ABSTRACT

Title of Dissertation: EFFECTS OF URBANIZATION AND INFILTRATION-BASED WATERSHED RESTORATION ON THE HYDRO-ECOLOGY OF HEADWATER STREAMS

Rosemary Margaret Fanelli, Doctor of Philosophy, 2016

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Urbanization profoundly alters the hydrologic routing of a landscape resulting in the degradation of downstream aquatic ecosystems. To mitigate these effects, watershed managers implement infiltration-based storm water control measures (SCMs), designed to convert stormwater runoff into groundwater recharge. However, the ability of infiltration-based SCMs to restore hydrological processes, and to reverse damage to the downstream ecosystem, remains poorly understood**.** To address this research gap, I examined the hydro-ecological effects of urbanization and SCM implementation in 11 headwater watersheds spanning an urbanization-restoration gradient (4 forest, 4 urban-degraded, and 3 urban watersheds restored with SCMs) near Annapolis, Maryland, USA. Regenerative stormwater conveyances (RSCs) were the type of SCM examined in the study. I used high-frequency precipitation, stream stage, and baseflow discharge collected at the watershed outlets to develop metrics characterizing watershed storage and stream responses to precipitation. I then conducted water quality sampling, temperature monitoring, and quantified aquatic insect community composition in the downstream

ecosystems. Finally, I employed high-frequency groundwater monitoring in one of the SCMs to identify potential mechanisms controlling their hydrological function.

The hydrological effects of urbanization were clearly observed across the study watersheds, but only one of the three restored watersheds modulated hydrology (e.g., a larger minimum runoff threshold relative to the other urban watersheds). However, baseflow in this stream was low compared to the forested streams, suggesting that enhanced infiltration of stormwater runoff did not recharge storage zones that support stream baseflow. Aquatic insect diversity and the percentage of sensitive taxa declined with increasing urbanization, with no significant effect of restoration. Water quality remained poor in both urban-degraded and urban-restored streams, with higher conductivity values, lower dissolved oxygen, and warmer stream temperatures than in forested streams. These water quality issues likely hampered recovery of sensitive taxa in downstream ecosystems. Groundwater monitoring in one of the SCMs indicated that high runoff delivery rates from the watershed limited infiltration within the SCM, which allowed the conveyance of untreated runoff to the downstream channel. The centralized design of RSCs, and their placement in areas of topographic convergence above channel heads, likely limits their effectiveness for restoring hydrological processes to urban watersheds.

EFFECTS OF URBANIZATION AND INFILTRATION-BASED WATERSHED RESTORATION ON THE HYDRO-ECOLOGY OF HEADWATER STREAMS

By

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Preface

This dissertation consists of an introduction, three research chapters, and a summary and conclusions section. All research chapters are presented in manuscript form with introduction, methods, results, discussion, conclusions, and any supplemental information. Tables, figures, and captions occur at the conclusion of each chapter. A single literature cited section occurs at the end for references made throughout the dissertation.

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Chapter 2: Downstream changes in water quality and insect community composition from urbanization and infiltration-based watershed restoration

Introduction

Headwater streams not only provide important ecosystem services locally, they also collectively contribute to the "functional integrity" of the downstream fluvial network (Nadeau and Rains 2007). Low-order streams process nutrients and organic matter, and influence overall downstream water quality (Alexander and others 2007; Mulholland and others 2008; Peterson and others 2001) due to their large wetted perimeter and prevalence across the landscape (Leopold and others 1964). Moreover, they provide a rich variety of aquatic habitats required for supporting diverse biological communities throughout the stream network (Freeman and others 2007; Meyer and others 2007). This growing body of research demonstrating the importance of headwater streams recently led to a new federal ruling which expands their protections to under the Clean Water Act (EPA 2014).

Headwater streams ecosystems are shaped by their stream flow regime, which is defined as the frequency, magnitude, duration, rate-of-change, and timing patterns of stream discharge (Bunn and Arthington 2002; Poff and Allan 1995; Poff and others 1997). The flow regime in a stream channel is controlled by physical and biological attributes of its watershed, including underlying geology; overlying soil characteristics; topography; climate; vegetation; and lastly, land use (Black 1997; Poff and others 1997). These watershed characteristics affect how precipitation is routed through the landscape and the way it eventually leaves the watershed (e.g., surface runoff, groundwater discharge to streams, or evapotranspiration), which ultimately controls patterns in streamflow in the downstream channel (Wagener and others 2007).

Urbanization profoundly changes flow regimes in headwater streams (Brown and others 2009; Rose and Peters 2001; Walsh and others 2005b). Impervious surfaces and soil compaction from urbanization reduce infiltration rates, even if the land remains vegetated (Gregory and others 2006; Olson and others 2013). The increased surface runoff generated is often conveyed directly into headwater stream channels through storm sewer systems, which causes channel incision (Booth 1990), bank erosion, and a loss of geomorphic complexity in headwater streams (Arnold and others 1982). Many urban headwater streams have even undergone burial during urbanization (Elmore and Kaushal 2008). Moreover, urban contaminants, such as heat, chloride, trace heavy metals, and hydrocarbons, are conveyed directly into stream ecosystems (Paul and Meyer 2001). These collective changes to the stream environment from urbanization have resulted in severe losses in stream biodiversity (Cuffney and others 2010; King and others 2011).

In an effort to mitigate these impacts and recover stream biodiversity, many practitioners have employed stream ecosystem restoration (Bernhardt and others 2005). In theory, ecological restoration of fluvial systems should work on multiple scales, simultaneously addressing local-scale limitations (e.g., habitat quality), watershed-scale stressors (e.g., water quality, altered flow patterns), as well as regional limitations for dispersal (Lake and others 2007; Palmer and others 1997). In practice, however, stream restoration is often predominantly focused on restoring the geomorphic pattern of a degraded channel (e.g., natural channel design; (Shields and others 2003). Responses by aquatic invertebrate communities to these types of stream restorations have been anemic (Ernst and others 2012; Palmer and others 2010; Sundermann and others 2011). Reasons for their failure have been attributed to dispersal constraints from regional pools (Brederveld and others 2011; Parkyn and Smith 2011), but also because they do not target diffuse, watershed-scale sources of degradation with their design (Bernhardt and Palmer 2011). For example, Larson and others (2001) hypothesized that poor invertebrate recovery in urban streams with large woody debris additions was due to continued high flows and sediment loads.

Ecologists have since called for watershed-scale restoration efforts addressing these diffuse stressors to recover aquatic biodiversity in degraded urban streams (Bernhardt and Palmer 2011; Walsh and others 2005a; Walsh and others 2009), including improved stormwater management. These strategies aim to enhance the infiltration of stormwater runoff, which reduces the amount of connected (or effective) impervious areas in the watershed (Walsh and others 2005a). Indeed, a variety of stormwater control measures (SCMs) designed to infiltrate stormwater (e.g., infiltration trenches or bioretention basins) or remove stormwater via evapotranspiration (green roofs) have been successful at reducing stormwater runoff and can provide other benefits to the stream ecosