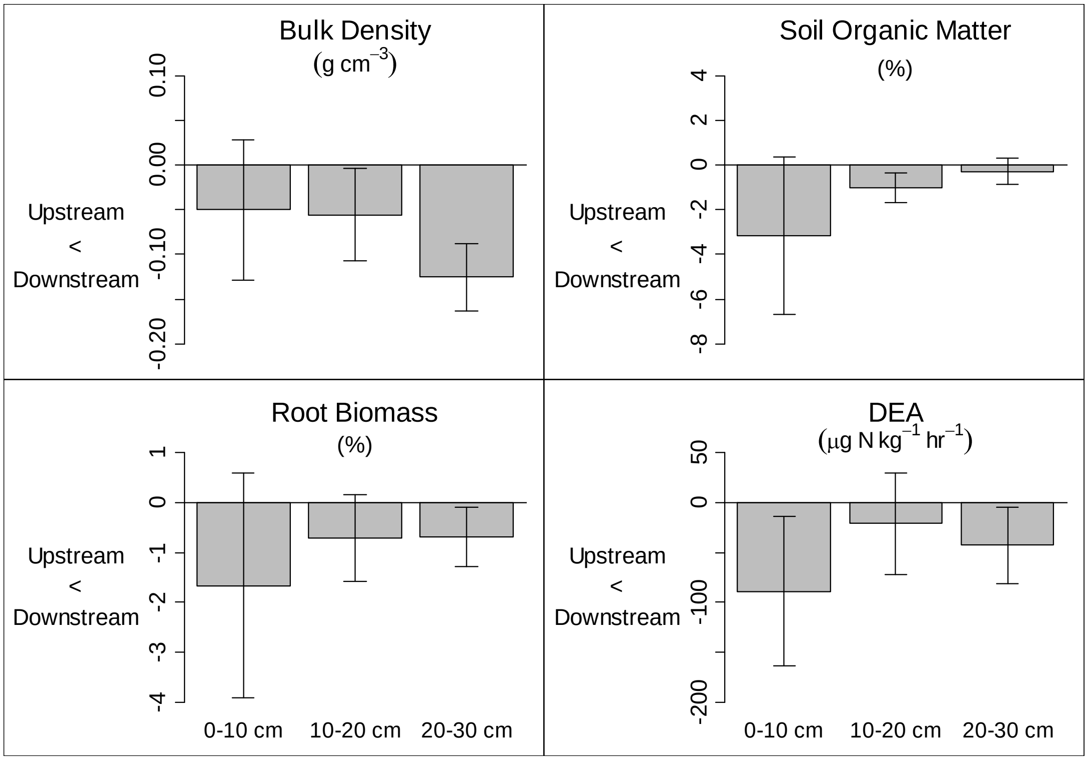


Figure 3.2. Differences in soil properties between upstream and downstream reaches at forested sites ($n = 5$). Each bar is the average of differences at each site and each is a different soil layer (0-10 cm, 10-20 cm, and 20-30 cm). Negative values indicate that downstream values were higher than upstream values and positive values the reverse. Error bars are standard errors.

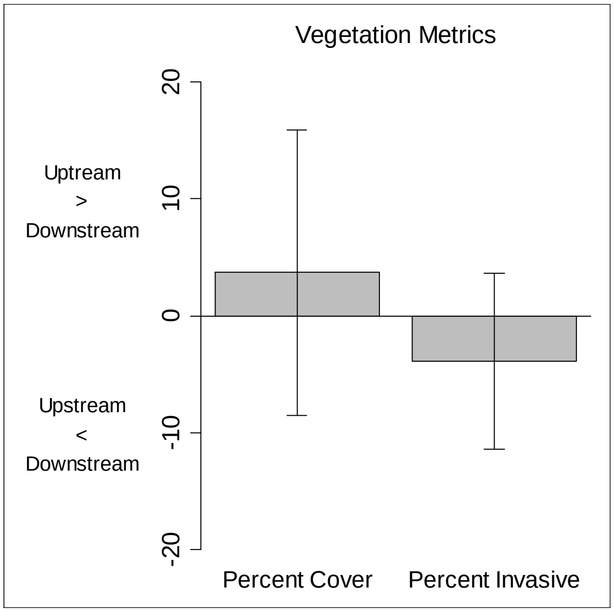


Figure 3.3. Differences in percent cover and percent invasive species between upstream and downstream reaches at forested sites (*n* = 5). Negative values indicate that downstream reaches had higher percent cover and percent invasive species than upstream reaches.

Stability Restoration Sites

Bulk density and root biomass were on average similar and SOM and DEA were slightly lower in both urban control and restored reaches compared to similar studies of restoration projects in both Maryland and North Carolina (Gift et al. 2010, Unghire et al. 2010; Tables 3.2 and 3.3). Bulk density approached values known to restrict root growth (1.5-1.8 g cm^{-3}; Daddow and Warrington 1983) at multiple restored reaches, particularly in the 20-30 cm layer, but only rarely at urban control reaches.

There was a significant interaction effect between depth class and reach type in the MANOVA of bulk density, SOM, and root biomass ($F = 10.57_{(1, 31)}$, $p = 0.003$), and there was a marginally significant interaction effect between physiographic province and reach type (F = 3.78$_{(1, 31)}$, p = 0.06). When the MANOVA was performed separately for sites in the Piedmont and Coastal Plain physiographic provinces, the depth class by reach type interaction effect remained significant only for the Coastal Plain sites (F = 0.22$_{(1, 2)}$, p = 0.68 for Piedmont sites and F = 11.11$_{(1, 29)}$, p = 0.002, respectively), and therefore the effect of reach type was analyzed separately within depth classes only for the Coastal Plain sites. Bulk density, SOM, and root biomass did not differ significantly between restored and control reaches in the Piedmont region (F = 0.09$_{(1, 2)}$, p = 0.79). In the Coastal Plain, bulk density, SOM, and root biomass did not differ significantly between restored and control reaches in the 0-10 cm soil layer but did differ significantly in the 10-20 cm soil layer and the 20-30 cm soil layer (F = 0.057$_{(1, 9)}$, p = 0.82 for the 0-10 cm layer; F = 8.281$_{(1, 9)}$, p = 0.02 for the 10-20 cm layer; F = 22.474$_{(1, 9)}$, p = 0.001 for the 20-30 cm layer). Bulk density was on average 19% (0.18 g cm^{-3}) higher, SOM was 16% (0.008 g g^{-1}) lower, and root biomass was 49% (0.005 g g^{-1}) lower in restored compared to control reaches in the 10-20 cm layer (Fig. 3.4; compare Urban values in Table 3.2 to Stability Restoration values in Table 3.3). In the 20-30 cm soil layer, bulk density was on average 11% (0.14 g cm^{-3}) higher, SOM was 12% (0.006 g g^{-1}) lower, and root biomass was 63% (0.004 g g^{-1}) lower in restored compared to control reaches (Fig. 3.4; compare Urban values in Table 3.2 to Stability Restoration values in Table 3.3). DEA did not differ significantly between restored and control reaches (F = 0.2508$_{(1, 26)}$, p = 0.62).

Analysis of vegetation data at stability restoration sites indicated that the difference in percent cover between restored and control reaches was not significantly different, whereas percent cover was marginally significantly different ($F = 0.2511_{(1, 9)}$, $p = 0.63$ and $F = 4.4642_{(1, 9)}$, $p = 0.06$ respectively). Percent invasive cover was on average 1.8 times higher (12 percentage points higher) in restored reaches relative to control reaches (Fig. 3.5).

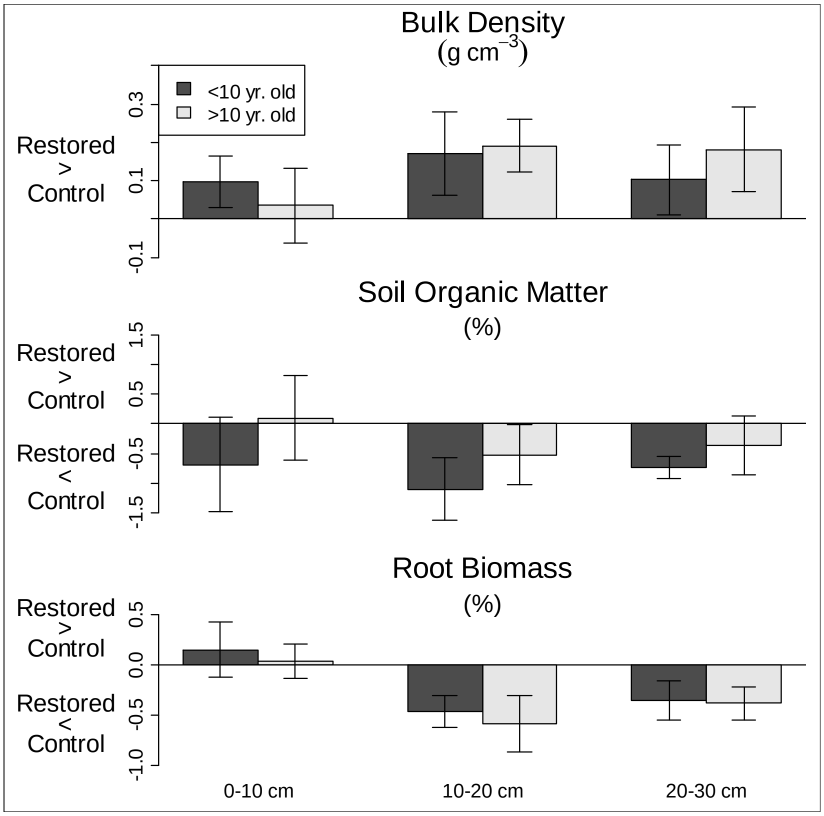


Figure 3.4. Differences in soil metrics between restored and control reaches at Coastal Plain stability restoration sites. Each bar is the average of differences at recent restoration (<10 years old, $n = 7$) or older restoration sites (>10 years old, $n = 6$) and each is a different soil layer (0-10 cm, 10-20 cm, and 20-30 cm). Negative values indicate that control values were higher than the restored values and positive values the reverse. Error bars are standard errors.

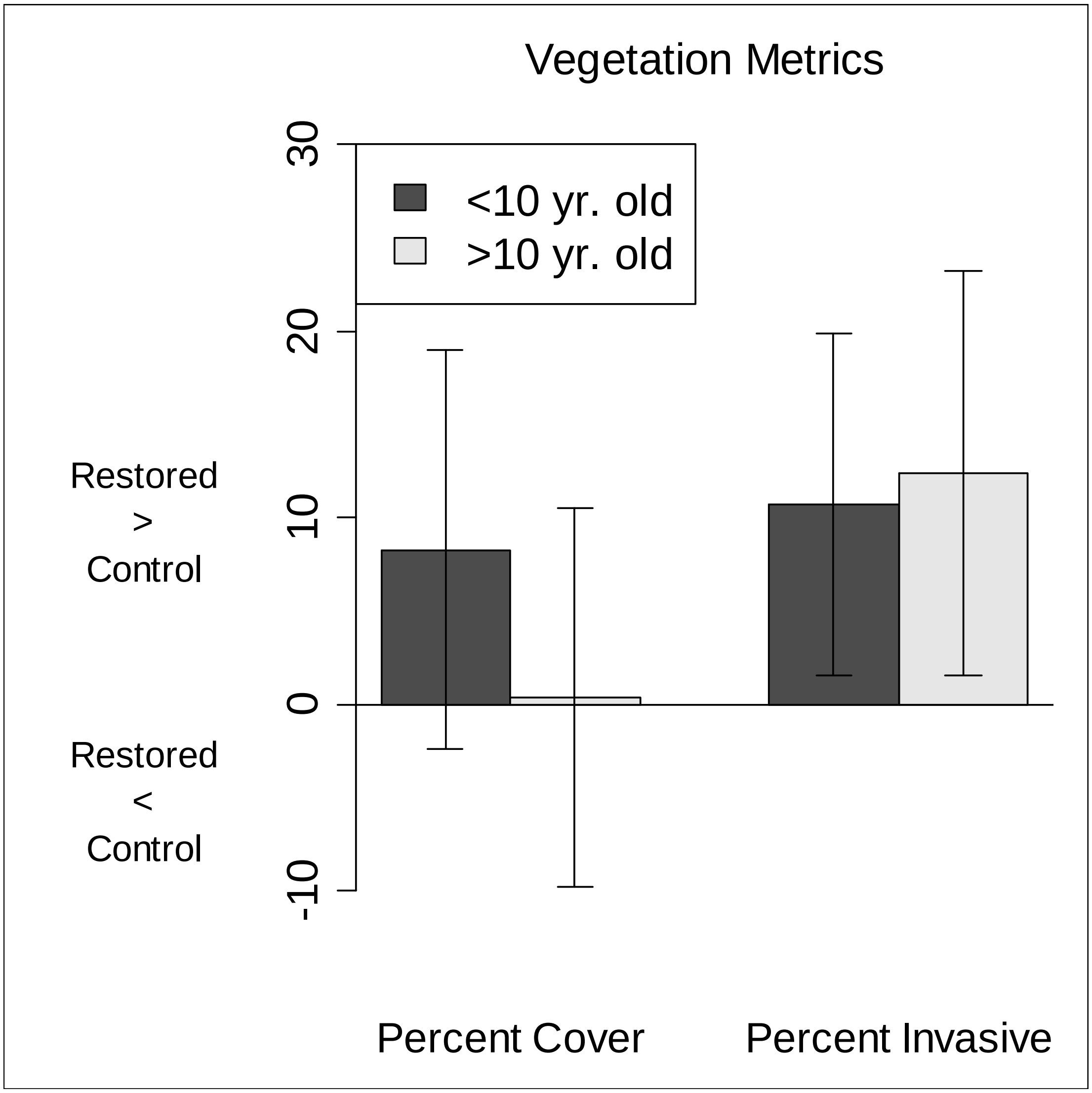


Figure 3.5. Differences in percent cover and percent invasive species between restored and control reaches at stability restoration sites separated by age class (Recent, <10 years old, *n* = 7; Older, >10 years old, *n* = 6). Negative values indicate that control reaches had higher percent cover and percent invasive species than upstream reaches.

Riparian Buffer Restoration Sites

Relative to stability restoration sites, riparian buffer restoration sites had on average lower bulk density and higher SOM, root biomass, and DEA at all soil layers except root biomass at the 0-10 cm soil layer, which was slightly lower compared to stability restoration sites (Table 3.3). Bulk density never exceeded values known to restrict root growth (1.5-1.8 g cm^{-3}; Daddow and Warrington 1983) at any riparian buffer restoration reach.

Bulk density, SOM, and root biomass did not differ significantly between the restored and control reaches at riparian buffer restoration sites ($F = 0.47_{(1, 22)}$, $p = 0.50$; Fig. 3.6). In analysis of DEA there was a significant reach by physiographic province interaction effect ($F = 15.045_{(1, 18)}$, $p = 0.001$) and therefore the difference in DEA between control and restored reaches was analyzed separately across the Piedmont and Coastal Plain sites. In the Coastal Plain, DEA did not differ significantly between restored and control reaches ($F = 0.697_{(1, 9)}$, $p = 0.42$). In the Piedmont, DEA was significantly different between restored and control reaches ($F = 18.89_{(1, 9)}$, $p = 0.002$), with restored reaches having on average twice as high DEA (146 μg N kg^{-1} hr^{-1} higher) compared to control reaches (Fig. 3.7; compare Urban values in Table 3.2 to Riparian Buffer Restoration values in Table 3.3). Neither percent cover nor percent invasive was significantly different between restored and control reaches at riparian buffer restoration sites ($F = 0.1355_{(1, 6)}$, $p = 0.72$ and $F = 0.0914_{(1, 6)}$, $p = 0.77$, respectively; Fig. 3.8).

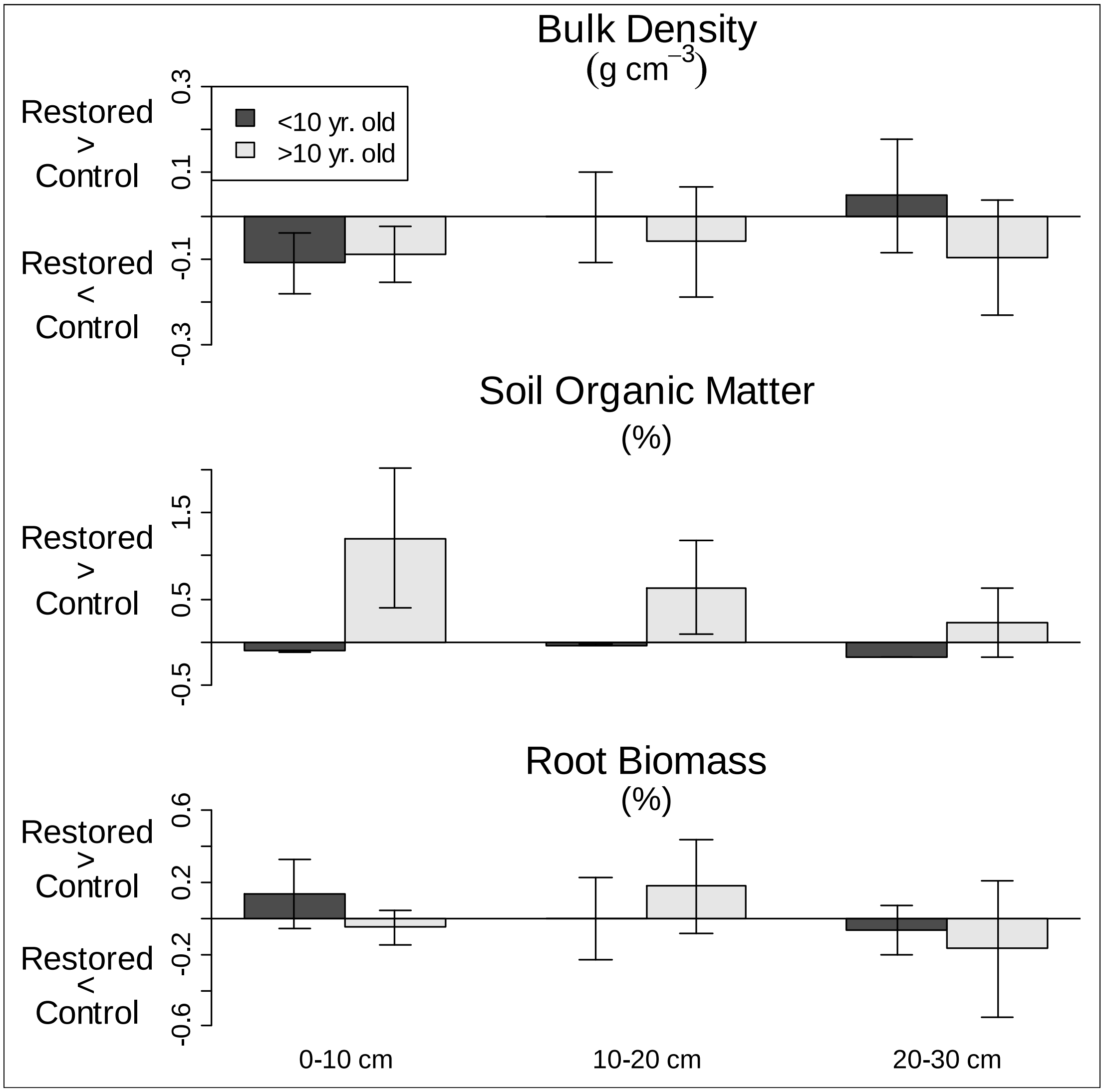


Figure 3.6. Differences in soil metrics between restored and control reaches at riparian buffer restoration sites. Each bar is the average of differences at recent restoration sites (<10 years old, *n* = 5) or older restoration sites (>10 years old, *n* = 5) and each is a different soil layer (0-10 cm, 10-20 cm, and 20-30 cm). Negative values indicate that control values were higher than the restored values and positive values the reverse. Error bars are standard errors.

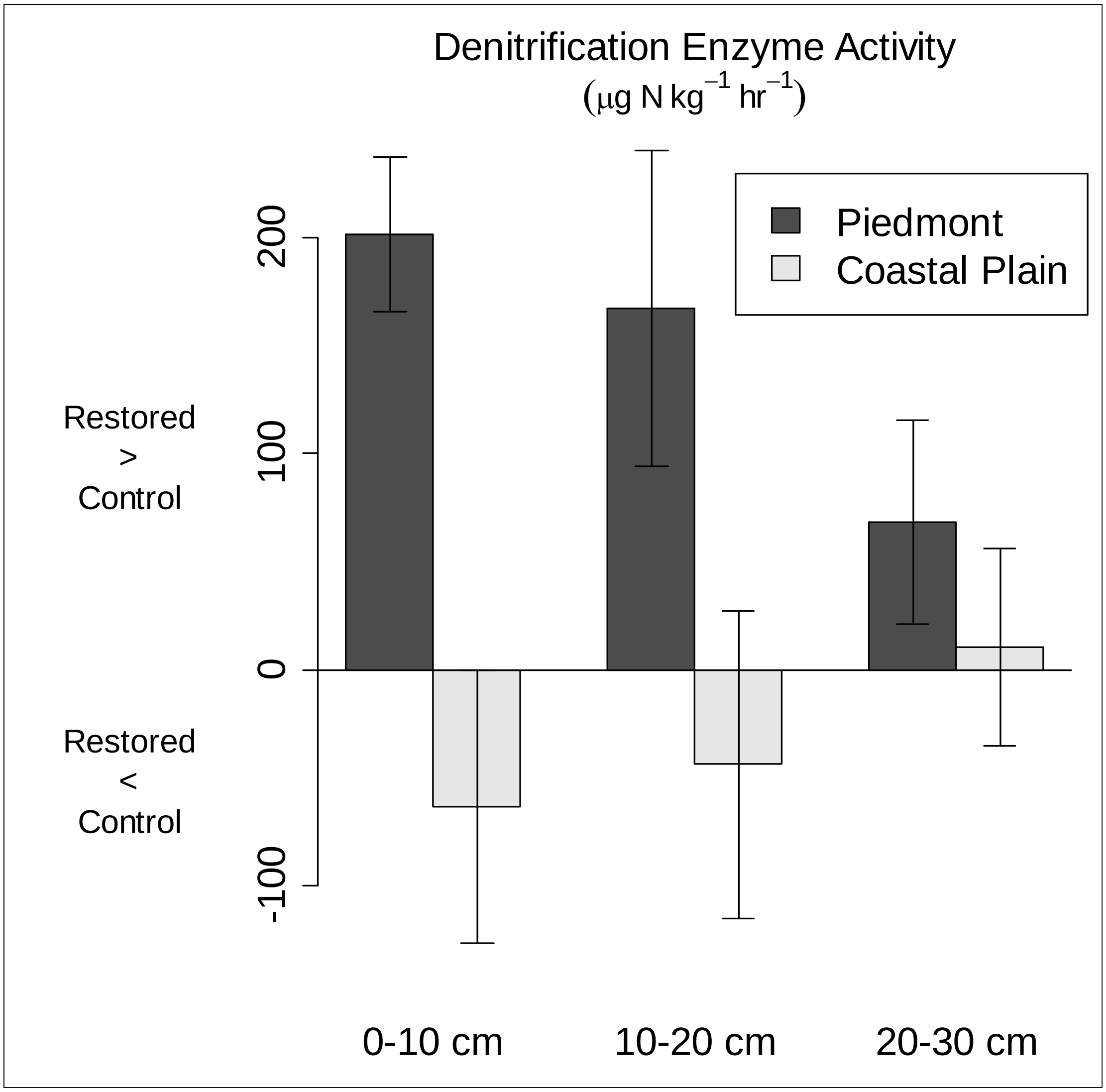


Figure 3.7. Differences in denitrification enzyme activity (DEA) between restored and control reaches at riparian buffer restoration sites. Each bar is the average of differences at sites in the Piedmont (*n* = 5) and Coastal Plain (*n* = 5) physiographic provinces and each is a different soil layer (0-10 cm, 10-20 cm, and 20-30 cm). Negative values indicate that control values were higher than the restored values and positive values the reverse. Error bars are standard errors.

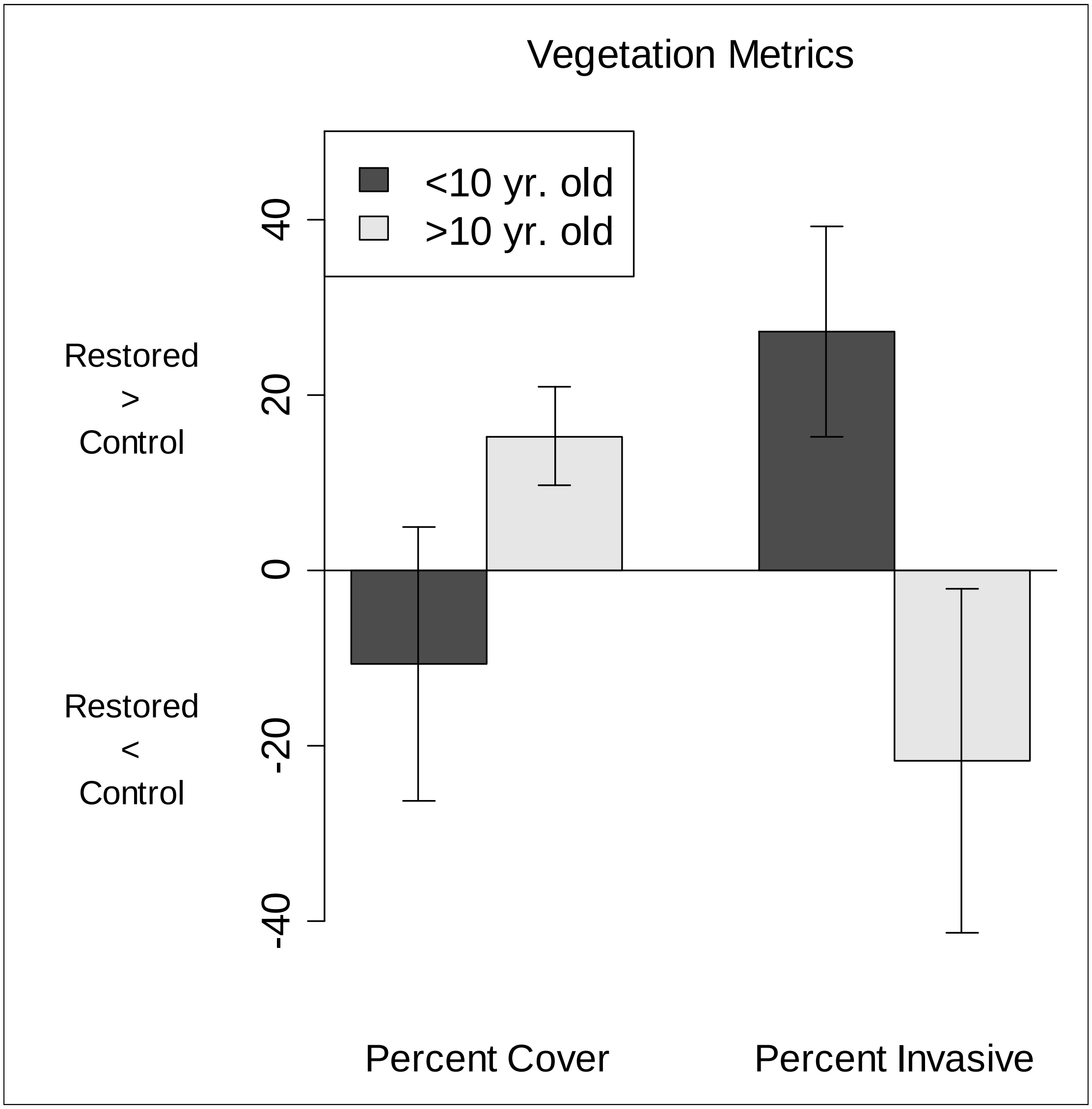


Figure 3.8. Differences in percent cover and percent invasive species between restored and control reaches at riparian buffer restoration sites separated by age class (Recent, <10 years old, *n* = 5; Older, >10 years old, *n* = 5). Negative values indicate that control reaches had higher percent cover and percent invasive species than upstream reaches. Error bars are standard errors.

Comparison of Forested, Urban, and Restored Soils

The first two components of the PCA explained 69%, 61%, and 58% of the variance in the 0-10, 10-20, and 20-30 cm soil layers, respectively. Biplots indicated that there were similar underlying gradients separating sites in the three soil layers (Fig. 3.9). The first axis of the 0-10 cm soils biplot represented a gradient from soils with high SOM, high root biomass, and low bulk density to soils with low SOM, low root biomass, and high bulk density. The second axis of the 0-10 cm soils biplot represented a gradient from soils with high percent fines, high DEA, and low C:N ratio to soils with low percent fines, low DEA, and high C:N ratio. The axes in the 10-20 and 20-30 cm soil layers represented similar gradients, with the exception that DEA was aligned with SOM and root biomass along the first axis.

Forested sites had on average lower bulk density, higher SOM, higher root biomass, higher moisture content, and lower DEA compared to urban and restored sites (Tables 3.2, 3.3). Stability restoration sites had higher bulk density in all soil layers and lower SOM and root biomass in the 10-20 cm soil layers on average compared to urban soils (Tables 3.2, 3.3), reflecting the significant difference between restored and control reaches at stability restoration sites reported above. Stability restoration sites also had higher bulk density and lower SOM, root biomass, and DEA in the 10-20 and 20-30 cm soil layers on average compared to riparian buffer restoration sites (Table 3.3). However, the magnitude of these differences between site types was often relatively small, as evidenced by the PCA biplots, in which there was no clear separation between forested, non-restored urban, or urban restored sites at any of the three soil layers.

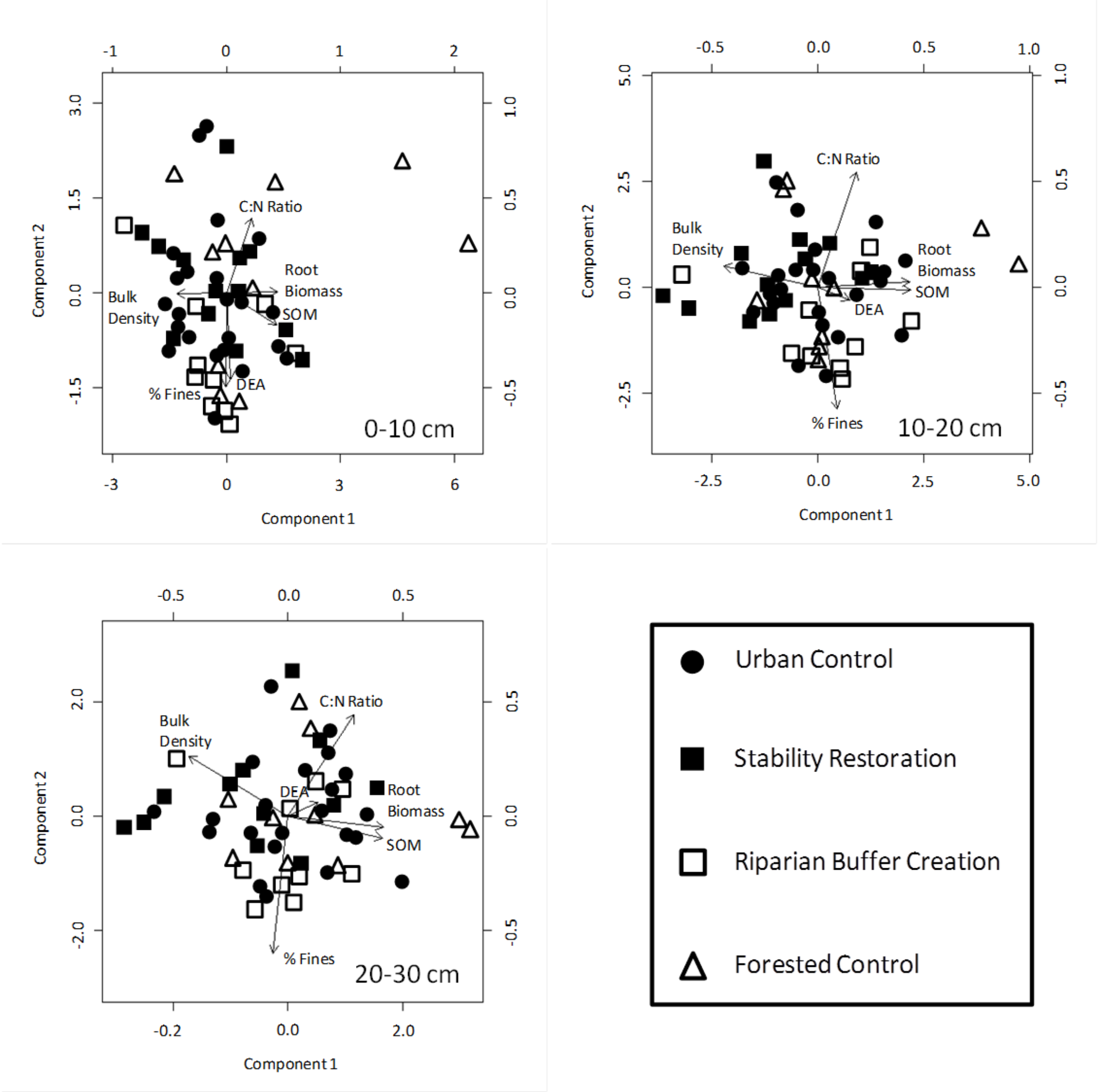


Figure 3.9. PCA biplots of soil metrics for the 0-10, 10-20, and 20-30 cm soil layers. Arrows indicate soil metrics and points indicate individual stream reaches. Stream reaches are divided into stability restoration reaches, riparian buffer restoration reaches, urban control reaches, and forested reaches. SOM = soil organic matter, DEA = denitrification enzyme activity.

Table 3.2. Average soil and vegetation metrics by depth class (cm) at forested and non-restored urban sites. Each cell lists the mean with the minimum and maximum in parentheses. SOM = soil organic matter, DEA = denitrification enzyme activity.

	Forested			Urban		
Soil Metric	0-10	10-20	20-30	0-10	10-20	20-30
Bulk Density ($g\ cm^{-3}$)	0.72 (0.14-1.21)	0.90 (0.35-1.11)	1.07 (0.63-1.33)	0.99 (0.77-1.25)	1.03 (0.73-1.3)	1.19 (0.83-1.56)
SOM (%)	7.8 (2.5-28.4)	4.5 (2.2-10.3)	3.6 (1.7-6.6)	4.7 (2.4-8.5)	3.8 (1.8-7.0)	3.3 (1.4-5.7)
Root Biomass (%)	2.2 (0.2-13.9)	0.8 (0.1-4.9)	0.7 (0.04-3.5)	0.4 (0.1-1.5)	0.5 (0.09-1.4)	0.4 (0.04-1.6)
Moisture Content (%)	47 (10-185)	34 (12-84)	29 (12-69)	25 (10-43)	25 (9-41)	24 (8-42)
Fines (%)	16.4 (5.5-30.8)	19.5 (4.8-30.8)	18.1 (6.0-27.8)	16.6 (4.7-29.4)	17.4 (6.7-30.7)	18.0 (8.9-30.6)
C:N Ratio	16.2 (11.8-25.3)	16.2 (11.2-21.2)	16.3 (11.9-21.7)	15.8 (11.1-28.5)	15.0 (10.6-24.2)	14.8 (9.3-25.1)
DEA ($\mu g\ N\ kg^{-1}\ hr^{-1}$)	167 (12-407)	83 (1-198)	63 (0.4-163)	291 (26-1128)	135 (11-494)	90 (0.6-326)
Percent Cover	46 (29-76)			48 (20-86)		
Percent Invasive	23 (0-75)			35 (0-83)		

Table 3.3. Average soil and vegetation metrics by depth class (cm) at stability and riparian buffer restoration sites. Each cell lists the mean with the minimum and maximum in parentheses. SOM = soil organic matter, DEA = denitrification enzyme activity.

	Stability Restoration			Riparian Buffer Restoration		
Soil Metric	0-10	10-20	20-30	0-10	10-20	20-30
Bulk Density (g cm^{-3})	1.02 (0.74-1.48)	1.18 (0.86-1.53)	1.30 (0.79-1.67)	0.90 (0.56-1.17)	0.97 (0.66-1.33)	1.14 (0.90-1.50)
SOM (%)	4.9 (2.6-9.7)	3.4 (1.9-6.5)	2.9 (1.5-5.1)	5.1 (1.5-11.0)	3.8 (1.3-5.8)	3.3 (1.5-4.5)
Root Biomass (%)	0.6 (0.1-1.9)	0.2 (0.04-0.5)	0.2 (0.03-0.7)	0.5 (0.1-1.3)	0.5 (0.04-1.4)	0.3 (0.06-0.7)
Moisture Content (%)	28 (10-52)	24 (10-45)	24 (12-43)	34 (13-74)	29 (10-51)	27 (10-42)
Fines (%)	15.0 (6.4-24.7)	17.2 (8.4-26.6)	17.3 (7.6-28.2)	21.2 (6.8-32.4)	22.9 (10.2-38.3)	23.3 (8.7-48.1)
C:N Ratio	17.2 (10.7-31.3)	16.4 (10.7-28.9)	15.9 (10.9-29.3)	12.9 (10.7-15.2)	13.2 (10.8-15.8)	12.8 (10.1-14.6)
DEA (μg N kg^{-1} hr^{-1})	400 (19-1733)	130 (6-671)	54 (5-272)	436 (84-734)	247 (38-462)	155 (24-382)
Percent Cover	50 (23-82)			53 (27-93)		
Percent Invasive	41 (3-95)			45 (17-79)		

DISCUSSION

As concerns over the health of the world's rivers and streams have grown, so too has the field of restoration ecology. For running-water ecosystems, the historical roots of restoration science can be traced to engineering disciplines and grew largely out of an emphasis on flood control (Palmer and Bernhardt 2006). Not unlike stormwater management projects a decade or more ago, the goal was largely to move water rapidly off the land and use streams as water conveyance systems. However, there is now a growing body of science demonstrating that enhancing infiltration of water into soils and allowing overbank flows (floodplain inundation) when storms exceed infiltration capacity is a more ecologically sustainable practice (Williams et al. 2009, Kline and Cahoon 2010). While many stream restoration projects today focus on these goals, the methods that are employed are highly variable and efforts are only now underway to quantify the ecological impacts or benefits of various methods (e.g., Filoso and Palmer 2011).

This study examined the hypothesis that restoration projects employing hard engineering methods such as grading stream banks and channel reconfiguration (stability restoration) result in the degradation of riparian soil structure and function. The results show that there were statistically significant impacts at stability restoration sites, primarily an increase in bulk density and a reduction in root biomass in sub-surface soil layers, and a moderate increase in the abundance of invasive species, and these impacts persisted for at least 10 years post-restoration. In contrast, riparian buffer restoration had generally neutral effects on riparian soils with some positive effects on denitrification enzyme activity (DEA) in the Piedmont region. Although neither type of restoration showed many positive effects on riparian soils relative to soils in urban non-restored sites, comparison of soil properties in these restored and urban sites to forested sites indicated that the soil properties in urban non-restored sites were not significantly different from those in most forested sites (Fig. 3.9). Therefore, with respect to the soil properties measured in this study, the soils in restored streams appear to have been relatively unimpaired prior to restoration activities.

I looked specifically at whether bulk density, SOM, root biomass, DEA, percent vegetative cover, and percent invasive species differed between restored reaches and upstream or downstream control reaches across two age classes (<10 years old and >10 years old) and three soil depths (0-10 cm, 10-20 cm, and 20-30 cm). No variables showed a significant age class by reach interaction, indicating that at both stability restoration projects and riparian buffer creation projects there was little change over time in the difference in soil properties between restored and control reaches. This lack of interaction between reach and age class may be due to the fact that restored and control reaches at recently restored sites did not differ significantly with respect to most soil variables, and thus there was no opportunity for soil properties to recover over time. However, there were a few variables that differed significantly between restored and control reaches at both newly restored sites and older restored sites, suggesting the alternative explanation that changes in soil properties were slow post-restoration and were not detectable within the timeframe of this study.

Stability Restoration Effects

Bulk density, SOM, and root biomass were statistically significantly different in stability restoration reaches relative to control reaches in both the 10-20 and 20-30 cm soil layer in Coastal Plain study sites. Differences were not significant in the Piedmont sites, however, there was low power to detect differences in the Piedmont sites (n = 2 stability restoration projects) and I therefore focus on the Coastal Plain sites here.

In both the 10-20 and 20-30 cm soil layers in Coastal Plain sites, bulk density was on average higher and SOM and root biomass were on average lower in both recent and older restoration sites compared to control sites (Fig. 3.4). The magnitude of the differences between restored and control reaches in both soil layers was relatively small for bulk density, being less than 0.2 g cm^{-3} (<20%) higher on average in the restored compared to control reaches in both soil layers (Fig. 3.4). However, at five of the 11 sites, bulk density in subsurface soil layers at restored reaches was within the range of values known to restrict root growth in silty soils (1.5-1.8 g cm^{-3}; Daddow and Warrington 1983), whereas bulk density in subsurface soil layers of control reaches at the same sites were below this range. Thus, the detectable increase in bulk density following restoration is in some cases indicative of compacted soils.

The magnitude of the difference in SOM between restored and control reaches at Coastal Plain sites was small (<10% lower in restored relative to control sites, see Fig. 3.4) relative to previous work on the effects of restoration on SOM. In a study of similar restoration projects in Maryland, Gift et al. (2010) found a reduction in SOM of about 40% at a depth of 10-30 cm in restored compared to urban control sites. Another study of a stream restoration project in North Carolina found a 33% reduction in SOM following restoration, although they sampled only the top 10 cm of soil, which had a higher SOM content (9.6%) than the 10-20 and 20-30 cm layers sampled here (maximum SOM of 7% - see Table 3.3) (Unghire et al. 2010).

In the surface soil layer (0-10 cm), SOM was not significantly lower in restored sites relative to control sites, which contrasts with the previous study of stream restoration by Unghire et al. (2010) but is in agreement with the previous study of stream restoration by Gift et al. (2010). One possible reason why SOM was similar in restored and control reaches in the surface soil layer is that the refill material used in stability restoration is often top soil removed during the grading and excavating process, which is typically high in SOM (Allison 1973). In addition, fertilizer and mulch are typically mixed into the soil during vegetation planting at the end of construction work. The most recently restored site sampled in this study (two years old at time of sampling) did have reduced SOM in the surface soil layer relative to a control section, which suggests rapid recovery of SOM in shallow soil as an alternative explanation to the finding of no difference in SOM in surface soil layers between restored and control reaches. However, several restoration projects that were 5-9 years old at the time of sampling had higher SOM in the restored reach relative to the control reach, which means that SOM would have had to recover within 3-5 years if recovery of SOM in shallow soil layers was the reason for the lack of a difference in SOM between restored and control reaches. Previous work has shown that development of soil organic carbon to equilibrium levels in riparian forests takes much longer, on the order of decades or longer (Bush 2008).

Root biomass was reduced from about 0.4% to 0.2% of dry soil weight at both the 10-20 and 20-30 cm soil layers in Coastal Plain sites (Fig. 3.4 and compare Urban and Stability Restoration in Tables 3.2 and 3.3). The magnitude of this loss is comparable to a previous study of root biomass at a similar soil depth in similar restoration projects in Maryland (Gift et al. 2010), suggesting that stability restoration reduces root biomass at these soil layers significantly. During stability restoration, large trees are often removed and channel banks are commonly graded and sometimes backfilled to achieve a designed cross-section profile. Once grading is complete, banks are reseeded and replanted with vegetation. Results presented here suggest that the establishment of vegetation is sufficient to recover root biomass in surface soil layers (0-10 cm), since root biomass was similar between restored and control reaches in this layer, but not in the 10-20 and 20-30 cm soil layers. Gift et al. (2010) attributed reduced root biomass at these depths to a lack of deep-rooted vegetation in restored sites. Lower root biomass in the 10-20 and 20-30 cm soil layers in this study may also be attributable to lack of deep-rooted vegetation because trees and shrubs are often planted sparsely in restored reaches (as evidenced by project design plans). Lower root biomass in subsurface soil layers may also be attributable to soil compaction, as bulk density in subsurface soil layers in multiple restored reaches exceeded values known to restrict root growth. Compaction is likely to have occurred through the use of heavy machinery for bank grading and placement of instream structures, as has been observed in previous studies of similar restoration activities as those involved in stability restoration (Merz and Finn 1951, Campbell et al. 2002, Bruland and Richardson 2005).

Taken together, the difference in soil properties between restored and control sites in the Coastal Plain suggest that activities associated with stability restoration were at least a mild disturbance for riparian soils. This disturbance may have been sufficient to promote the establishment and persistence of invasive vegetation despite planting of native vegetation at all stability restoration sites surveyed. Soil disturbance can facilitate establishment of invasive species (Burke and Grime 1996, Olander et al. 1998, Davis et al. 2000, Jesson et al. 2000, Morgan and Carnegie 2009), and results from this study indicated that percent invasive (number of invasive species observations relative to all vegetation observations) was on average almost twice as high in restored reaches compared to control reaches in stability restoration sites (Fig. 3.5).

The lack of any age class by reach interaction effect in the analysis of soil and vegetation metrics indicated that the magnitude of the differences in bulk density, SOM, root biomass, and percent invasive between restored and control sites was similar in both the newer (<10 years old) and older restoration sites (>10 years). This suggests that there was little recovery in riparian soils from the impact of stability restoration after more than 10 years. These results are consistent with a previous study of two restoration projects in Maryland restored in 1997 and 1999, which found that both SOM and root biomass were lower in restored sites relative to control sites in a 10-30 cm soil layer (Gift et al. 2010). In contrast, previous work on riparian buffer restoration in the Piedmont region has found that 10 years is usually sufficient to recover water quality benefits similar to natural riparian buffers (Orzetti et al. 2010).

Denitrification enzyme activity (DEA) was highly variable between restored and control sites and did not show a strong response to stability restoration. Denitrification rate is influenced by organic matter content under certain conditions (Duncan and Groffman 1994, Pavel et al. 1996, Ullah and Faulkner 2006, Hernandez and Mitsch 2007, Sutton-Grier et al. 2010), which suggests the possibility that the reduction in SOM caused by stability restoration was not of sufficient magnitude to influence denitrification potential. However, it is also possible that other factors besides SOM that were not measured in this study, such as soil nitrate (Groffman et al. 1991, Jordan et al. 1998), were limiting denitrification. I was also unable to sample restored sites before and after restoration, which could also explain why I did not detect a large effect of restoration on DEA, given that soil properties were variable between control and restored sites in the absence of restoration.

Riparian Buffer Restoration Effects

Riparian buffer restoration sites showed no difference in overall soil properties or vegetation metrics between restored and control reaches (Figs. 3.6, 3.8). DEA was significantly higher in restored sites relative to control sites across both age classes and all soil depths, but only in Piedmont sites (Fig. 3.7). DEA in restored reaches in the Piedmont was on average twice as high as DEA in control sites. Thus, there is evidence that soils in riparian buffer restoration sites in the Piedmont were more favorable for denitrifying bacteria and could potentially process more nitrate than non-restored soils. It is possible that increased SOM was responsible for this increase in DEA. Even though the analysis combining Coastal Plain and Piedmont sites showed no difference in soil metrics between restored and control sites, all restored reaches in the Piedmont had higher SOM at all soil depths compared to control reaches except the 20-30 cm soil layer at one site. However, as with DEA in stability restoration sites, the possibility that other factors besides SOM were controlling DEA makes it unclear whether restoration was driving increased denitrification rates or whether background differences between restored and control reaches were responsible. In addition, DEA is a measure of potential denitrification under nitrate-saturated conditions. Thus, even if DEA is higher in riparian buffer restoration soils, actual denitrification will not necessarily be higher because denitrification will require interaction of soils with groundwater (Groffman et al. 1992, Hunter and Faulkner 2001, Groffman et al. 2006, Orr et al. 2007).

Previous work has shown that restoration of riparian buffers was associated with improvements in water quality and macroinvertebrate communities in Piedmont streams 5-10 years after establishment (Orzetti et al. 2010). In general, restored riparian buffers have been found to reduce nutrient runoff to streams, and can have significant benefits for water quality within 5-10 years (CBP 1997, Claussen et al. 2000, Lee et al. 2003), though results may be variable (Parkyn et al. 2003). The results presented here suggest the possibility that riparian buffer restoration had a moderate benefit for denitrification in riparian soils, at least in Piedmont streams. Although there is insufficient evidence to say with certainty that riparian buffer restoration in the Piedmont enhanced denitrification potential, the fact that there were no detectable negative impacts from riparian buffer creation up to 13 years after restoration, combined with previously reported benefits of riparian buffer establishment, suggests that riparian buffer creation is a more ecologically favorable restoration design than stability restoration.

Comparison of Urban, Restored, and Forested Soils

For restoration projects to facilitate recovery of riparian soils from urban conditions to forested conditions, there must be differences between urban and forested soils, which I did not find at these study sites. Although forested sites had on average lower bulk density, higher SOM, and higher root biomass compared to urban sites (Table 3.2), forested soils grouped closely with urban reaches in the PCA biplot (Fig. 3.9), indicating the magnitude of the differences in soil properties between these two types of streams was not substantial on average.

Previous work has shown that soils in urban areas have higher metal concentrations relative to rural soils (Paterson et al. 1996, Thuy et al. 2000, Manta et al. 2002) and that riparian soil profiles and associated biogeochemical processes change with urbanization (Groffman et al. 2003). Thus, it is possible that soils in the urban study sites were impacted by urban development, but the impacts were not detected with the metrics used in this study. One forested stream had significantly higher SOM, higher root biomass, and lower bulk density than all urban streams, and it is possible that this stream best represents the forested condition in the study area. In any case, the PCA results show that the magnitude of the effects of stability restoration and riparian buffer creation were low relative to the variation in soil properties seen between sites.

Conclusion

This study suggests that stability restoration projects have relatively minor but negative and lasting impacts on several ecologically important soil and plant characteristics while riparian buffer restoration projects may be associated with positive to neutral effects on soils and vegetation. The large number of restoration projects sampled relative to previous studies of the impacts of stream restoration on riparian soils (at least 10 of each type compared to two and one in Gift et al. 2010 and Unghire et al. 2010, respectively) makes these conclusions relatively robust.

The grading, channel construction, and installation of instream structures with heavy machinery typical of stability restoration projects appeared to constitute a minor disturbance to riparian soils. Bulk density was increased in restored reaches, in some cases to levels typically associated with compacted soils (Daddow and Warrington 1983). SOM was decreased in restored reaches, but not to the same extent seen in previous studies of stream restoration projects (Gift et al. 2010, Unghire et al. 2010). In addition, the magnitude of changes in SOM was minor relative to variation of this property between urban and forested reference streams (Tables 3.2 and 3.3). However, root biomass was significantly reduced in sub-surface soil layers and percent invasive species was on average higher in restored reaches compared to control reaches. In addition, there was no evidence of recovery of any soil properties over the time span covered by restoration projects (16 years). These results add to previous work, which has shown that spatial variability in SOM, DEA, and nutrient concentrations may be substantially reduced following restoration activities (Bruland and Richardson 2005, Bruland et al. 2006, Unghire et al. 2010). Riparian buffer establishment appeared to enhance denitrification potential in the Piedmont and such projects were less susceptible to establishment of invasive species, suggesting they are a more ecologically benign stability restoration method.

These results suggest that the benefits of stability restoration projects for riparian soils may be few. Given the high cost of these projects, alternative approaches should be considered. One way to reduce the impact to soils and the cost of stability restoration projects would be to forego bank grading and channel reconstruction and install only instream structures, which can reduce erosion and provide grade control and beneficial habitat on their own if installed properly (Radspinner et al. 2010). If channels are highly incised, then designs could include raising the channel invert by filling and using instream structures (e.g., Brown 2000, Shea et al. 2005), i.e., working only in the active channel to avoid disturbance of riparian soils. In addition, the need for hard engineering stability restoration projects should be critically examined and riparian buffer restoration projects should be considered as an alternative restoration design where lack of riparian vegetation contributes to channel degradation.

Chapter 4: Diatom community response to urbanization and stream restoration in the Coastal Plain physiographic province, Maryland

ABSTRACT

The effects of geomorphic stability restoration projects on benthic diatom communities have rarely been investigated. However, the importance of benthic algae to stream food webs and primary productivity and the widespread implementation of stability restoration projects, particularly in urban areas, suggest that any impacts of restoration on diatom communities could be important for stream ecosystems. I sampled benthic diatoms in urban streams subject to stability restoration projects in the Coastal Plain physiographic province of Maryland. I compared richness, Simpson's diversity index, β diversity (calculated as the difference in species composition between streams), and community composition in restored streams to nearby forested and urban stream sites to determine how urbanization was impacting diatom communities and whether stability restoration projects mitigated any effects of urbanization on diatom communities. Diatom richness and diversity did not differ significantly between restored, urban, and forested streams. β diversity among forested streams was also similar to β diversity among urban streams and among restored streams. These results suggest that urban development does not necessarily cause a loss of benthic diatom diversity in individual streams and that stability restoration has little effect on benthic diatom

communities. However, multivariate analysis of species assemblages showed that communities in urban and restored streams were clearly separated from communities in forested streams. Differences in community composition appeared to be driven by a gradient of pH and conductivity, which were low in forested streams relative to urban and restored streams. Restoration had no detectable effect on species composition, though the possibility that restoration altered community properties not measured here (e.g., percent sensitive taxa, percent motile taxa) cannot be ruled out. Results of the species composition analysis suggest that urbanization-induced changes in water chemistry can significantly alter the species composition of benthic diatom assemblages. In particular, urbanization appears to cause a replacement of species characteristic of low pH, low conductivity streams with species characteristic of higher pH and conductivity. Continued urban development may thus drive a reduction in regional diatom diversity that is not mitigated by stability restoration.

INTRODUCTION

Human alteration of ecosystems continues to drive a decline in biodiversity throughout the world (Butchart et al. 2010). One of the leading causes of diversity decline is habitat loss or degradation, which has prompted efforts to mitigate the causes of degradation and to restore natural ecosystem structures and functions (Clewell and Aronson 2007**,** Nelleman and Corcoran 2010). Although the science of ecological restoration has progressed rapidly in the past few decades (Young et al. 2005**,** Falk et al. 2006), there is still a need to identify the drivers and patterns of biodiversity loss and to assess the ability of restoration to mitigate diversity loss, particularly for non-vertebrate taxonomic groups (Balian et al. 2008**,** Clausnitzer et al. 2009, Rey Benayas et al. 2009, Walpole et al. 2009, Rands et al. 2010).

Freshwater streams and lakes are some of the most impacted ecosystems in terms of species loss (Ricciardi and Rasmussen 1999, Sala et al. 2000, Dudgeon et al. 2006, Strayer and Dudgeon 2010, Vörösmarty et al. 2010). Threats to freshwater biodiversity include overexploitation, flow regulation by dams and irrigation, competition and predation pressures from invasive species, and land development, including urbanization (Meybeck 2003, Bour 2008, Lévêque et al. 2008, Strayer and Dudgeon 2010). Urban development leads to increased stormwater runoff, which can increase the frequency and intensity of erosive flow events (Dunne and Leopold 1978, MacRae and Rowney 1992, Hardison et al. 2009). In addition, water quality is often degraded by increased nutrient and toxicant concentrations (Paul and Meyer 2001, Walsh et al. 2005). Previous work has demonstrated that fish and invertebrate communities are negatively impacted when urban land cover in a watershed increases above about 15% or impervious surfaces increase above about 5-10% (Booth and Jackson 1997, Wang et al. 1997, Wang et al. 2000, Ourso and Frenzel 2003, Roy et al. 2003). Whether fish and invertebrate diversity follows a threshold response or a more linear response to impervious surface cover has been questioned (Booth et al. 2004, Cuffney et al. 2010), but there is consensus that diversity declines with increasing urban development.

Algal community response to urbanization is understood to a lesser degree compared to fish and invertebrate taxa, but studies have consistently shown a shift in species composition over a rural-urban gradient (Chessman et al. 1999, Sonneman et al. 2001, Fore and Grafe 2002, Newall and Walsh 2005, Bere and Tundisi 2011). However, the response of algal richness and diversity to urban development has been variable across studies (Sonneman et al. 2001, Bere and Tundisi 2011), suggesting the response may be regionally distinct (Potapova et al. 2005). Furthermore, algal community response to urbanization may be more strongly related to changes in water quality parameters such as nutrient concentrations than to the intensity of urban development (Winter and Duthie 2000, Coles et al. 2009).

Restoration targeted toward improvement of channel stability is a common approach used in urban streams to try and mitigate the impacts of increased stormwater runoff and associated bank erosion and channel incision (Shields et al. 2003, Booth 2005, Hassett et al. 2005, Radspinner et al. 2010). Though stability restoration projects are primarily concerned with establishing a designed channel form, habitat improvement is often stated as a project goal (Shields et al. 2003), and restoration of habitat complexity is often assumed to be beneficial to bitotic diversity (Palmer et al. 2010). A handful of studies have shown improvement in habitat condition and increased fish diversity following stability restoration (Shields et al. 1995, Baldigo et al. 2008, Ernst et al. 2010). However, other studies have failed to detect an increase in habitat complexity following restoration of urban streams (Tullos et al. 2009, Violin et al. 2011, Chapter 1), and the majority of research to date has shown that habitat restoration fails to increase diversity of fish and invertebrates (Roni et al. 2008, Palmer et al. 2010, Louhi et al. 2011, Stranko et al. 2011, Sundermann et al. 2011). One study of algal response to channel manipulation found a weak increase in diversity at a single restored reach (Passy and Blanchet 2007).

The conflicting results in the literature make it uncertain whether stability restoration is beneficial for biotic communities, and the effect on algal communities in particular remains an open question. Therefore, in this chapter, I investigate the impacts of urbanization and stability restoration on benthic diatom communities in small headwater streams of the Coastal Plain physiographic province, Maryland. Diatoms have received less attention than fish and invertebrate communities, but represent an important component of stream food webs as primary producers. Moreover, benthic algal diversity has been associated with primary production (Passy and Legendre 2006). Though this association may vary with frequence of stream disturbance (Cardinale et al. 2005), the streams I sampled have primarily sand beds, a large proportion of which is mobilized during storm events (Chapter 2), and therefore alterations to diatom communities may have broad implications for stream ecosystem functioning. In addition, benthic diatom assemblages are known to respond to disturbance at the reach scale (Leland 1995, Pan et al. 1999, Fore and Grafe 2002), suggesting they should be a useful indicator of the effects of urbanization and channel alteration through stability restoration.

To determine the impacts of urbanization and stability restoration on benthic diatom communities, I sampled benthic diatoms in multiple urban and restored streams and compared them to communities sampled from forested streams. The response of algal diversity to urbanization has been inconsistent (Sonneman et al. 2001, Potapova et al. 2005), and few studies have examined algal response to physical channel manipulation (Passy and Blanchet 2007), therefore my hypotheses are based on related studies of fish and invertebrate responses. Given that the diversity of fish and invertebrates generally declines with urbanization (Paul and Meyer 2001, Walsh et al. 2005) and that habitat restoration has generally failed to improve invertebrate diversity (Palmer et al. 2010), I hypothesized the following: 1) diatom communities in urban and restored streams would show decreased diversity relative to forested streams, 2) there would be greater species turnover among communities in forested streams compared to communities among urban and restored streams, and 3) community composition would be more strongly related to water quality indicators than restoration status (i.e., there would not be a detectable effect of restoration on community composition).

METHODS

Study Sites

Diatom sampling was conducted on August 27 and 29, 2009 at two forested streams, three restored streams, and four urban streams in Anne Arundel County, Maryland (N 39°03’00”, W 76°37’00”; Fig. 4.1). Both forested streams had <10% impervious surface cover in their watersheds, whereas all urban and restored sites had >17% impervious surface cover (Table 4.1). All restored streams and all but one urban stream were located in the city of Annapolis, Maryland. The remaining urban stream (South Fork Jabez Branch) was located near the two forested streams (Fig. 4.1) and had a watershed with the lowest amount of impervious cover of all urban streams (18% for South Fork Jabez Branch compared to >37% for the other urban sites). All restoration projects were stability restoration projects, involving a combination of channel manipulation and bank stabilization activities (see Chapter 1 for details).

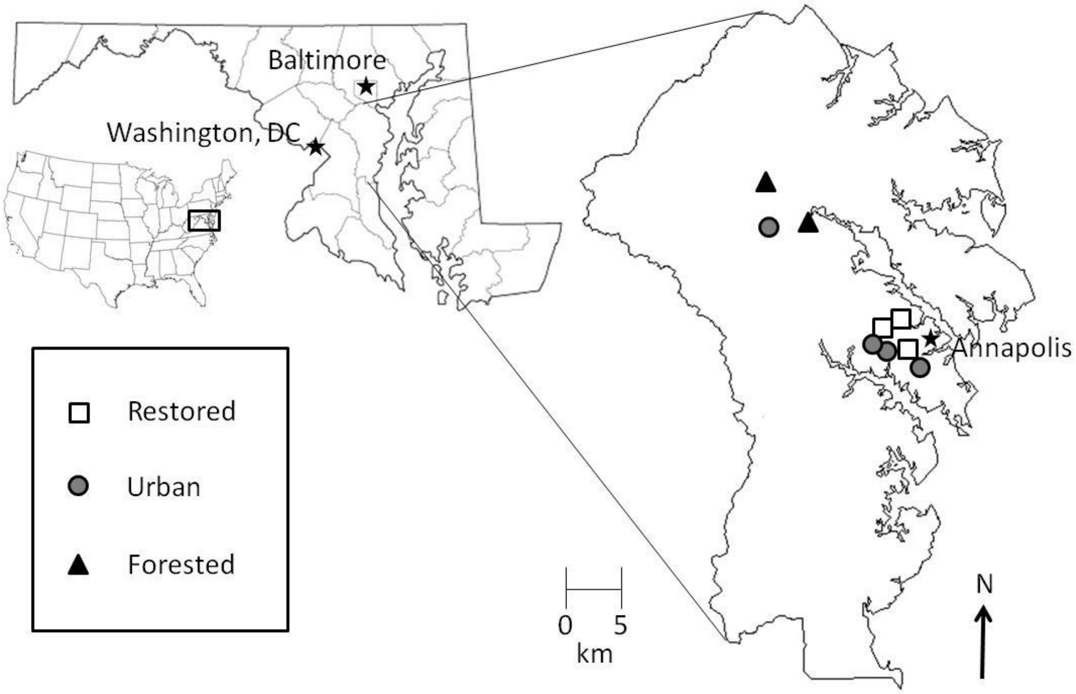


Figure 4.1. Map of Anne Arundel County, showing location of study streams.

Table 4.1. Land use characteristics for study streams. This is a subset of the streams listed in Table 1.1.

Stream (GPS Coordiantes)	Type (Year restored if restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Severn Run Trib.1 (SRT) (N 39°04’34”, W 76°37’07”)	Forested	2	2.1	31	10	55	14	0	100
Severn Run Trib. 2 (ST2) (N 39°06’21”, W 76°39’03”)	Forested	1	0.5	14	4	79	0	1	95
South Fork Jabez Branch (JAB) (N 39°03’60”, W 76°39’06”)	Urban	2	2.4	34	18	28	34	0	100
Harbor Center East (HCE) (N 38°58’33”, W 76°32’31”)	Urban	1	0.3	91	76	1	0	11	87
Harbor Center West (HCW) (N 38°58’29”, W 76°33’02”).	Urban	2	0.8	79	69	2	0	0	100

Table 4.1 cont.

Stream (GPS Coordiantes)	Type (Year restored if restored)	Order	Watershed Area (km^2)	% Urban in watershed	% Impervious in watershed	% Forest in watershed	% Ag in watershed	% Imp. in buffer	% Forest in buffer
Spa Creek Trib. at Hilltop Lane (SAH) (N 38°57’49”, W 76°30’31”)	Urban	1	0.1	89	63	6	5	0	96
Spa Creek (SPA) (N 38°58’23”, W 76°31’03”)	Restored (2001)	1	0.9	87	60	7	1	4	77
Weems Creek at Bristol Drive (WAB) (N 38°59’11”, W 76°31’10”)	Restored (1997)	1	0.5	79	38	10	0	1	95
Cowhide Branch (CB) (N 38°59’30”, W 76°32’14”)	Restored (1994)	1	0.4	85	67	15	0	0	100

Diatom Sample Collection and Processing

At each site, one diatom sample was taken from sand substrate in each of two flowing water habitats (i.e., runs) by pushing a Petri dish lid (28 cm^3 volume) into the substrate and capping with a spatula (Moulton et al. 2002). Samples at all sites were collected within two days of each other (August 27 and 29) and no significant rain events occurred between sample dates. Samples were cleaned by boiling in 25 ml 30% hydrogen peroxide (H_2O_2) for one hour, adding 30 ml 70% nitric acid (HNO_3), and boiling for an additional hour. Samples were then rinsed five times with DI water and concentrated to an appropriate density. Small aliquots of the sample mix were transferred onto glass cover slips, dried overnight and sealed to glass slides by heating in Z-Rax® mounting medium (W.P. Dailey, University of Pennsylvania, Philadelphia, Pennsylvania, USA) on a hot plate. Slides were viewed at 1000x power with a BH-2 compound microscope (Olympus Corporation, New Hyde Park, New York, USA). At least 200 diatom valves were identified to the variety level while scanning across a randomly selected row on the slide. If fewer than 200 valves were encountered on the first row, additional rows were randomly selected until at least 200 valves were identified. Taxa were not matched to known species but were instead identified by genus name and a unique number.

Water Chemistry Analysis

Water samples were collected at the downstream end of the study reach in each stream and were kept cool until returned to the laboratory. Conductivity and pH were measured on unfiltered samples within 24 hours of collection. Conductivity was measured with a multiprobe (model 85, YSI Incorporated, Yellow Springs, Ohio, USA), and pH was measured with a benchtop pH meter (Accumet excel XL15, Fisher Scientific, Pittsburgh, Pennsylvania, USA). Samples for ammonium (NH_4^+), phosphate (PO_4^{3-}), nitrite+nitrate (NO_2^-+NO_3^-), total dissolved phosphorus (TDP), total dissolved nitrogen (TDN), silicate (Si), chloride (Cl^-), sulfate (SO_4^{2-}), and dissolved organic carbon (DOC) were passed through a 0.7 mm glass fiber filter (Whatman, Kent, UK) and frozen until analysis. Ammonium was determined using the Berthelot reaction and Si was determined using the reduction of silicomolybdate to molybdenum blue in ascorbic acid; both were measured on a Technicon TrAAcs-800 nutrient analyzer (SEAL Analytical, Mequon, Wisconsin, USA). Nitrite+nitrate was determined using a copper-cadmium reduction column and TDP and TDN were determined using persulfate oxidation; all were measured on a Technicon Autoanalyzer II (SEAL Analytical, Mequon, Wisconsin, USA). Sulfate and Cl^- were measured using a DX 120 ion chromatograph (Dionex Corporation, Sunnyvale, California, USA). Dissolved organic carbon was measured on a TOC-5000 total organic carbon analyzer (Shimadzu Corporation, Kyoto, Japan). Measurements of all dissolved inorganic and organic analytes were conducted at the Nutrient Analytical Services Laboratory, Chesapeake Biological Laboratory, Solomons, Maryland, USA. Concentrations of all analytes at all streams are given in Appendix III.

Data Analysis

The South Fork Jabez Branch urban stream was considered an outlier compared to the other urban streams because of the low amount of impervious surface in its watershed relative to the other urban streams (18% for South Fork Jabez

Branch compared to >37% for the other urban sites). This site was included in multivariate analyses of species composition across all sites, but was not included as an urban stream in analyses of species richness, diversity, and β diversity.

Individual counts from the two sampling locations in each study stream were combined into one sample and only the first 400 valves counted at each study stream were analyzed. I calculated richness and Simpson's diversity for each stream. I tested the null hypothesis that richness and diversity did not differ among stream types (forested, urban, and restored) using separate ANOVAs for each metric. To assess the similarity of communities within stream types (forested, urban, and restored), I calculated β diversity for each stream type using the equation, $\frac{S}{\bar{\alpha}}$, where S = total number of species at all sites in the stream type and $\bar{\alpha}$ = average number of species at each site within the stream type (Whittaker 1972).

I used Correspondence Analysis (CA) combined with vector fitting of water quality variables to explore how similar streams were in terms of community composition and to determine whether water quality variables explained patterns of variation in community composition between streams. I performed two CAs: one with all sites included and one with just the urban and restored sites (excluding South Fork Jabez Branch). The second CA was used to assess the influence of restoration on diatom species composition by examining whether restored streams clustered separately from nearby urban streams. Both CAs were performed using the *cca* function and vector fitting was performed using the *envfit* function, both in the vegan package of R version 2.13.0 (Oksanen et al. 2011, R Development Core Team 2011).

RESULTS

Forested, urban, and restored streams had, on average, similar levels of richness and Simpson's diversity ($F = 0.6349_{(2, 5)}$, $p = 0.57$ and $F = 0.6067_{(2, 5)}$, $p = 0.58$ for richness and diversity, respectively; Fig. 4.2). β diversity (i.e., difference in communities among streams within a stream type) was slightly lower across forested streams compared to urban and restored streams, but the difference was small (difference of 0.16 between forested and urban streams; Fig. 4.3).

When all streams were included in the CA, all restored and urban streams except South Fork Jabez Branch clustered together in the resulting biplot, indicating that they had similar diatom communities (Fig. 4.4). South Fork Jabez Branch and the two forested sites were clearly separated from the restored/urban streams and from each other in the biplot (Fig. 4.4), indicating that each of these streams had a relatively unique species assemblage. The CA of just the restored sites and urban sites revealed no apparent clustering of either restored streams or urban streams (Fig. 4.5).

All water quality variables except NO_2^-+NO_3^- were highly correlated with the first CA axis of the biplot with all sites included. However, only pH and conductivity were significantly related to the variation in species composition among sites (Table 4.2). Nitrite+nitrate was not a significant water quality factor, but was the only variable highly correlated to the second CA axis. Therefore, I retained only pH, conductivity and NO_2^-+NO_3^- in the final biplot of the CA model (Figs. 4.4, 4.5). Vector fitting of these 3 water quality variables on the CA biplot with only restored and urban streams included showed that pH and conductivity were highly correlated with the first axis and $NO_2^-+NO_3^-$ concentration was highly correlated with the second axis. Thus, the first axis represented a gradient of increasing conductivity and decreasing pH (Fig. 4.5) and the second axis represented a gradient from low to high $NO_2^-+NO_3^-$ concentration (see also Appendix III).

There were 122 unique taxa identified across all streams, with an average of 41 species per stream. Dominant taxa (defined arbitrarily as those species attaining >10% relative abundance in a stream – though 10% has been used previously, e.g., Kelley (1998), Finney et al. (2000)) made up more than 50% of the species assemblage at the Severn Run Trib 1 forested site and the South Fork Jabez Branch and Spa at Hilltop urban sites, whereas dominant taxa made up less than 20% of the species assemblage at the Severn Run Trib 2 forested site and the Spa Creek restored site (Fig 4.6). Percent dominant taxa fell between 20% and 40% of the species assemblage at the remaining sites.

Achnanthes 5, *Achnanthes* 2, and *Navicula* 13 were common species at all restored and urban streams (excluding South Fork Jabez Branch; Fig. 4.7). *Navicula* 28 was common at two urban streams (Harbor Center West and Harbor Center East) and *Achnanthes* 13 was common at one urban site (Spa Creek Trib. at Hilltop Lane). Of these species, only *Achnanthes* 2 was common at one forested site (Severn Run Tributary 2). Common species at the two forested sites and South Fork Jabez Branch included *Eunotia* 1, *Eunotia* 7, and *Navicula* 1, which were all found in low numbers at some of the urban and restored sites. *Fragilariforma* 1 was common at South Fork Jabez Branch and was only found in low numbers at the two forested sites.

Table 4.2. Correlations of water quality variables with the first and second axes from the Correspondence Analysis with all sites included. R^2 is the squared correlation coefficient for the overall model and is the goodness of fit statistic that is tested for significance. Significance of the correlation coefficient is indicated by an asterisk (*).

Water Quality Variable	Correspondence Axis 1	Correspondence Axis 2	R^2
pH	0.989	-0.147	0.79*
Conductivity	0.997	0.074	0.69*
NH_4^+	0.988	0.154	0.05
PO_4^{3-}	0.989	0.147	0.07
$NO_2^-+NO_3^-$	-0.161	-0.987	0.64
TDP	0.922	0.124	0.07
TDN	0.955	-0.298	0.04
Si	0.998	-0.067	0.60
Cl^-	0.999	0.041	0.37
SO_4^{2-}	0.957	0.291	0.07
DOC	0.945	0.326	0.07

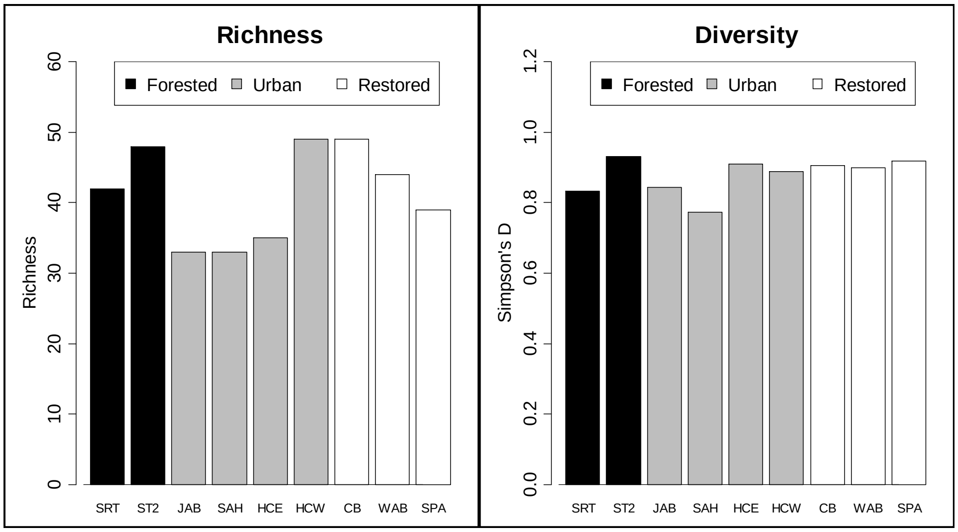


Figure 4.2. Richness (total number of species) and Simpson's diversity index at individual forested (black bars), urban (grey bars), and restored (white bars) streams. Metrics were calculated for the first 400 valves indentified at each stream.

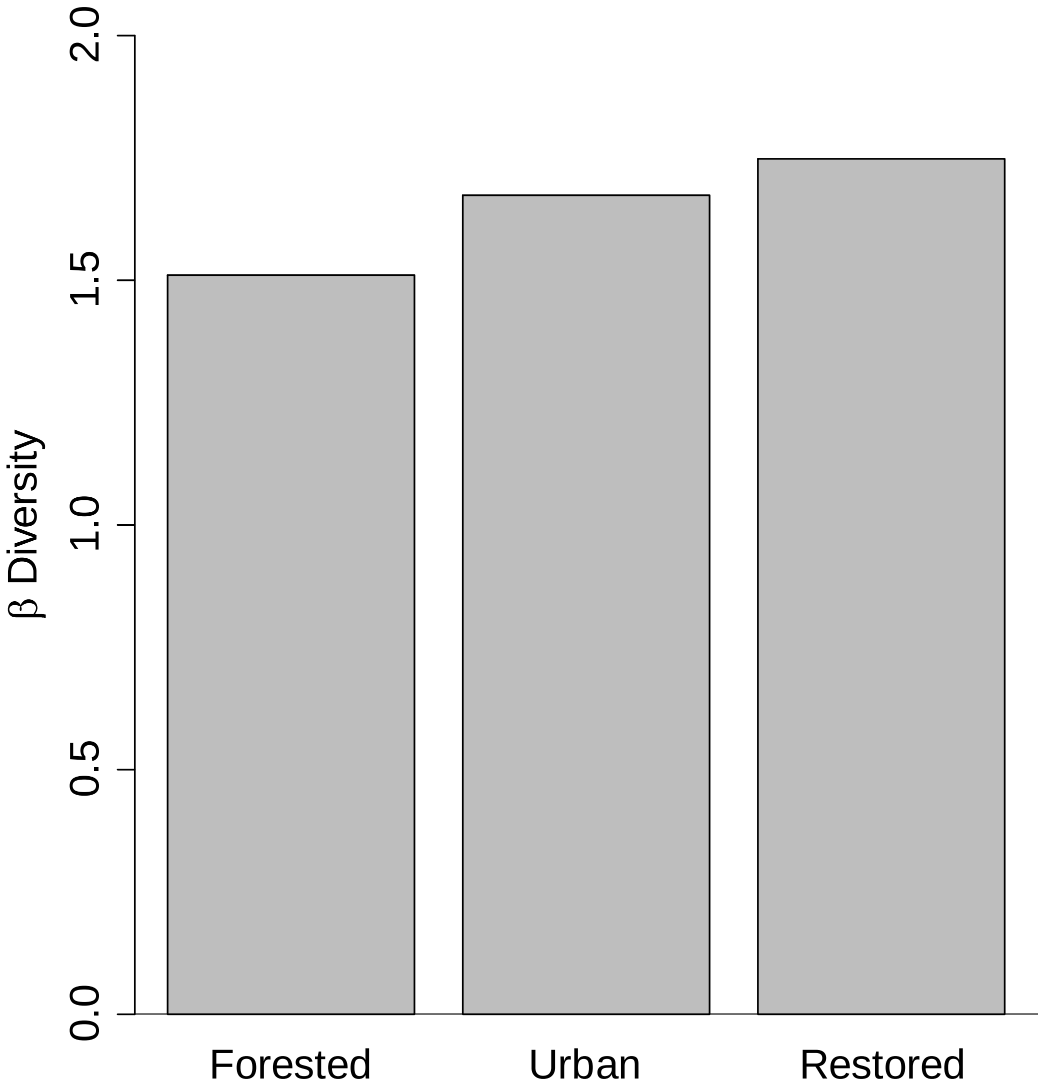


Figure 4.3. β diversity within forested, urban, and restored stream categories. β diversity was calculated as the difference in species assemblages between streams within a stream type. South Fork Jabez Branch was excluded as an urban site in calculations of β diversity.

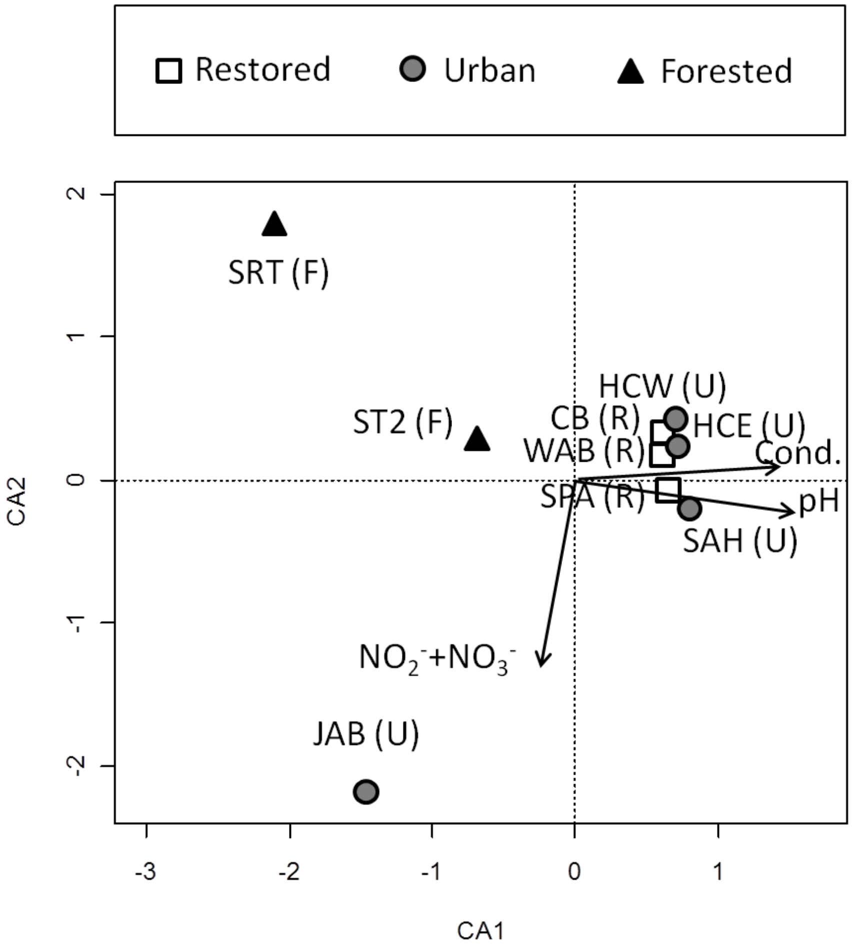


Figure 4.4. Biplot of the first and second axes from the Correspondence Analysis (CA) of diatom assemblages at the nine study sites. Stream sites are located with symbols and identified as three-letter codes with site type in parentheses. F = Forested sites: Severn Run Tributary 1 (SRT) and Severn Run Tributary 2 (ST2). U = Urban sites: South Fork Jabez Branch (JAB), Spa Creek Tributary at Hilltop Lane (SAH), Harbor Center East (HCE), and Harbor Center West (HCW). R = Restored sites: Cowhide Branch (CB), Weems Creek at Bristol Drive (WAB), and Spa Creek (SPA). Environmental variables are plotted as arrows ($NO_2^-+NO_3^-$ = nitrite+nitrate concentration (mg N/L), Cond. = conductivity (µS/cm)). As the focus of this plot is to see how sites were distributed, diatom species are omitted for clarity.

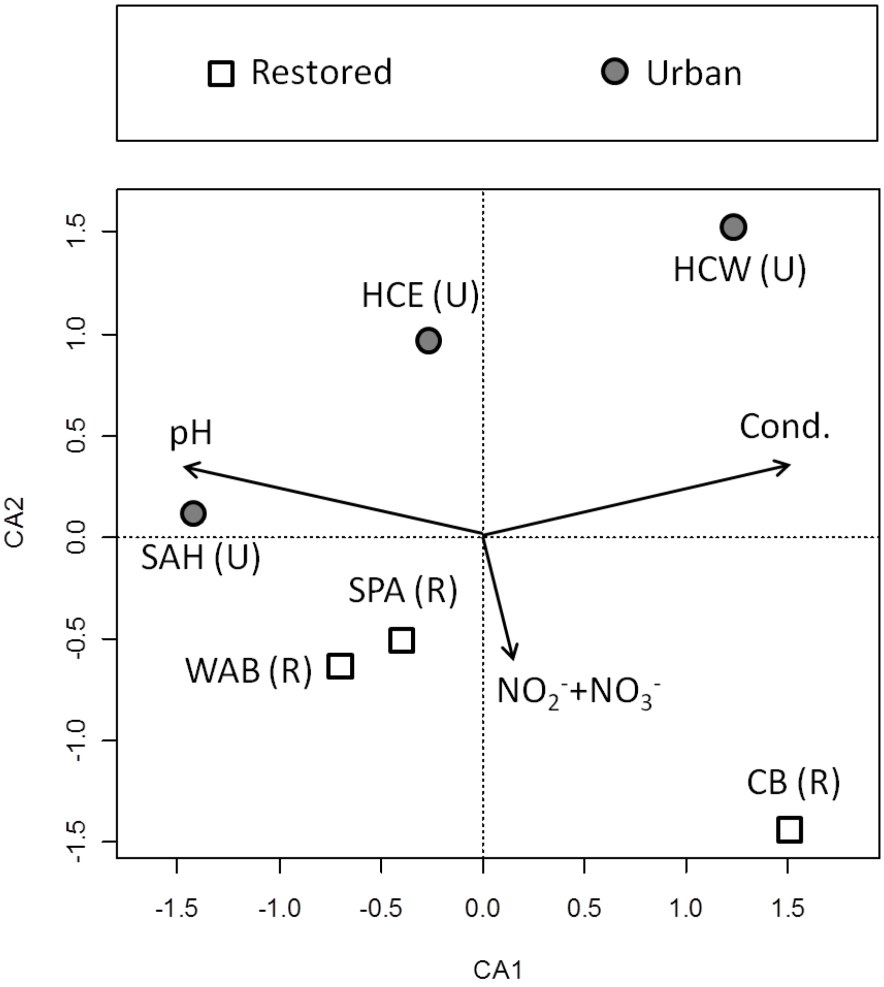


Figure 4.5. Biplot of the first and second axes from the Correspondence Analysis of diatom assemblages at the three restored streams and three of the four urban streams (excluding South Fork Jabez Branch). Stream sites are located with symbols and identified as three-letter codes with site type in parentheses. U = Urban sites: Spa Creek Tributary at Hilltop Lane (SAH), Harbor Center East (HCE), and Harbor Center West (HCW). R = Restored sites: Cowhide Branch (CB), Weems Creek at Bristol Drive (WAB), and Spa Creek (SPA). Environmental variables are plotted as arrows ($NO_2^-+NO_3^-$ = nitrate+nitrite concentration (mg N/L), Cond. = conductivity (µS/cm)). Individual diatom species are omitted for clarity.

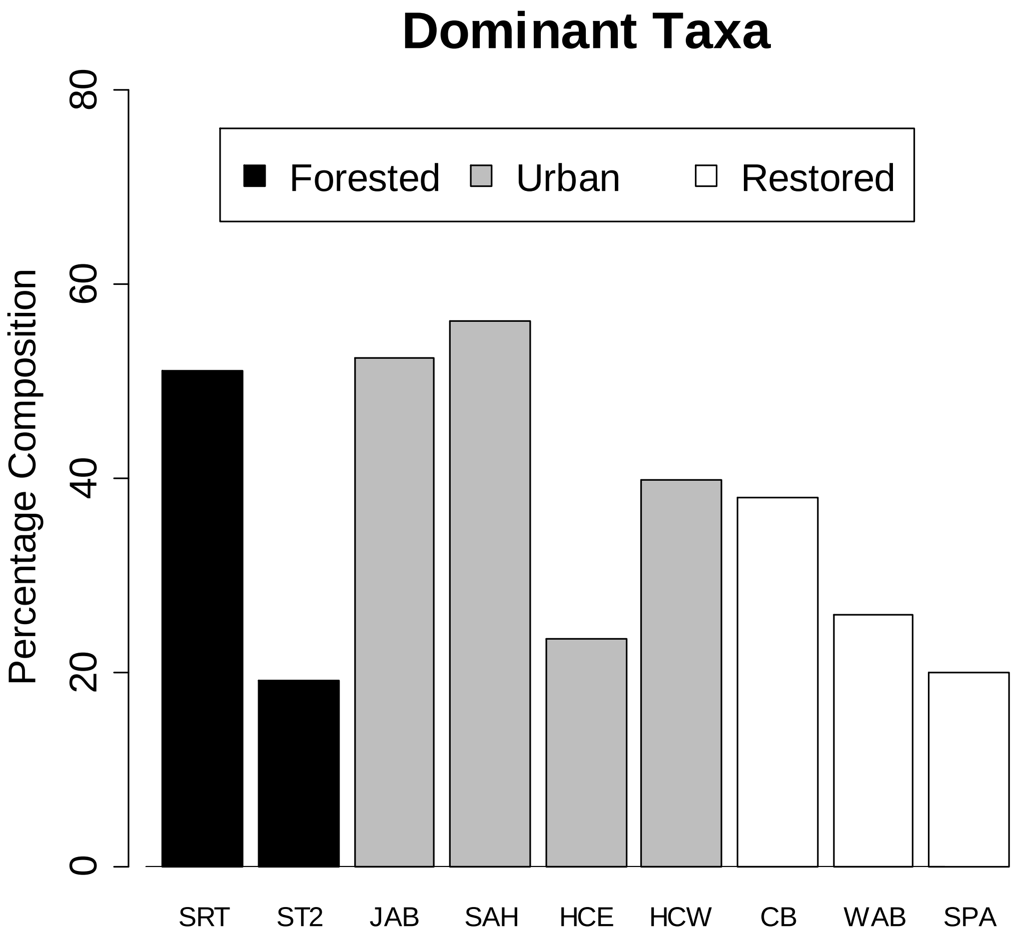


Figure 4.6. Relative abundance of dominant taxa (>10% of the species assemblage) at forested (black bars), urban (grey bars), and restored (white bars) streams. Forested sites: Severn Run Tributary 1 (SRT) and Severn Run Tributary 2 (ST2). Urban sites: South Fork Jabez Branch (JAB), Spa Creek Tributary at Hilltop Lane (SAH), Harbor Center East (HCE), and Harbor Center West (HCW). Restored sites: Cowhide Branch (CB), Weems Creek at Bristol Drive (WAB), and Spa Creek (SPA).

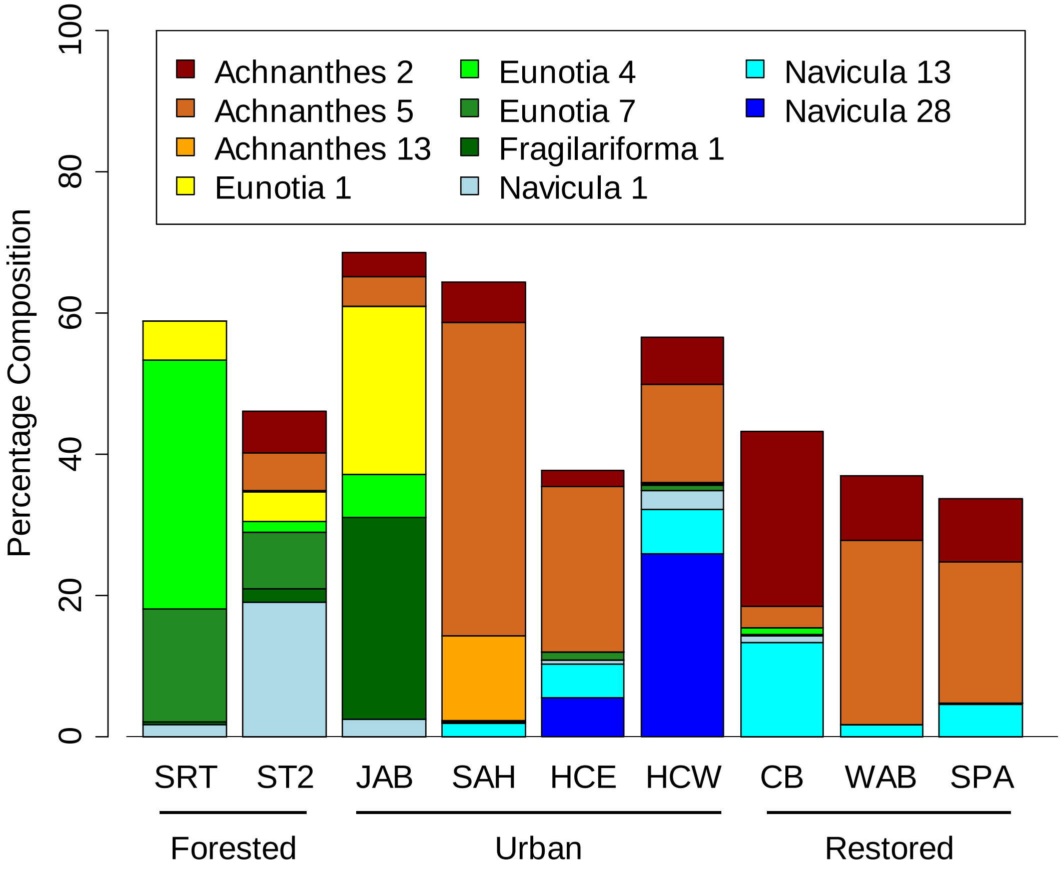


Figure 4.7. Relative abundance at each study stream of the 10 taxa that were dominant in at least one stream (>10% of species assemblage in an individual stream). Each color in a bar represents the relative abundance of that dominant taxon in that site, with species identified in the legend. Forested sites: Severn Run Tributary 1 (SRT) and Severn Run Tributary 2 (ST2). Urban sites: South Fork Jabez Branch (JAB), Spa Creek Tributary at Hilltop Lane (SAH), Harbor Center East (HCE), and Harbor Center West (HCW). Restored sites: Cowhide Branch (CB), Weems Creek at Bristol Drive (WAB), and Spa Creek (SPA).

DISCUSSION

The continuing loss of biodiversity on the global scale has prompted a great deal of interest in using restoration as a tool to reduce biodiversity loss (Rey Benayas et al. 2009, Bullock et al. 2011). Efforts at large-scale restoration in river systems are growing, with many examples of watershed-scale restoration projects now in existence (Williams et al. 1997). However, considerable amounts of money and effort are still invested in stream restoration projects that span only a few kilometers or less (Bernhardt et al. 2005). Previous work has generally shown that macroinvertebrate diversity does not increase following stability restoration in urban streams (Tullos et al. 2009, Stranko et al. 2011, Violin et al. 2011), but the response of algal communities has rarely been investigated (Passy and Blanchet 2007). Stability restoration projects are common throughout the US (Bernhardt et al. 2005), and any measureable impacts from such projects on algal diversity could be significant, particularly given that algal diversity has been linked with primary productivity in certain streams (Cardinale et al. 2005, Passy and Legendre 2006).

In this study I found that benthic diatom richness and diversity in urban, forested, and restored streams was comparable, suggesting that urbanization does not necessarily lead to a loss of diatom diversity in individual streams and that restoration has had little effect on diatom diversity. However, diatom community composition in forested streams was substantially different from communities in urban and restored streams, which had similar communities. Differences among communities appeared to be driven by water chemistry and suggest that urban development may drive some regional homogenization of diatom assemblages, which is not addressed by stability restoration.

Diatom Diversity in Urban Streams

I hypothesized that urban streams would have reduced species richness and diversity relative to forested reference streams, but results indicated that richness and diversity were similar in urban and forested streams. The fact that Simpson's diversity index was similar between forested and urban streams also suggests that evenness was not affected by urban development (i.e., the percentage of the species assemblage composed of dominant species was similar in urban and forested streams; Fig. 4.6). Thus, there is strong evidence that urban development has not substantially reduced benthic diatom diversity in individual streams in the Maryland Coastal Plain physiographic province.

This result contrasts with several studies on fish and macroinvertebrate diversity in urban streams, which have reported reduced diversity in urban streams relative to forested streams (Paul and Meyer 2001). However, many of these same studies in addition to several others have found that an intact riparian buffer along urban streams can mitigate the impacts of urbanization and help maintain diversity (Steedman 1988, Horner et al. 1997, Moore and Palmer 2005, Urban et al. 2006). The urban streams sampled in this study had >85% forested cover within a 30 m buffer along the study reach, which may explain why diversity was not reduced in the urban streams sampled.

The similar level of diversity in urban and forested streams may also be attributable to loss of diversity in forested streams compared to historic conditions (e.g., pre-European settlement). Both forested streams had some urban development in their watersheds and may have been impacted by past agricultural use, and therefore should be considered as best available reference streams (Stoddard et al. 2006). However, there was clearly no correlation between diversity and impervious surface cover. For example, one urban stream (Harbor Center East) had the highest level of impervious surface cover in the watershed but higher diversity than the other two urban sites, two of the restored sites, and one of the forested sites (Table 4.1, Fig. 4.2). This lack of correlation further supports the result that diversity was not reduced by urbanization.

In Chapter 1 I found no difference in channel complexity between urban and forested reference streams. Channel complexity has been used previously as a measure of habitat heterogeneity (e.g., Schlosser 1982, Shields et al. 1998, Milner et al. 2008, Violin et al. 2011), which theory suggests should increase diversity (Ricklefs and Schluter 1993). The fact that urban streams had relatively high channel complexity suggests the possibility that rather than being degraded, habitat heterogeneity in urban streams was sufficient to support an equally diverse diatom community as forested reference sites.

The similar levels of diversity between forested and urban streams may also be due to similar stream bed disturbance regimes. Stream bed disturbance can have a significant influence on algal communities (Peterson 1996), with studies generally supporting the hypothesis that streams with intermediate levels of disturbance have the highest diversity dependent on productivity (Lake 2000, Biggs and Smith 2002, Cardinale et al. 2006). I found that one forested site (Severn Run Tributary 1) and one urban site (South Fork Jabez Branch) had similar levels of bed sediment mobility at the reach scale (Chapter 2). I also found that grain size significantly influenced bed sediment mobility (Chapter 2). All urban and forested streams in this study exhibited similar grain size distributions (i.e., bed sediment was sand-dominated), which suggests that disturbance regimes should be comparable between the forested and urban sites. The possibility that similar disturbance regimes between forested and urban sites might counteract any negative impacts of urbanization to keep diversity similar needs to be tested in regions where urbanization does significantly alter the bed disturbance regime relative to forested reference streams.

Previous work on benthic diatoms also supports the conclusion that urbanization has had little impact on within-stream diversity. Passy and Blanchet (2007) found that degraded and channelized streams had reduced β diversity but similar levels of richness to forested streams. Sonneman et al. (2001) found that species richness did not vary consistently across a rural to urban gradient in Australia, and Potapova et al. (2005**)** actually found that diversity increased with urbanization in some metropolitan areas. Thus, the observation that benthic diatom diversity is not necessarily reduced following watershed urbanization appears to be a general pattern that applies beyond the focal region in this study.

Given the inconsistent response of diatom diversity to urbanization, future work on diatom response to urban development may need to consider alternative metrics besides richness and diversity. The reason is that some urbanization impacts can enhance diversity, for example, excess nutrients typical of urban streams (Walsh et al. 2005) may drive increased productivity, which has been linked to algal diversity – dependent on the disturbance regime (Cardinale et al. 2005). Therefore, metrics that incorporate autoecological information such as presence of sensitive taxa may be more useful as indicators of urban impacts than total community richness and diversity.

Diatom Diversity in Restored Streams

Restored streams did not differ significantly from urban or forested streams in terms of diatom species richness or diversity. This result was consistent with previous work that looked at response of fish and invertebrate diversity to habitat restoration, which has more often than not found no effect of restoration on diversity (see reviews by Roni et al. 2008, Palmer et al. 2010). Increases in biodiversity following restoration have generally been described in streams that were channelized, straightened, or cleared of all wood in the stream (Mueller and Liston 1994**,** Jungwirth et al. 1995, Brooks et al. 2004, Lepori et al. 2005, Muotka et al. 2002). Restored streams in this study were located in watersheds with a relatively high amount of impervious surface cover, but were not directly channelized or straightened prior to restoration. In addition, restored streams were found to have similar channel complexity to urban streams (Chapter 1), suggesting that habitat heterogeneity was not greatly improved by restoration, which may have limited the potential for diatom diversity to respond to restoration.

The condition of the diatom community prior to restoration is unknown, which leaves open the possibility that restoration did impact diatom diversity. In addition, I was unable to distinguish live from dead diatoms, and it is possible that much of the community I sampled was comprised of diatoms washed in from upstream. Although I did observe biofilms on the substrate at all sites at the time of sampling, which indicates there was living diatoms within the study reach, the inability to distinguish upstream-derived from local-derived diatoms limits the ability of this sampling to determine local-scale effects. Furthermore, community richness and diversity may not be the best indicators of restoration effects on diatom communities. For example, Passy and Blanchet (2007) suggested that β diversity within a single study site may be a more appropriate diversity metric to evaluate restoration than richness. Thus, while it is clear that restoration did not alter richness or diversity, investigation of additional diversity metrics would provide beneficial insight into whether and how restoration is altering diatom communities.

Diatom Communities

Although forested, urban, and restored streams were similar in terms of diversity, forested streams harbored unique diatom communities compared to most urban and restored streams. The difference in community composition between streams was seen in the Correspondence Analysis (CA) biplot of all sampled streams (Fig. 4.4). In this biplot, all urban and restored streams except the South Fork Jabez Branch urban site clustered together and were clearly separated from the two forested streams (Fig. 4.4). Furthermore, dominant species at the two forested sites and the South Fork Jabez Branch urban site were similar and distinct from dominant species at remaining urban and restored streams, which had similar dominant species (Fig. 4.7).

Differences in community composition among sites appeared to be explained by water chemistry, as water quality metrics correlated highly with the first two axes of the CA (Fig. 4.4, Table 4.2). The two forested sites and the South Fork Jabez Branch urban site were primarily separated from the other urban and restored sites by pH and conductivity, which were low in the forested sites and the South Fork Jabez Branch urban site relative to the other urban and restored sites (Appendix III). The South Fork Jabez Branch and the two forested sites were separated from each other by a gradient of NO_2^-+NO_3^- concentration, which was relatively high in the South Fork Jabez Branch urban site relative to the two forested sites (Appendix III).

The clustering of the South Fork Jabez Branch urban site with the two forested sites is not too surprising, given that the South Fork Jabez Branch had the lowest impervious surface cover of any urban stream sampled in this study – more than three times lower than the other urban streams. The South Fork Jabez Branch is also clustered along a latitudinal gradient with the two forested sites (Fig. 4.1). Groundwater conductivity and pH increase slightly along this latitudinal gradient (Chapelle 1983, Chapelle and Knobel 1983, Bolton and Hayes 1999), indicating that some variability in diatom communities between urban and forested streams may be due to natural variability in water chemistry, although urbanization is known to increase conductivity (Paul and Meyer 2001), e.g., through increased chloride inputs as a result of the application of roadway deicer (Kaushal et al. 2005).

Comparison of restored to urban streams was not complicated by latitudinal gradients because restored streams were clustered geographically with all urban sites except the South Fork of Jabez Branch. Correspondence Analysis of just these six sites indicated no distinct clustering of restored sites separate from the urban sites (Fig. 4.5), suggesting that restoration within urban streams had little effect on species composition. There was some separation of the urban and restored sites along the second CA axis, but this axis was highly correlated with NO_2^-+NO_3^- concentration, suggesting that the second CA axis represented a nutrient gradient rather than a restoration gradient.

It is possible that restoration altered diatom communities in ways that were not detected using only species abundances. For example, I found that restoration did alter bed sediment movement patterns, with patches of streambed near obstructions less likely to receive sediment deposition during flow events (Chapter 2). Streambeds receiving less deposition could promote a greater proportion of non-motile (attached) diatom taxa (Kutka and Richards 1996), which would not have been detected in this study. Future work incorporating metrics such as percent motile taxa should yield greater insight into diatom community response to restoration. Nevertheless, the strong correlation of water quality variables with community composition within urban and restored sites in Annapolis suggests that watershed-scale influences on water chemistry are a significant driver of benthic diatom community composition, a result that has been shown previously in regional studies of diatom assemblages (e.g., Pan et al. 1999, Fore and Graf 2002).

The close grouping of all urban sites except the South Fork Jabez Branch urban site in the CA with all sites included (Fig. 4.4) would seem to suggest that differences in community composition were greater among forested sites than among urban sites (i.e., β diversity higher in forested sites). However, the relatively wide distribution of urban sites along the first axis in the CA with only urban and restored sites included (Fig. 4.5), along with the computed values of β diversity (Fig. 4.3), show that β diversity was similar among urban sites and among forested sites when the South Fork Jabez Branch site was included as a forested site. The wide separation between forested and urban sites in the CA with all sites included instead suggests that urbanization alters water chemistry which in turn drives a shift in diatom species composition. Shits in diatom species composition could impact stream ecosystem functions, because the identity of species in a community and their functional roles and interactions, rather than diversity per se, may ultimately control ecosystem functions (Tilman et al. 1997, Hooper et al. 2005, Cardinale et al. 2011). In addition, increasing urban development in the region could cause a regional decline in biodiversity as species characteristic of forested sites are lost. Such an effect of urbanization has been described previously (McKinney 2006), but additional sampling of urban and forested sites in the region will be necessary to resolve the latitudinal differences between sites and determine whether specific species are consistently absent in urban streams.

Conclusions

I surveyed benthic diatom assemblages in forested, urban, and restored streams to gain an understanding of how urbanization alters diatom communities and whether stability restoration can mitigate these impacts. Diatom richness and diversity were similar in forested and urban streams, suggesting that in contrast to most studies of fish and invertebrates, diatom diversity within individual stream reaches is not adversely impacted by urbanization. Similarly, the amount of difference in species composition among forested sites was similar to that among urban sites, which, given that the urban streams were confined to a relatively small geographic area (a few adjacent watersheds), implies that β diversity among streams in neighboring watersheds may be maintained even in the face of urban development. However, the clear differences in community composition between forested and urban streams suggests that over larger scales, urbanization may homogenize diatom assemblages as species unique to forested sites are lost.

Stability restoration projects had no detectable effect on diatom richness, diversity, or species composition, though the possibility that restoration did alter other properties of diatom communities (e.g., percent sensitive taxa, percent motile taxa) cannot be ruled out. Species composition appeared to be driven primarily by differences in water chemistry, primarily gradients of pH, conductivity, and nutrient enrichment. Thus, it is unlikely that the implementation of this common and widespread restoration technique has affected diatom diversity to a significant degree. Efforts to maintain and restore regional diatom diversity will instead need to address water quality changes, which will require restoration efforts that extend beyond physical manipulation of individual stream reaches.

Conclusions

I investigated the impacts of channel stability restoration on stream ecosystems, measuring the response to restoration of in-channel geomorphic forms, geomorphic processes, and diatom communities, and beyond the channel, the response of riparian soil structure and function. Although previous work has explored restoration-driven changes to both the physical structure and resident biological communities of streams in urban areas, there has been little integration of research on geomorphic forms and processes with research on biological communities and the structure of riparian soil and plant communities. I have researched both the geomorphological and ecological impacts of multiple channel stability restoration projects in the Coastal Plain Physiographic province, Maryland.

Channel stability restoration had little effect on geomorphic channel form, measured as channel complexity (Chapter 1), and little effect on diatom communities (Chapter 4) as both channel complexity and diatom communities were similar in restored streams compared to urban streams. I attribute the lack of an effect of restoration to overriding influences beyond the in-channel reach-scale at which restoration was performed.

In the case of channel complexity, my results indicate that streams in urban watersheds do not necessarily have reduced channel complexity, as channel complexity in urban and forested streams was similar. Thus, there is little opportunity for restoration of urban streams to improve channel complexity since it is not reduced to begin with. The finding of relatively high channel complexity in urban streams contradicts a widely held assumption that urban development reduces channel complexity (Walsh et al. 2005). These results are partly explained by the fact that the urban streams I studied were not intentionally straightened or channelized. However, all urban study streams had an extensive riparian forest buffer, and I found no relationship between impervious surface cover in the watershed and channel complexity, suggesting that a forested riparian buffer can mitigate the impacts of urbanization on channel complexity.

Diatom communities in restored and urban streams were generally similar to each other but substantially different from communities in forested streams. The separation of communities was well explained by a gradient of pH and conductivity and conductivity is known to increase with urban development (Paul and Meyer 2001). Therefore, water chemistry changes induced by urban development in the watershed appeared to be the primary driver of diatom communities, and this is not changed by geomorphic restoration on the reach scale.

Channel stability restoration did alter patterns of bed sediment movement (Chapter 2), channel movement rates (Chapter 2), and riparian soils and vegetation (Chapter 3) to some degree. Bed sediment movement patterns were altered by the addition of channel and bank stabilization structures during restoration, which acted as flow obstructions and decreased the probability of grain movement in surrounding bed patches during high flow events. This effect translated into a reduced proportion of bed sediments mobilized on the reach scale, but only in streams with both sand and gravel in the bed sediments. Channel widening and thalweg scour (a measure of channel incision) were reduced in some restored streams in one period relative to an urban stream, which was attributed to the alteration of channel form rather than the placement of bank protection features, which had no effect on bank erosion rates. Restoration may have succeeded somewhat in meeting the goals of stabilizing channels, but whether the restoration was ecologically beneficial remains in question, particularly given that forested stream channels were highly dynamic geomorphically.

The ecological impacts of stability restoration on riparian soils were generally negative, with increased bulk density and invasive species and decreased root biomass in restored streams. In comparison, riparian buffer restoration had generally neutral effects on riparian soils, with some benefit observed for denitrification potential in Piedmont sites.

Overall, channel stability restoration affected patterns of bed sediment movement and channel movement rates (Chapter 2) at the expense of negative impacts to riparian soils and vegetation (Chapter 3). Even the effect on channel movement rates was seen to be overwhelmed by watershed-scale influences by the fact that at one restored stream upstream-derived sediment inputs caused significant channel aggradation (Chapter 2). Stability restoration had no effect on channel complexity (Chapter 1), even though the restoration was designed to alter channel form, and had no detectable effect on diatom communities using the metrics examined here (Chapter 4). Taken together, these results suggest impacts to stream ecosystems from urban development are not effectively managed by in-channel restoration at the reach-scale, but instead will require watershed-scale restoration efforts. For example, stormwater management is likely to alter geomorphic process rates, because it addresses the cause of channel incision and bank erosion, namely altered hydrology. The importance of watershed-scale restoration has been recognized (Williams et al. 1997), but this work adds support to this conclusion, particularly in showing that even restoration that directly manipulates channel form in an effort to alter geomorphic processes has no ecologically significant impact on channel form and that impacts on geomorphic processes can be overwhelmed by watershed-scale influences.

This is not to say that stability restoration has no ecological effect on stream ecosystems. The alteration of bed sediment disturbance patterns by the addition of flow obstructions (Chapter 2) may alter the resistance and resilience of benthic algal communities to flood disturbances (Mulholland et al. 1991, Peterson and Stevenson 1992, Matthaei et al. 2003, Effenberger et al. 2006), which in turn could impact primary productivity and nutrient assimilation which are responsive to algal biomass changes (Fisher et al. 1982, Martí et al. 1997). The impact of stabilization structures on these processes deserves further investigation. However, installation of instream structures can be effective at stabilizing channels without bank grading and channel reconstruction (Radspinner et al. 2010), which could reduce the cost of stability restoration projects and reduce the impact to riparian soils observed here. In addition, riparian buffer conservation and restoration is often used to stabilize stream banks (Hassett et al. 2005) and should be considered as an alternative to stability restoration in urban streams, particularly given the positive to neutral benefits of riparian buffer restoration observed here (Chapter 1 and Chapter 3) and the fact that establishment of forested buffers will eventually provide flow obstructions to the channel in the form of logs and other wood.

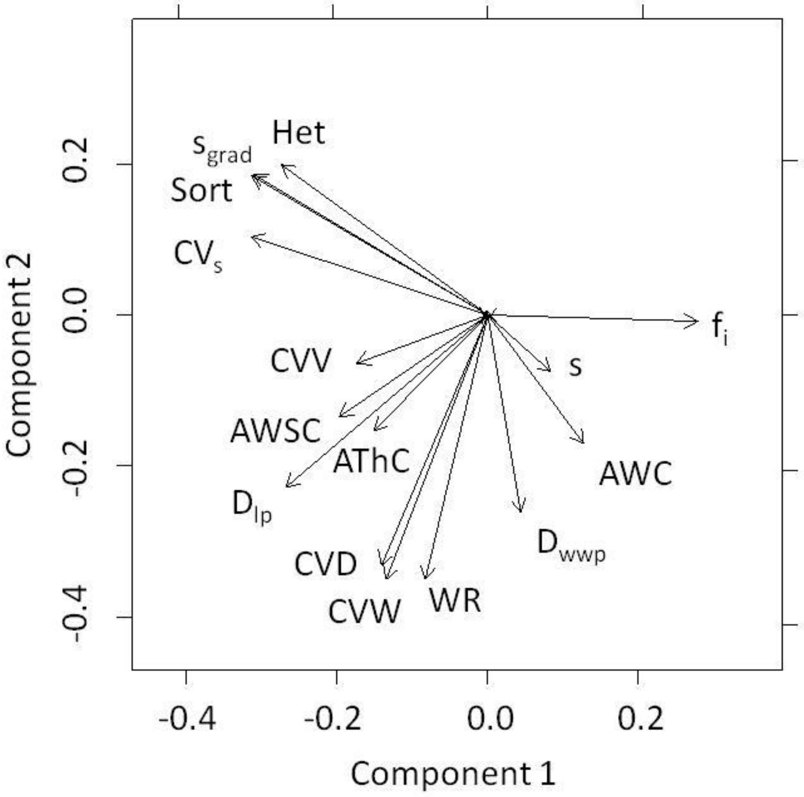
This work helps answer important questions regarding the impacts of urban development on stream ecosystems and how to manage such impacts. In particular, I have added to understanding of the impacts of urban development on channel complexity and diatom communities and addressed the effectiveness of channel stability restoration in altering geomorphic processes and its impacts on stream and riparian ecosystems. This work should be valuable in efforts to manage the impacts to stream ecosystems from future urban development.

Appendix I

Metric Abbreviations

AWC = Average Width Concavity
AWSC = Average Water Surface Concavity
AThC = Average Thalweg Concavity
CV_s = Sediment CV
CVD = CV of Maximum Depth
CVV = CV of Maximum Velocity
CVW = CV of Width
D_{lp} = Fractal Mean of the Longitudinal Profile
D_{wwp} = Fractal Mean of the Wetted Width Profile

LR = Longitudinal Roughness
F_i = Fredle Index
Het = Sediment Heterogeneity
s = Sinuosity
s_{grad} = Gradation Coefficient
SD = Stnd. Dev. Thalweg Profile
Sort = Sorting
WR = Width Residual

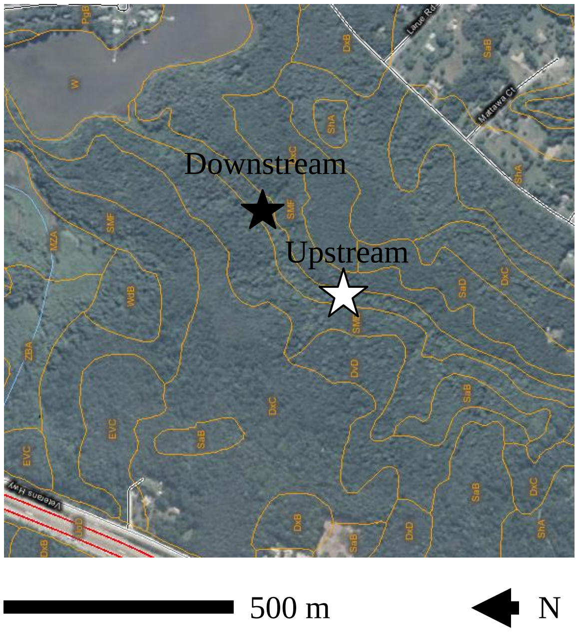


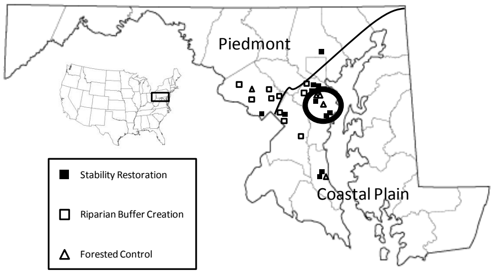
Appendix I. Biplot of components 1 and 2 from the PCA using all 17 complexity metrics showing complexity metric vectors. Loadings of LR and SD were very low on components 1 and 2, therefore labels for these metrics and all sites are omitted for clarity.

Appendix II – Soil Maps

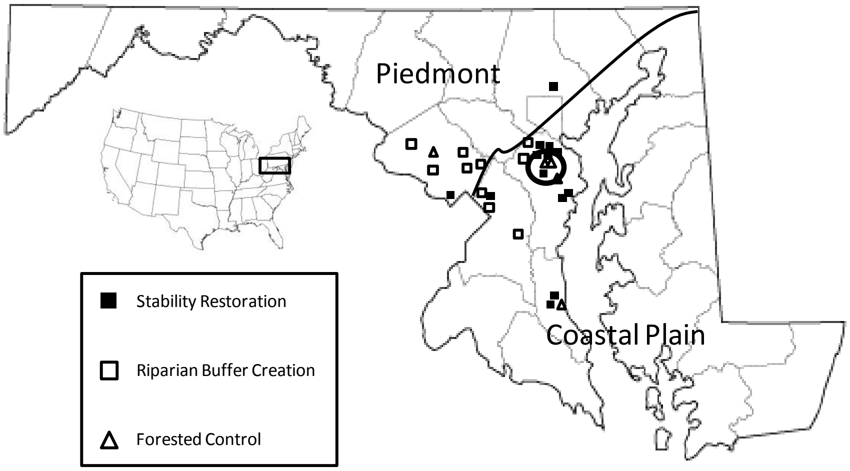
Each page of this appendix contains a table listing the site (F = forested, S = stability restoration, R = riparian restoration), soil series, whether the soil is upland- or floodplain-derived, the soil texture, and the drainage class of the soil. Following the table is the soil map of the site, with the restored and control reaches (or upstream and downstream reaches at forested sites) identified by white and black stars, respectively. Below the soil map is the map of where sites are located in Maryland.

Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Severn Run Tributary 1 (F)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained

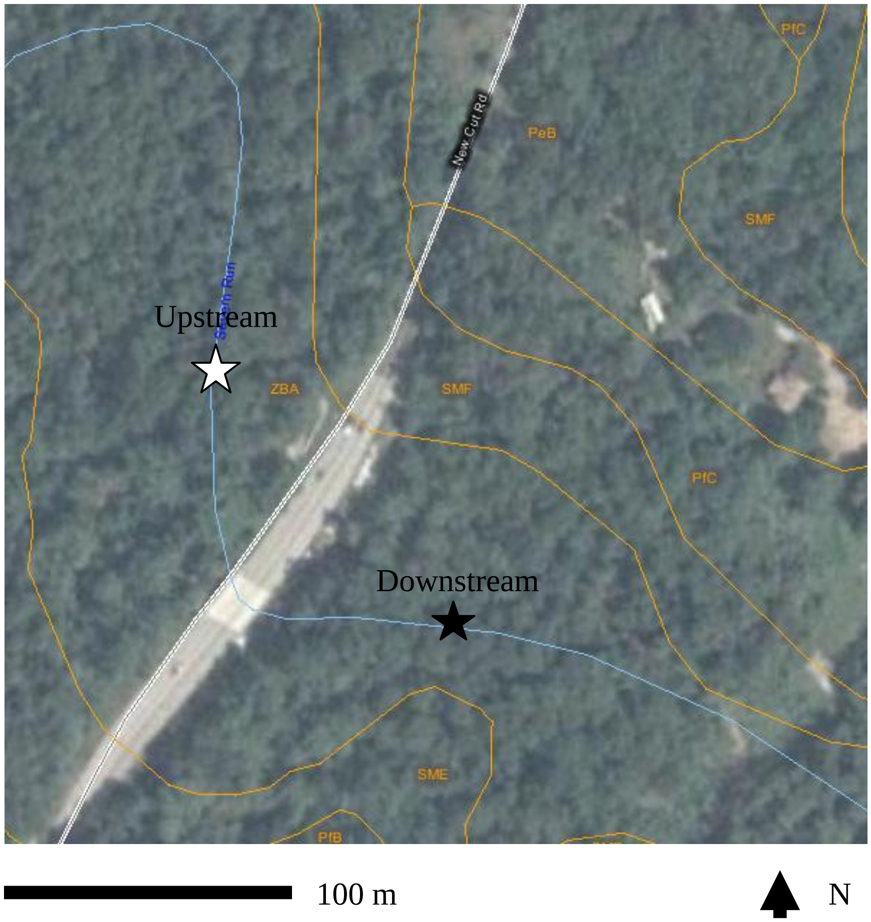


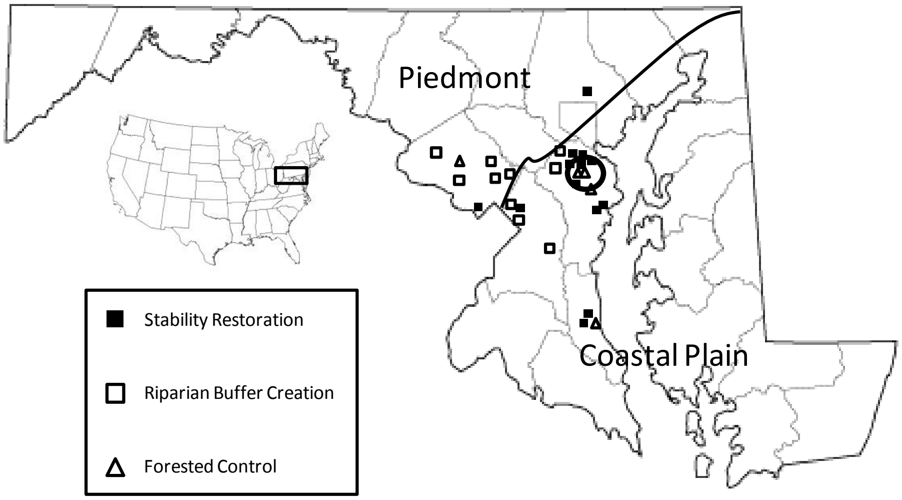


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Severn Run Tributary 2 (F)	Sassafras and Croom	Upland	Sandy loam	Well drained

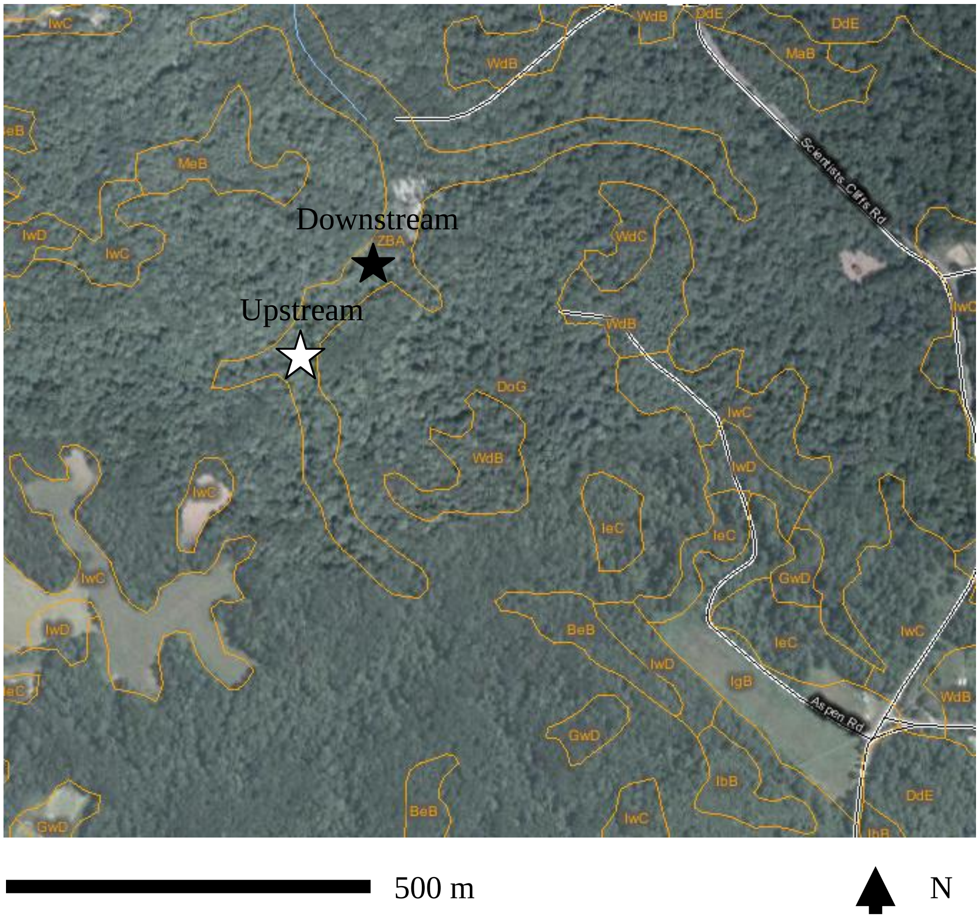


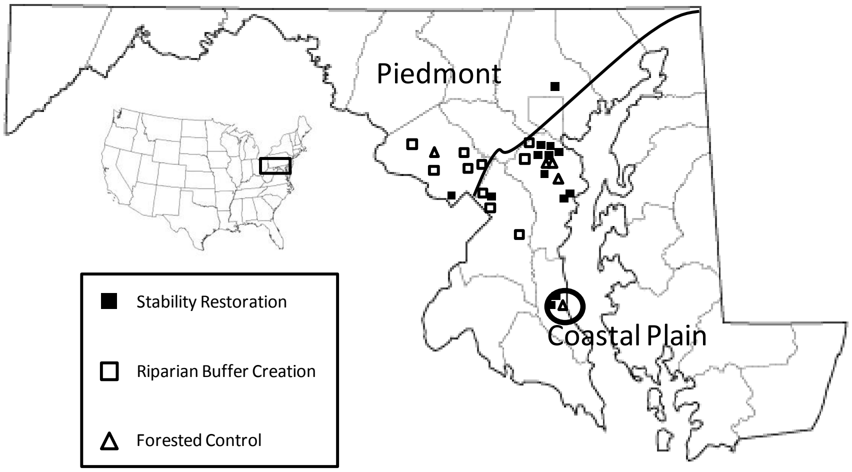
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Severn Run Mainstem (F)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained



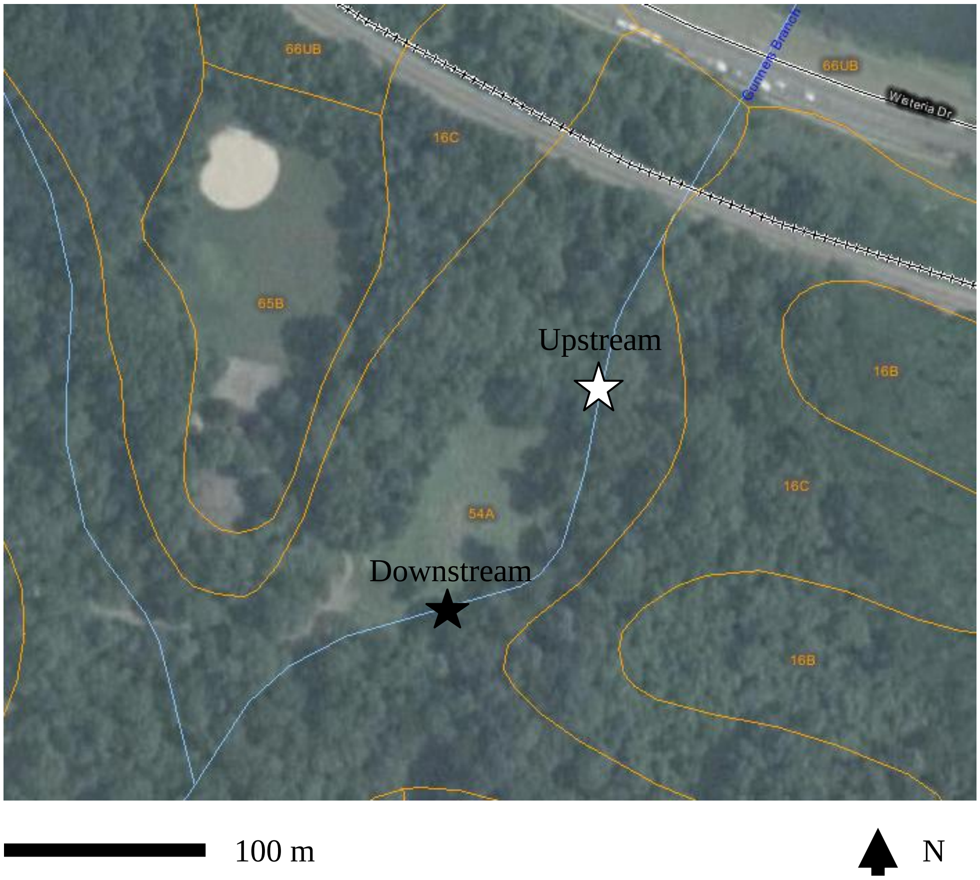


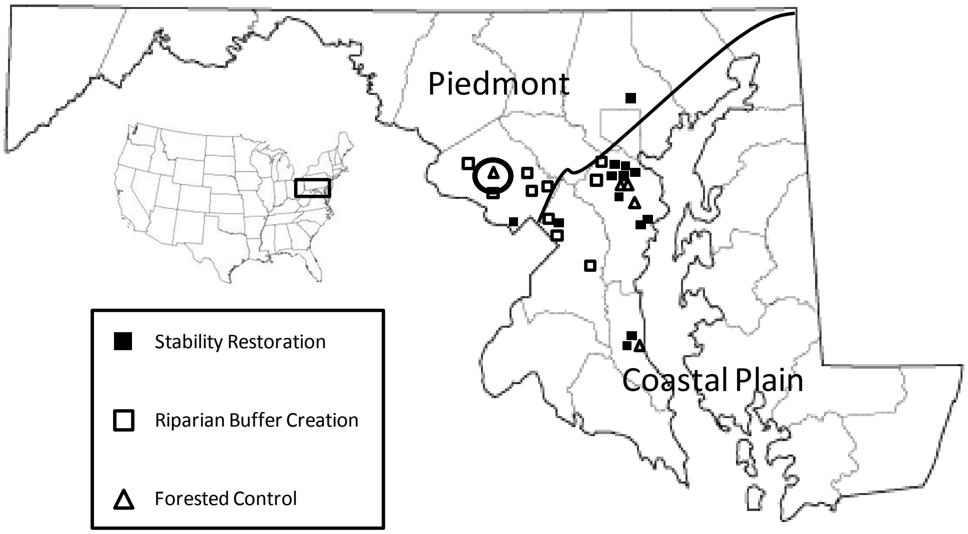
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Parkers Creek Tributary (F)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained



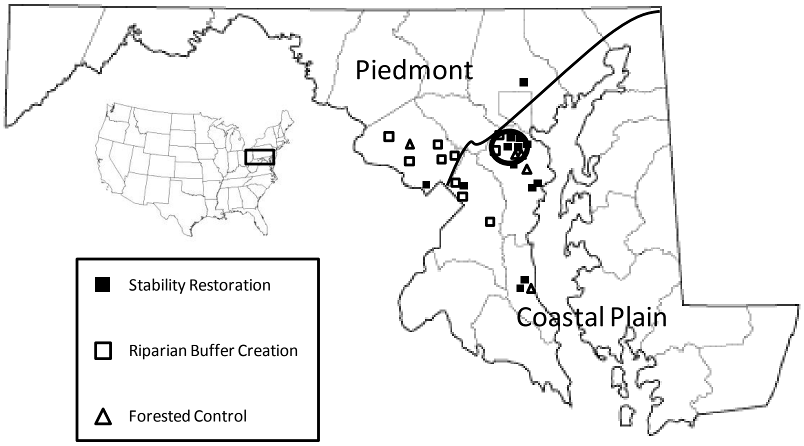


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Gunners Branch (F)	Hatboro	Floodplain	Silt loam	Poorly drained

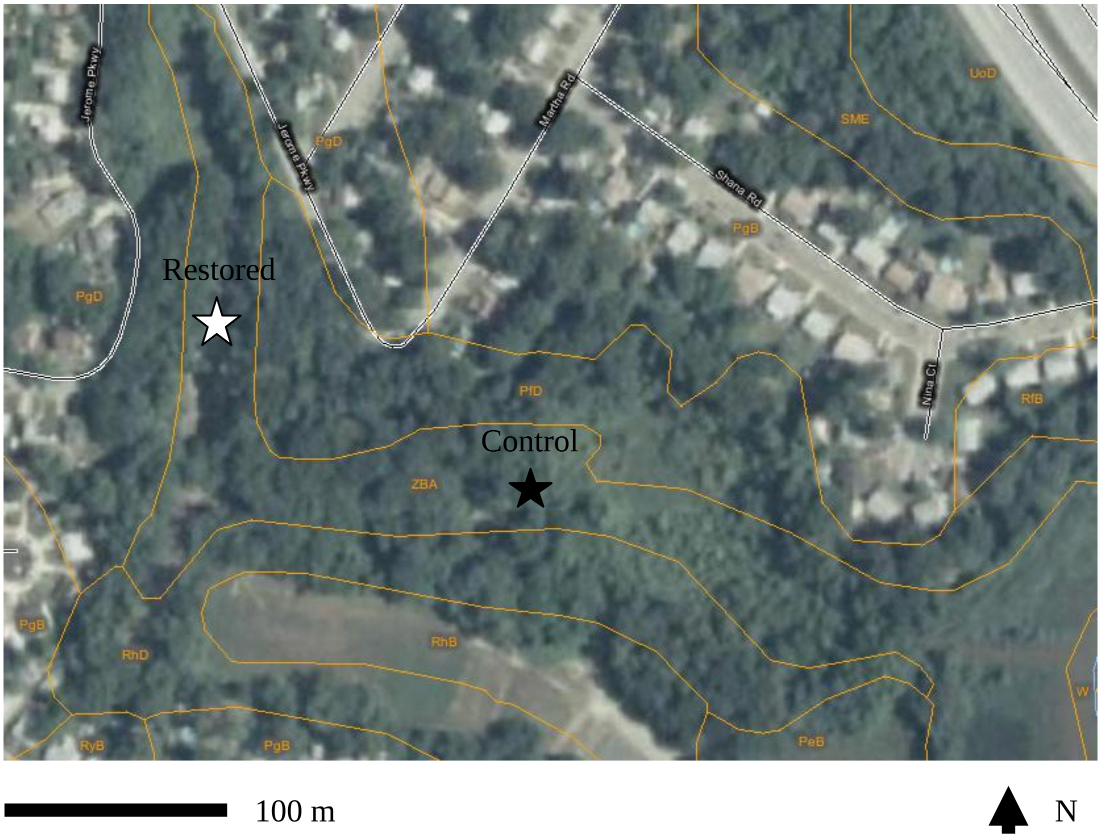


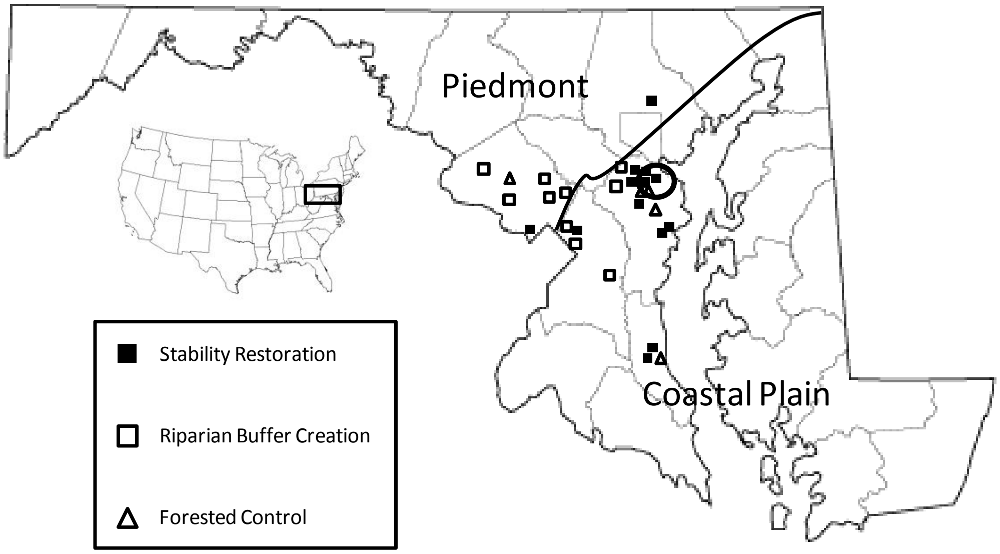


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Elvaton Towne Centre (S)	Fallsington/ Zekiah and Issues	Floodplain	Silt loam/ Sandy loam	Poorly drained

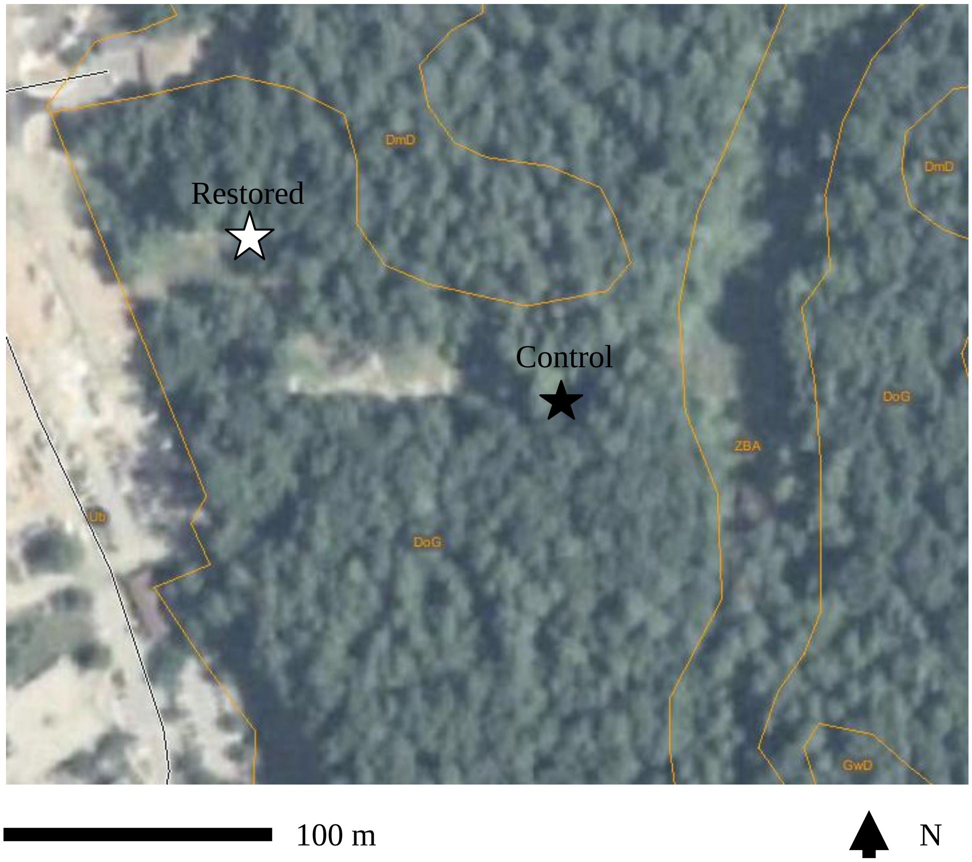


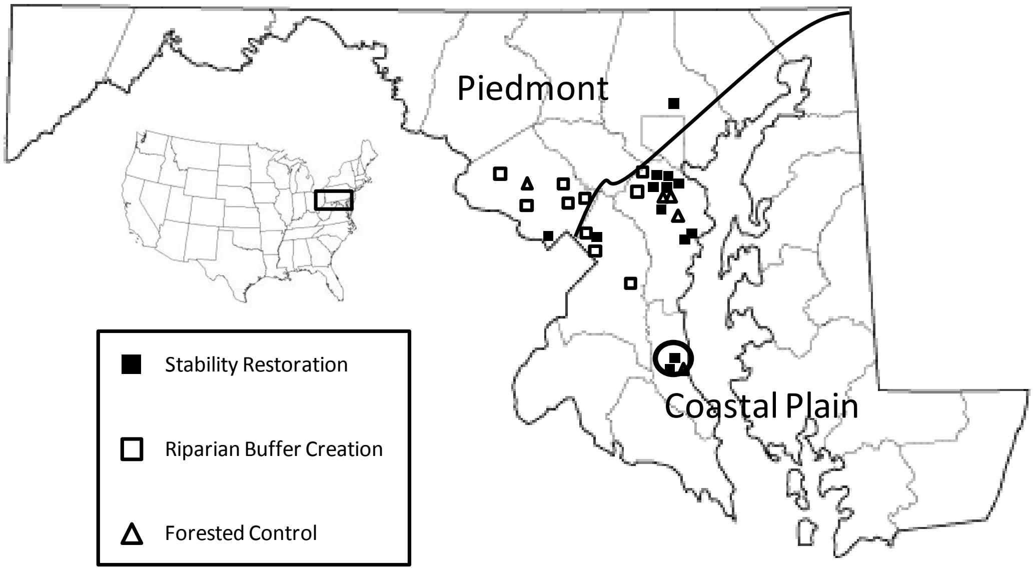
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Harundale Town Center (S)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained



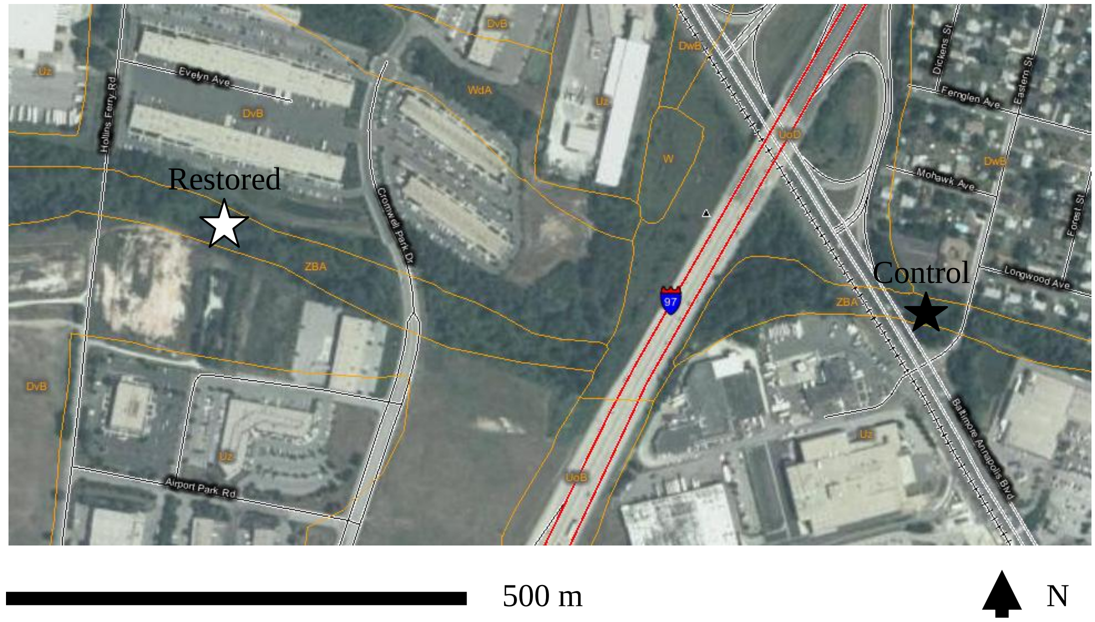


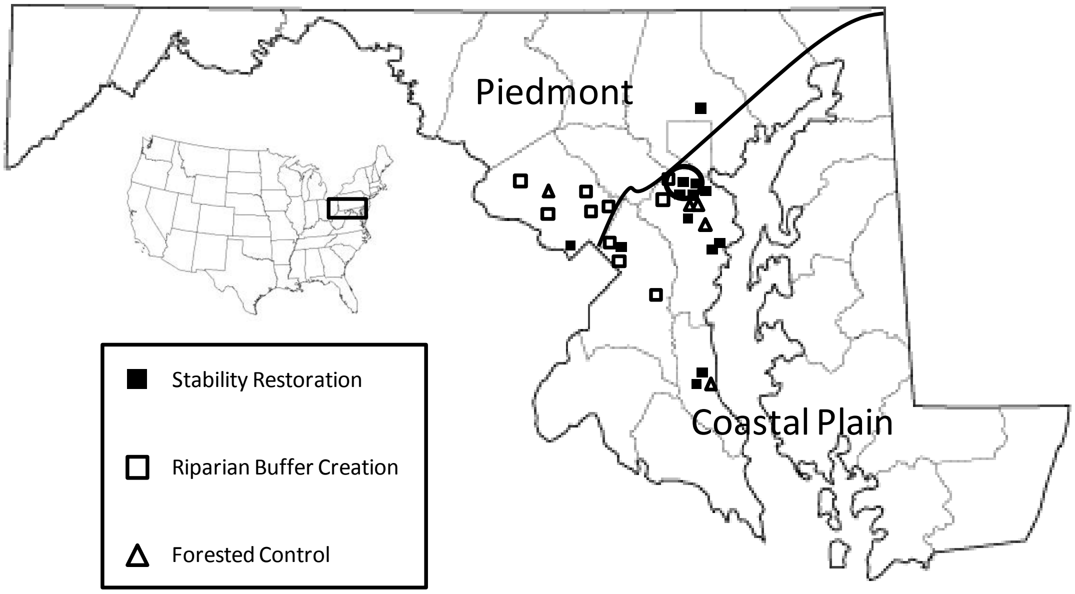
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Hunting Creek (S)	Downer Dodon Complex	Upland	Loamy sand	Well drained



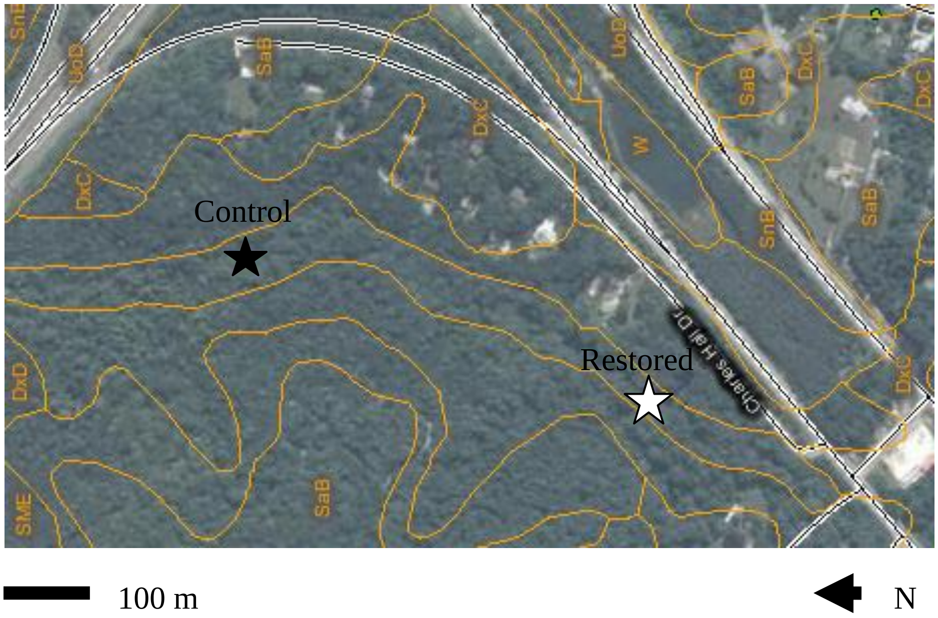


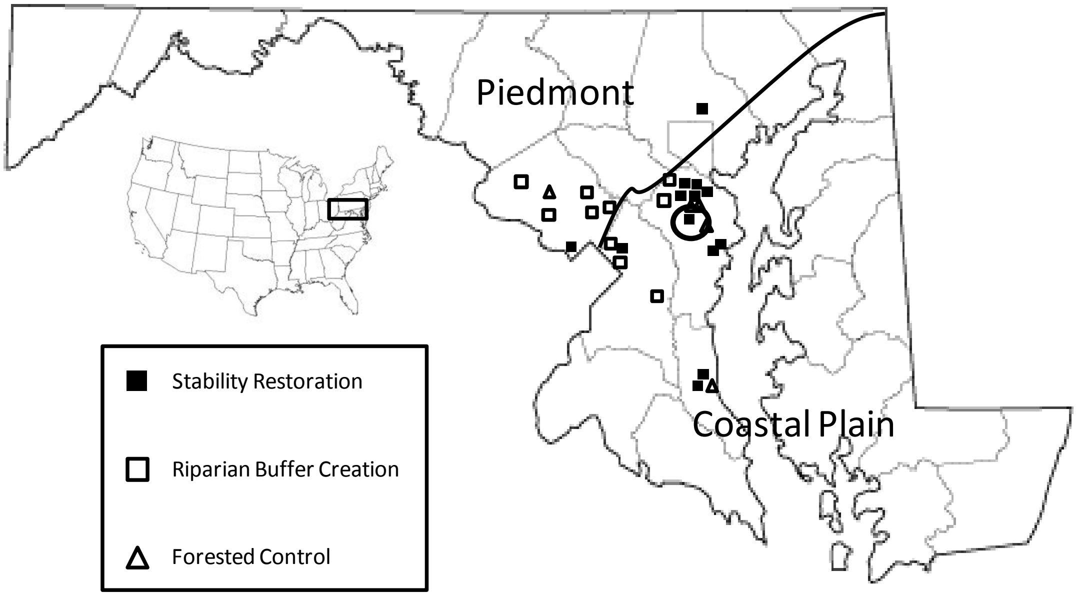
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Muddy Bridge Branch (S)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained



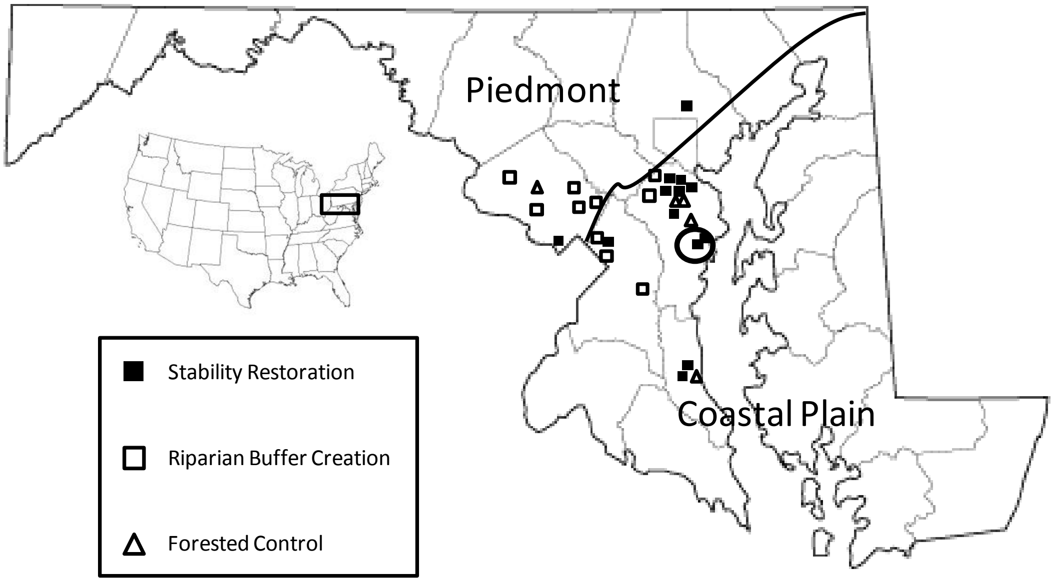


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
South Fork Jabez Branch (S)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained

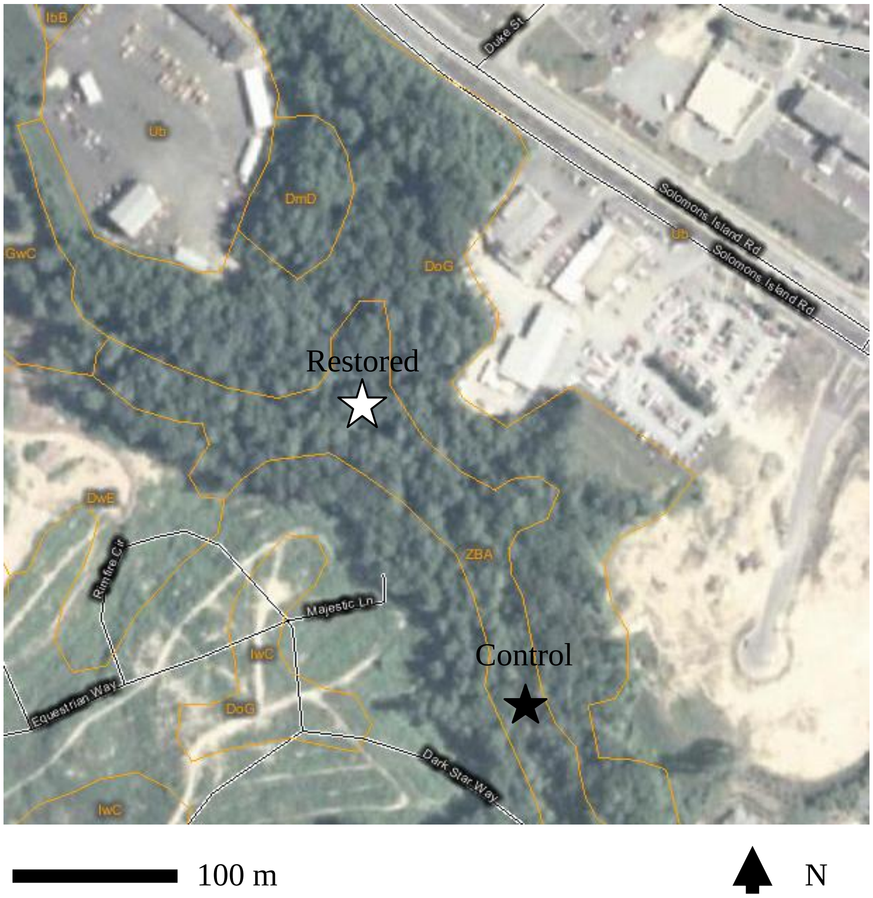


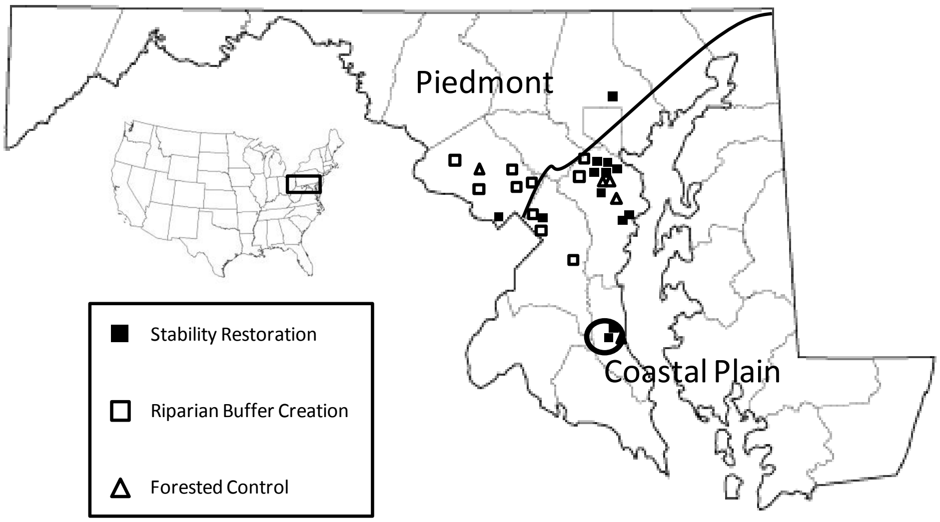


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Spa Creek at Cherry Grove Ave. (S)	Widewater and Issue	Floodplain	Loam/ Silty loam/ Sandy loam	Somewhat poorly drained

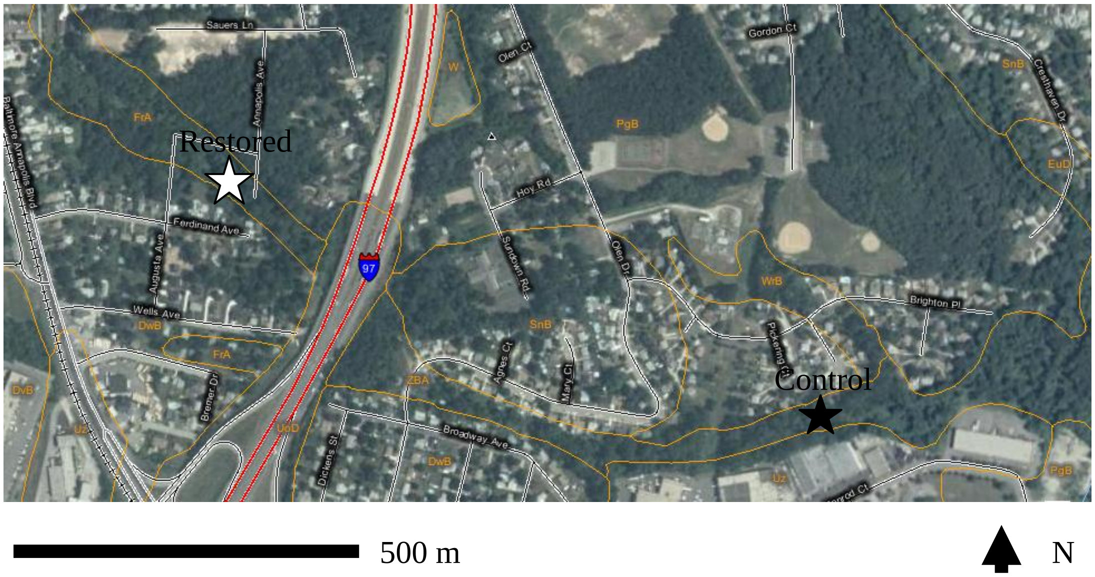


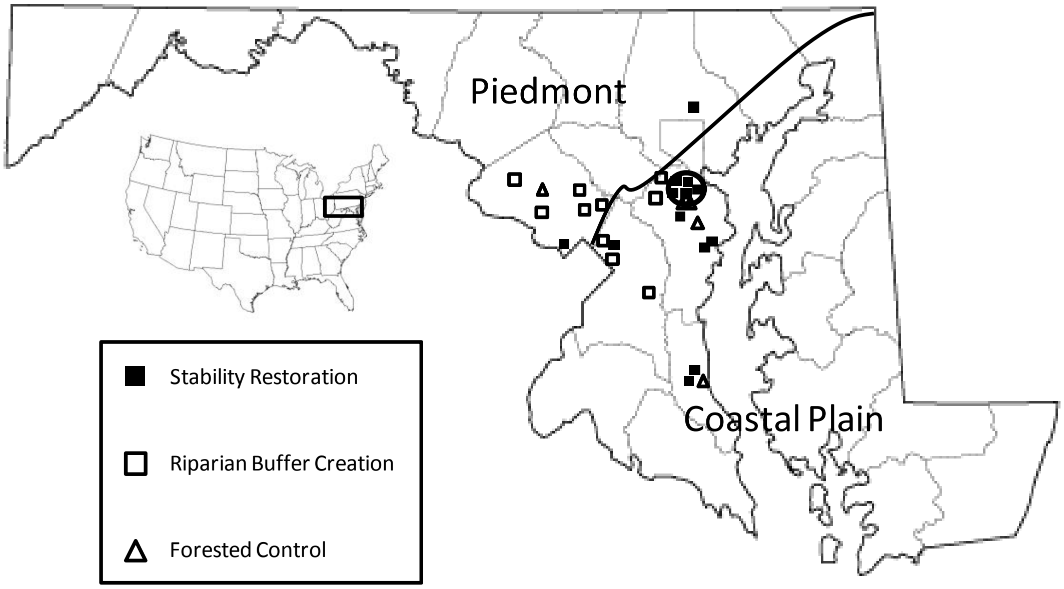
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Sullivans Branch (S)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained



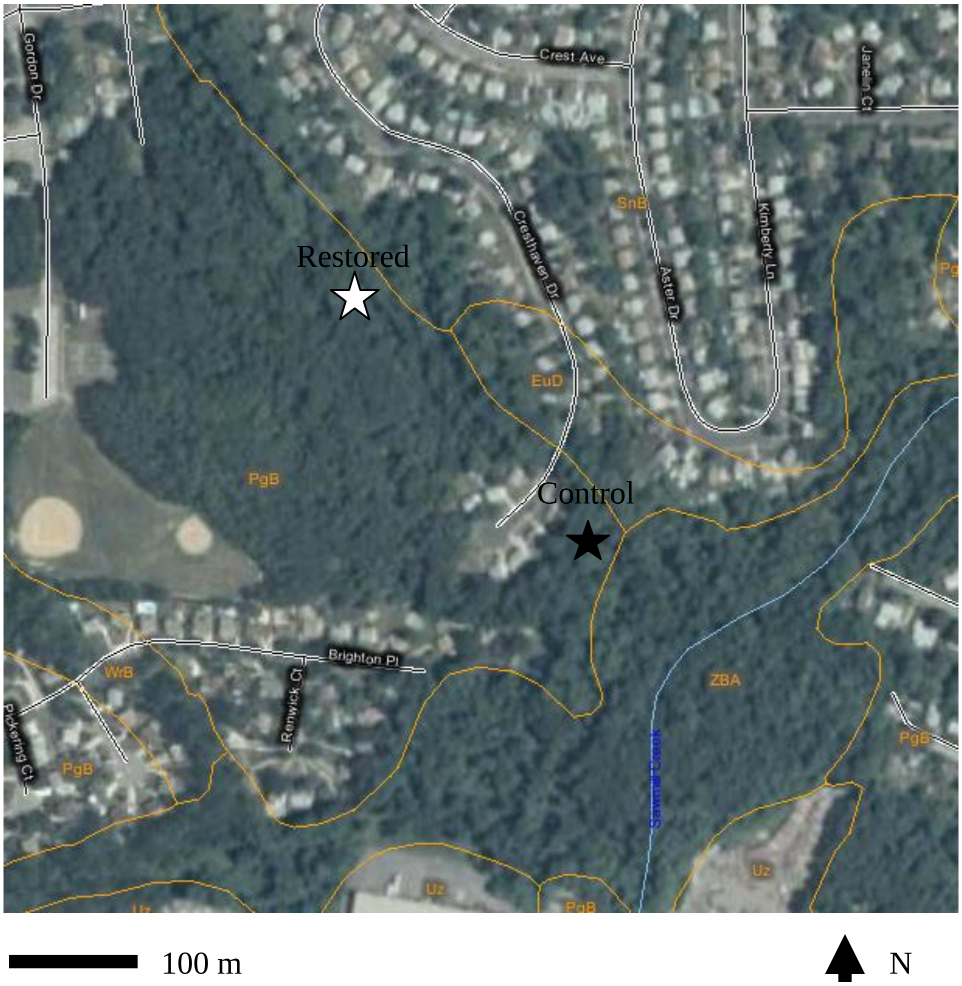


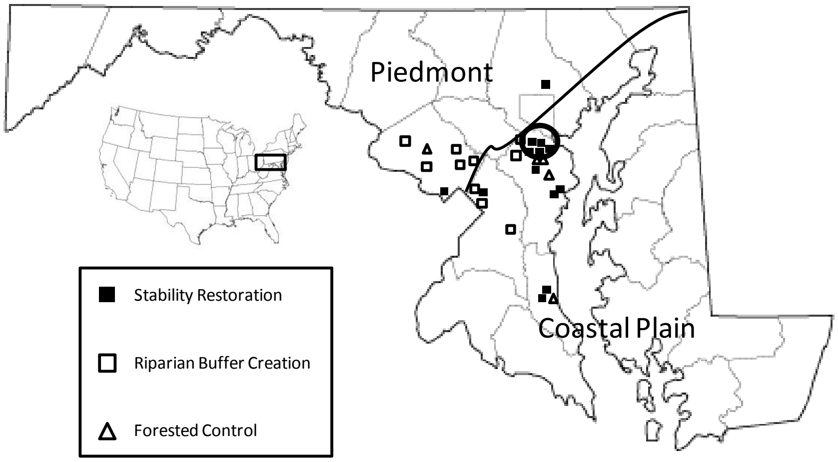
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Tributary 9 of Sawmill Creek (S)	Fallsington/ Zekiah and Issue	Floodplain	Silt loam/ Sandy loam	Poorly drained



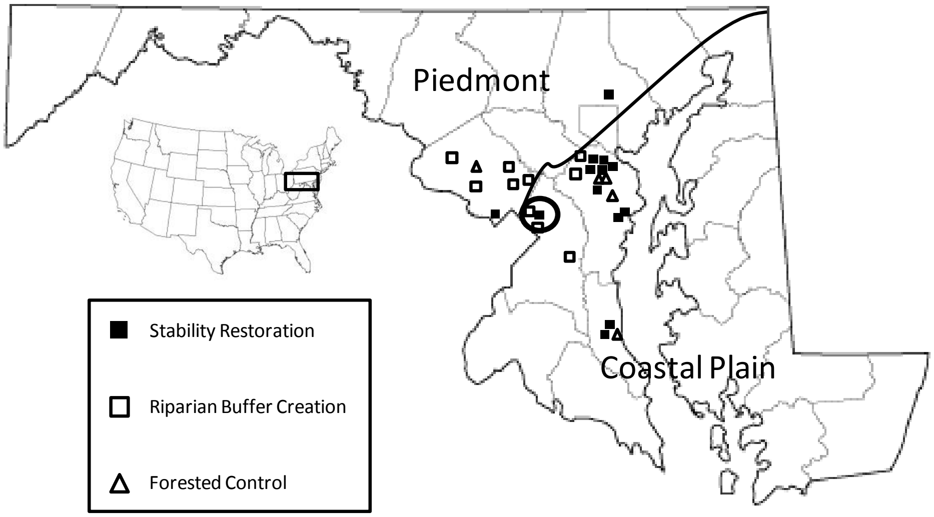


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Tributary 10 of Sawmill Creek (S)	Patapsco-Fort Mott	Upland	Sand/ Sandy loam	Somewhat excessively drained

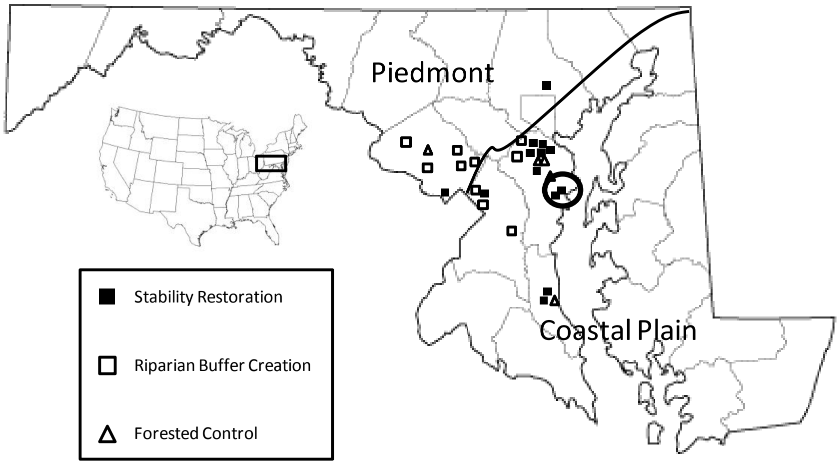




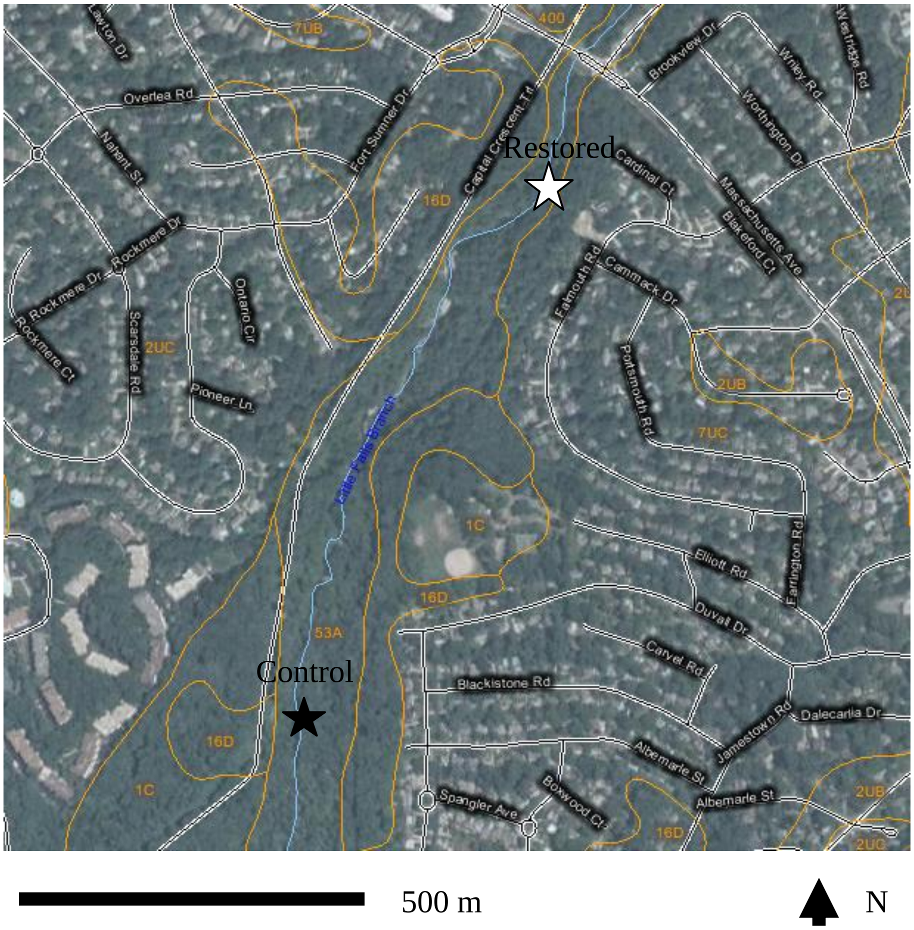
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
University Park Stream (S)	Zekiah	Floodplain	Silty loam	Poorly drained

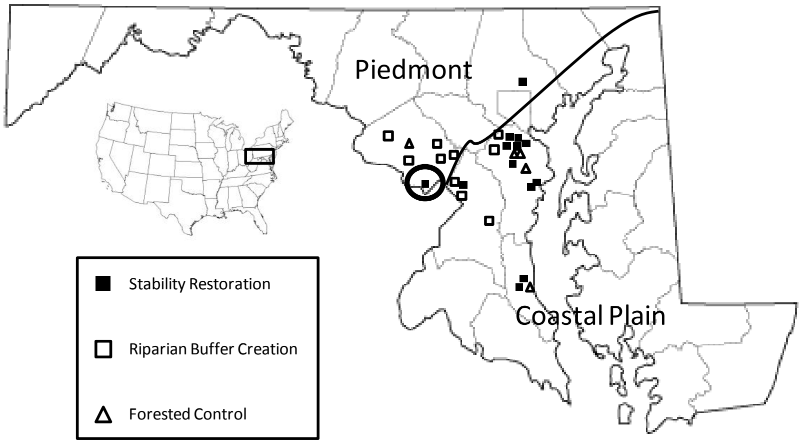


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Weems Creek at Bristol Dr. (S)	Annapolis	Upland	Sandy loam	Well drained

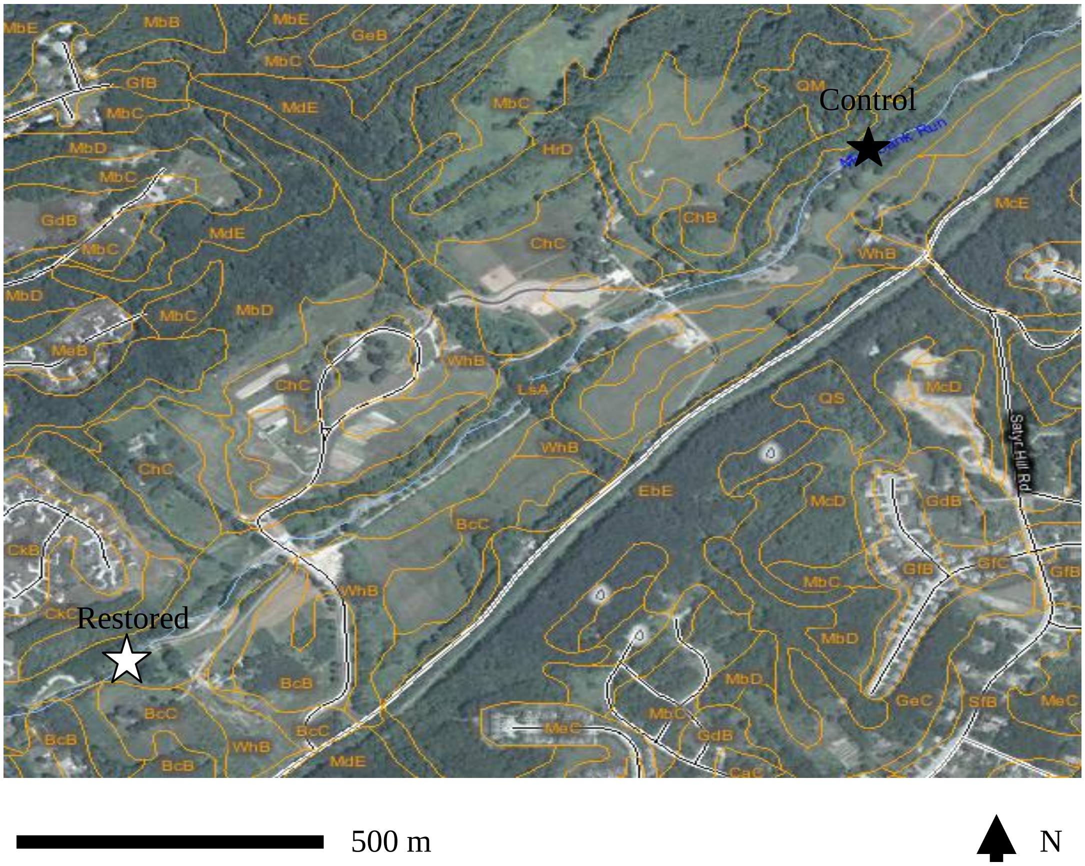


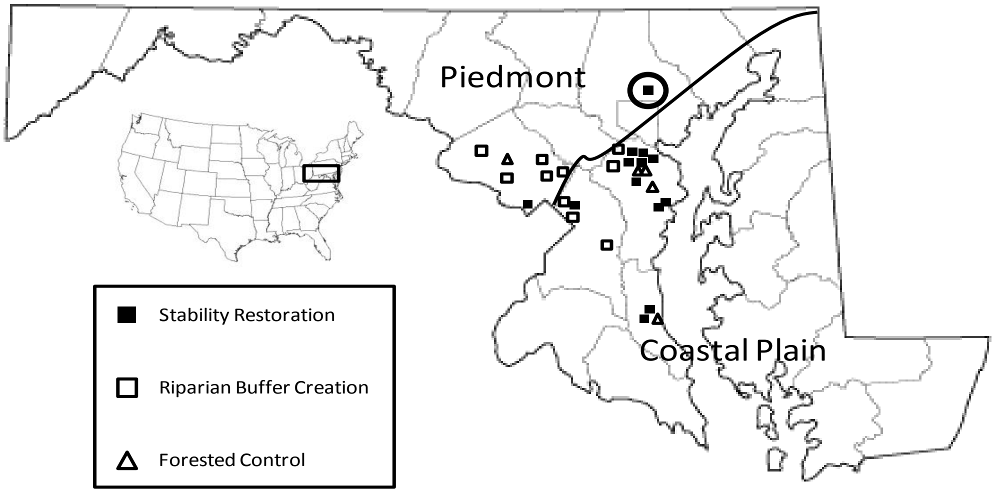
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Little Falls Branch (S)	Codorus	Floodplain	Silt loam	Moderately well drained



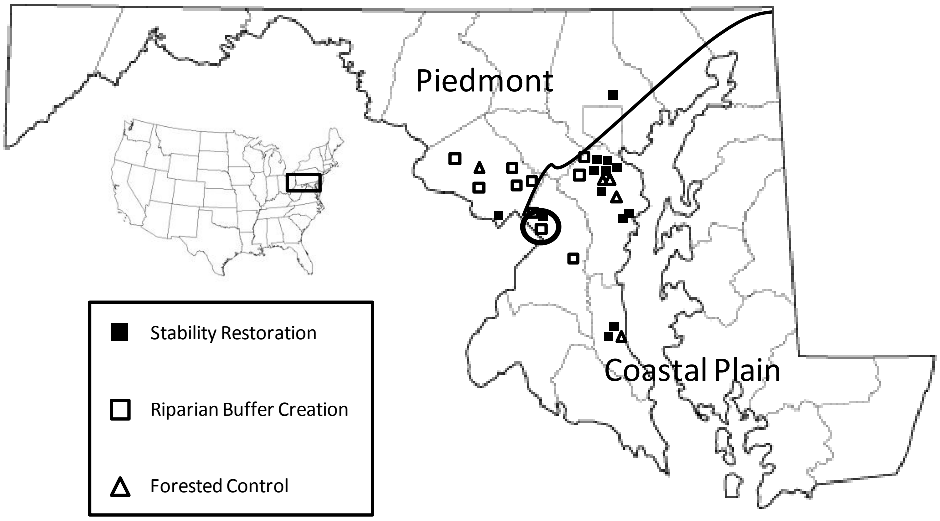


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Minebank Run (S)	Lindside	Floodplain	Silt loam	Moderately well drained

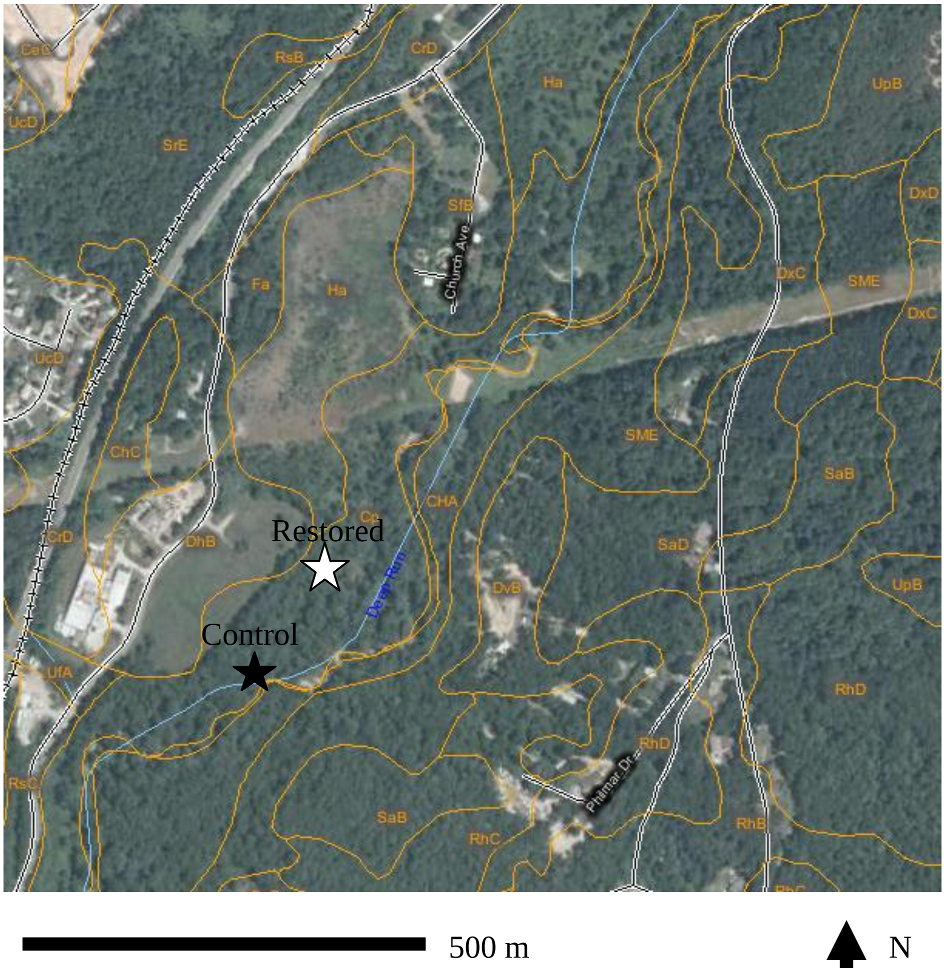


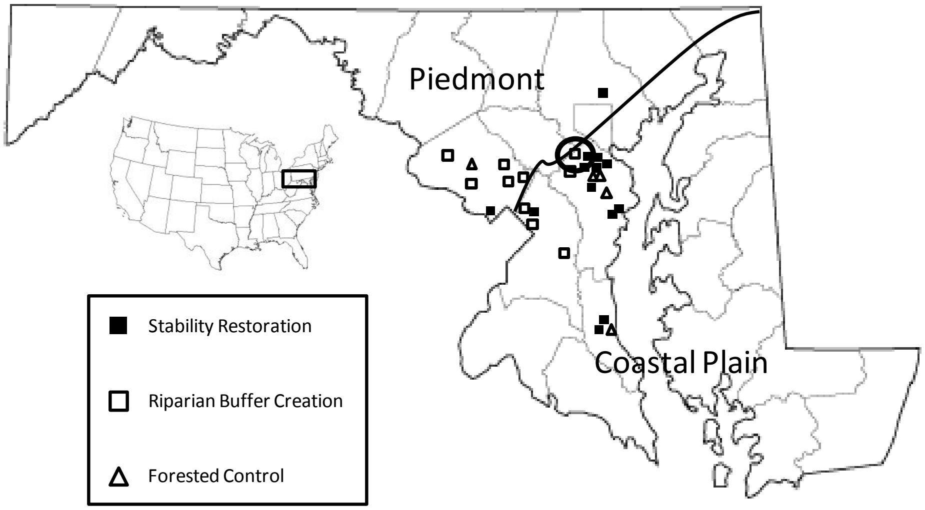


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Anacostia River (R)	Zekiah	Floodplain	Silty loam	Poorly drained

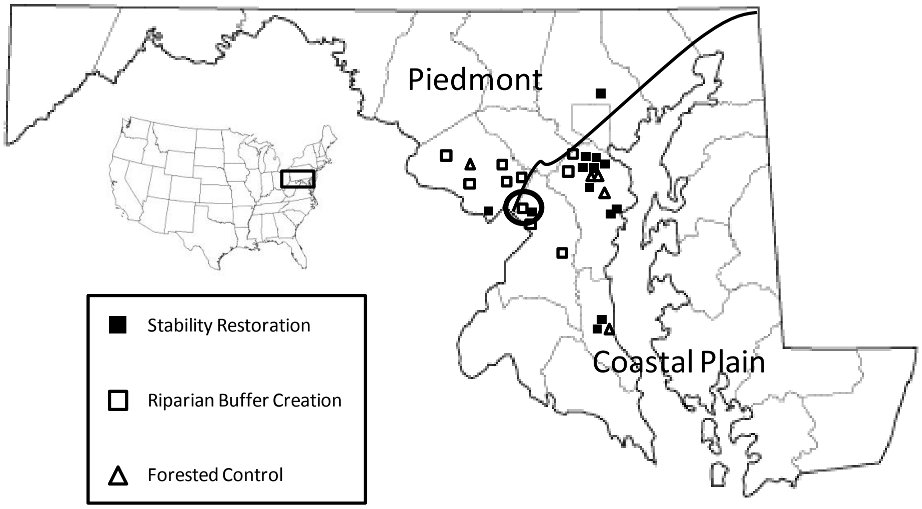


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Bollack Farm Stream (R)	Codorus and Hatboro	Floodplain	Loam	Moderately well drained

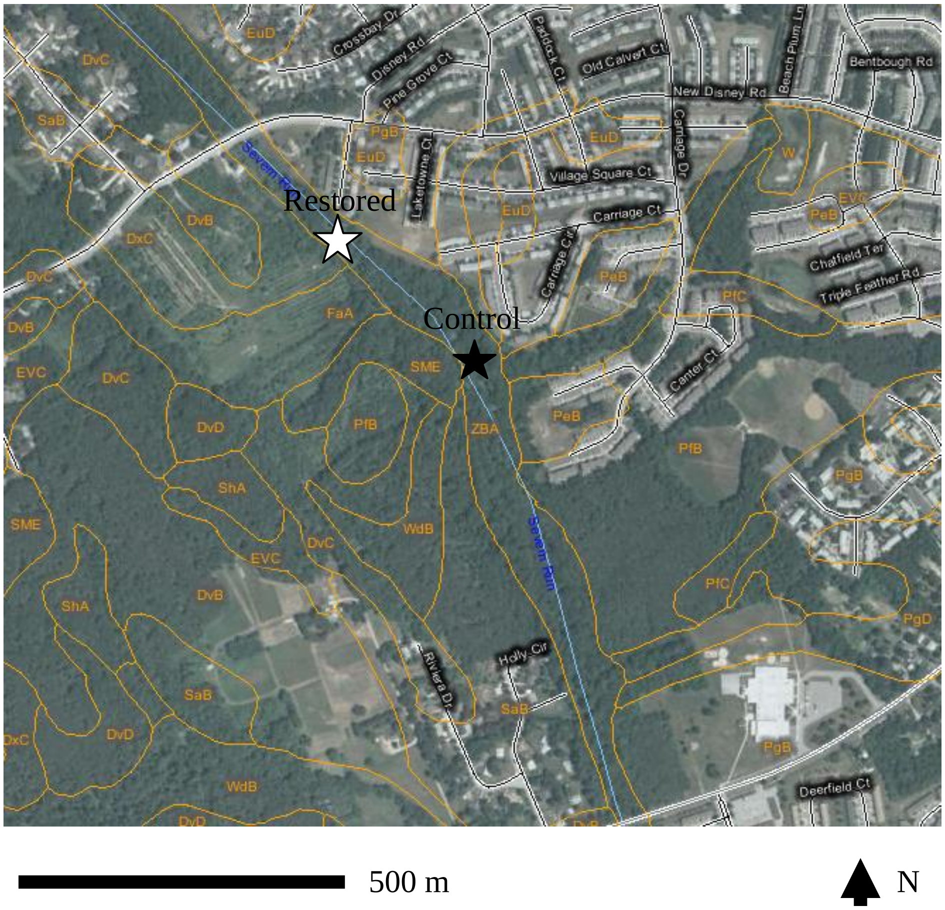


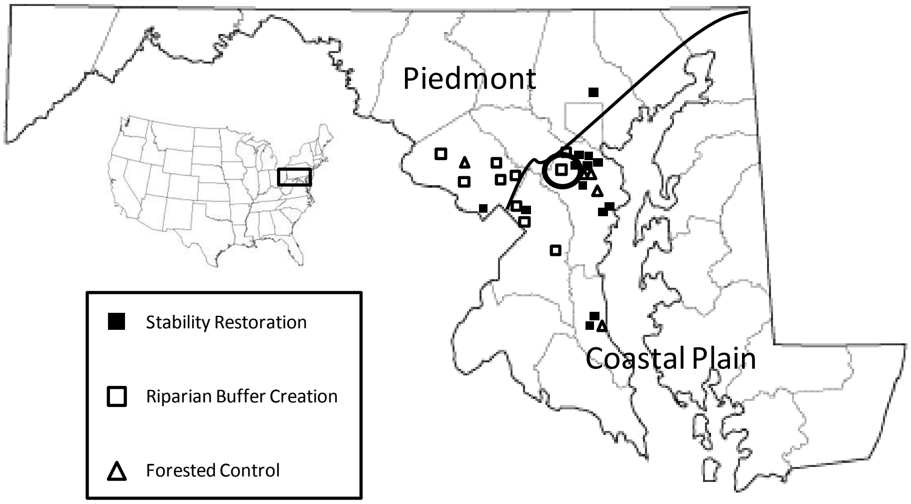


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Northwest Branch Lower (R)	Codorus and Hatboro	Floodplain	Loam	Moderately well drained

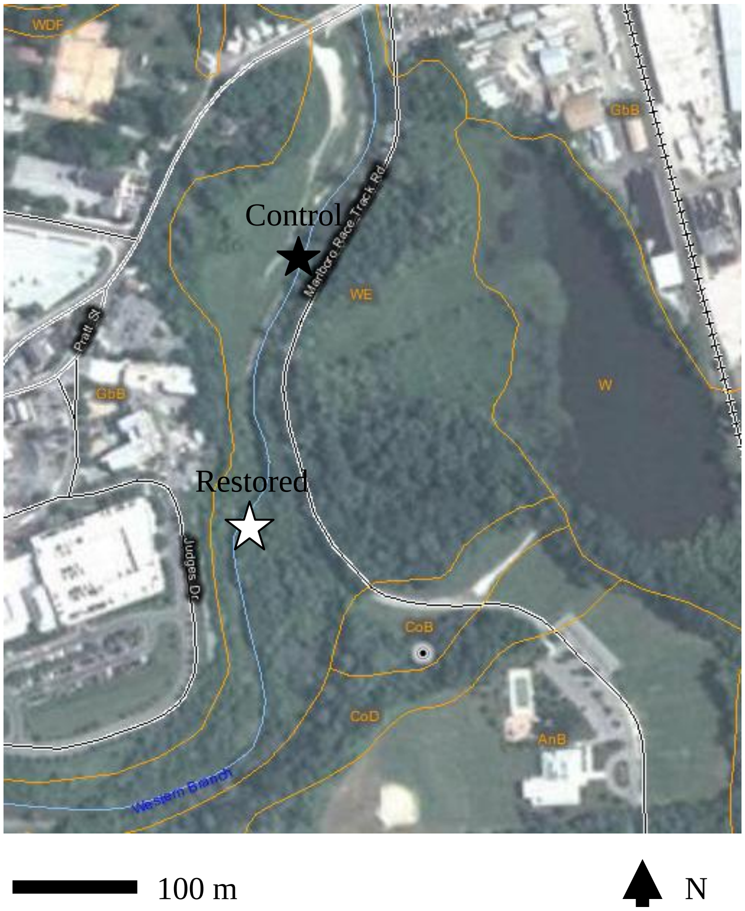


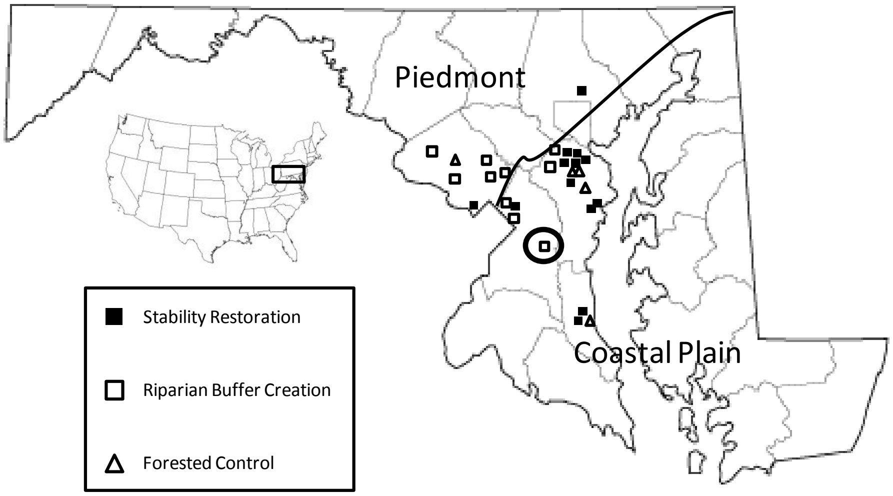
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Severn Run Mainstem (R)	Zekiah and Issue	Floodplain	Silt loam	Poorly drained



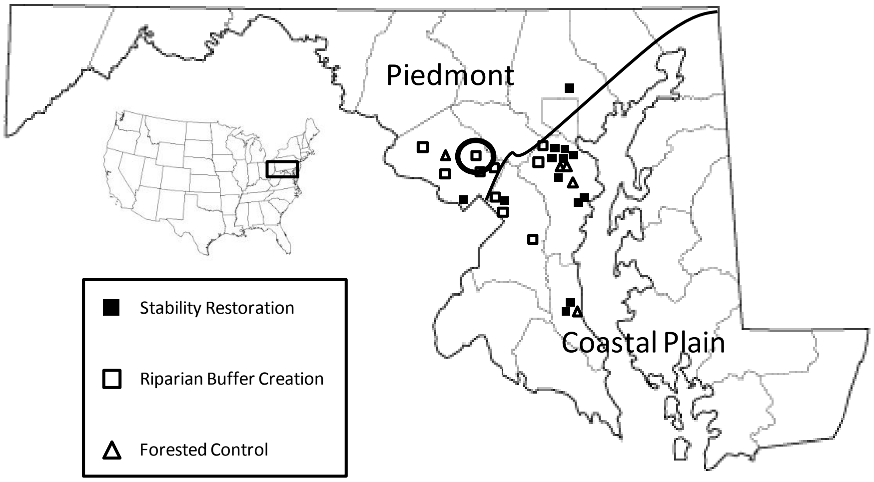


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Western Branch Patuxent River (R)	Widewater and Issue	Floodplain	Loam/ Silty loam/ Sandy loam	Somewhat poorly drained

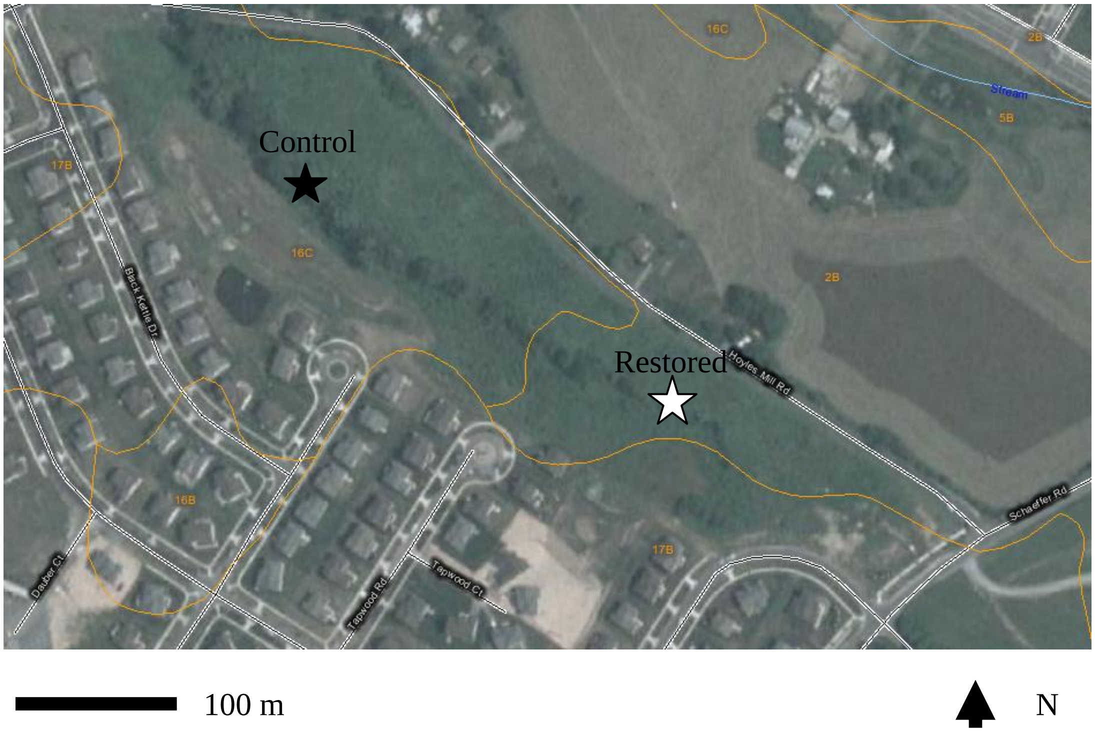


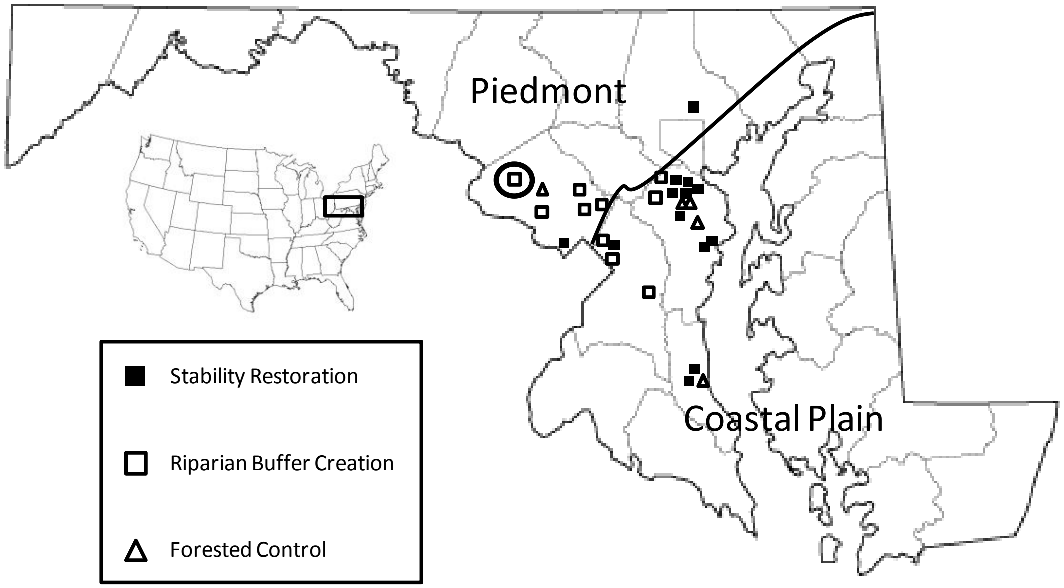


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Hawlings R. Trib. at St. Albert Dr. (R)	Baile/Hatboro	Upland/ Floodplains	Silt loam	Poorly drained

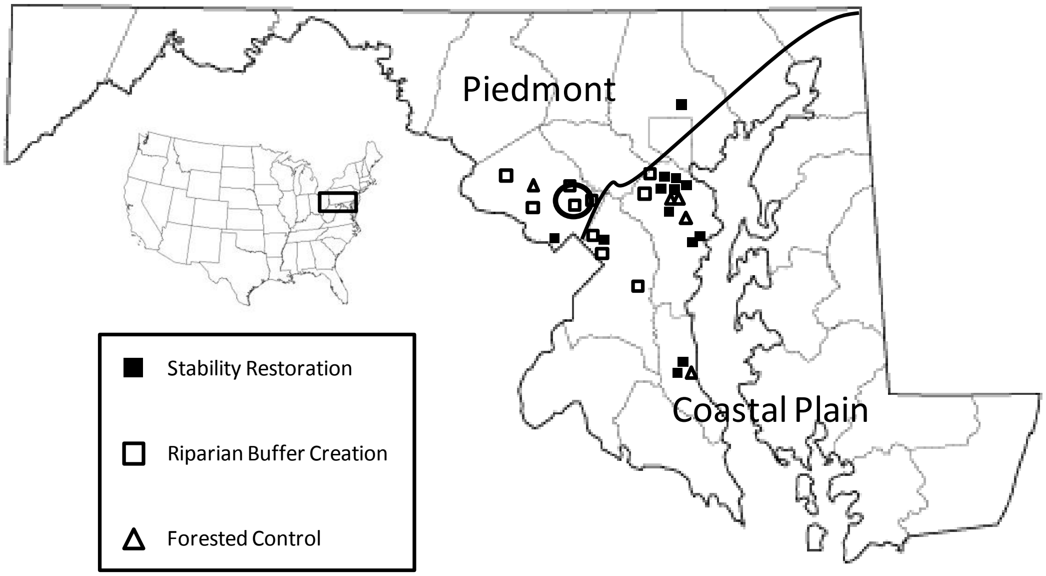


Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Leaman Farm Stream (R)	Glenelg/ Brinklow-Blocktown	Upland	Silt loam	Well drained

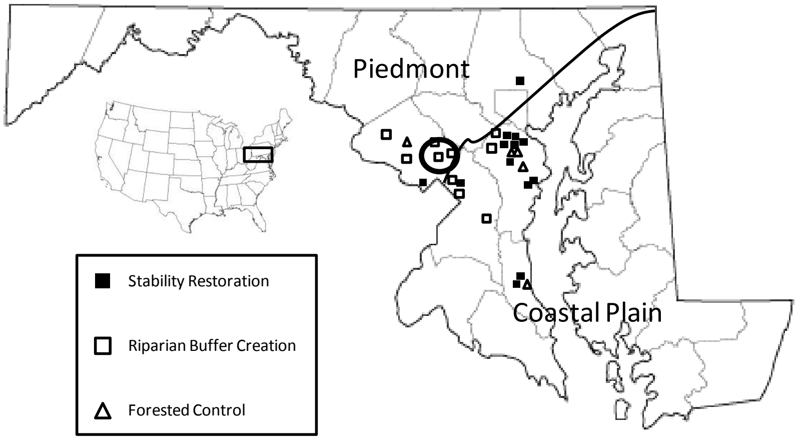




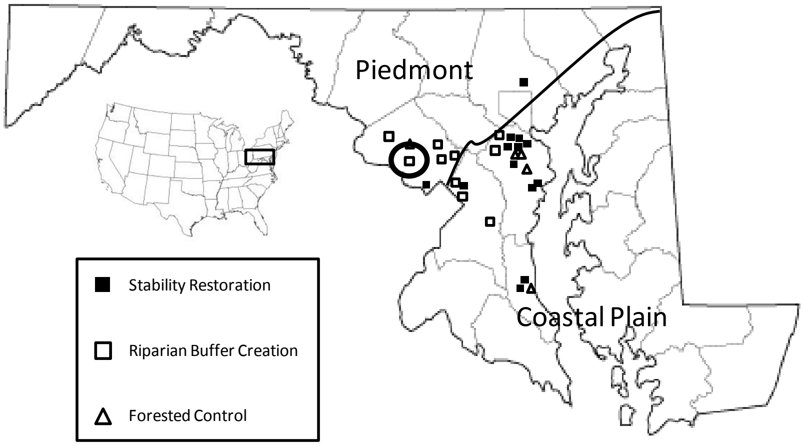
Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Northeast Branch Headwater (R)	Baile	Upland	Silt loam	Poorly drained



Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Northwest Branch Upper (R)	Hatboro	Floodplain	Silt loam	Poorly drained



Site Name	Soil Series	Upland or Floodplain	Soil Texture	Drainage Class
Watts Branch Tributary at Valley Dr. (R)	Baile/Hatboro	Upland/ Floodplain	Silt loam	Poorly drained



Appendix III – Water Chemistry Data at Diatom Sampling Sites

Stream	pH	Cond. (μS/cm)	NH_4^+ (mg N/L)	PO_4^{3-} (mg P/L)	NO_2^-+ NO_3^- (mg N/L)	TDP (mg P/L)	TDN (mg N/L)	Si (mg/L)	Cl^- (mg/L)	SO_4^{2-} (mg/L)	DOC (mg C/L)
Severn Run Trib. 1 (F)	5.66	36	0.004	0.0021	0.194	0.0039	0.41	0.12	7.04	6.67	2.43
Severn Run Trib. 2 (F)	5.77	73	0.004	0.0023	0.150	0.0035	0.39	0.10	16.86	11.21	5.26
S Fk. Jabez Br. (U)	6.12	82	0.004	0.0021	1.740	0.0045	1.92	0.91	29.62	5.33	1.24
Harbor Cent. E (U)	6.80	345	0.065	0.0026	0.867	0.0082	1.26	3.01	106.50	7.32	3.24
Harbor Cent W (U)	6.83	374	0.042	0.0034	0.379	0.0133	0.67	3.71	117.71	3.43	2.74
Spa Trib. at Htp Ln (U)	7.11	190	0.025	0.0044	0.215	0.0334	0.54	6.78	53.76	8.26	4.51
Spa Creek (R)	7.06	247	0.007	0.0030	0.671	0.0073	0.68	4.94	45.10	17.05	2.28
Weems Cr. Bristol (R)	7.06	226	6.480	0.0528	1.030	0.3427	13.82	3.12	46.64	13.13	24.11
Cowhide Branch (R)	6.46	395	0.048	0.0047	0.731	0.0037	0.60	2.44	183.16	6.07	1.36

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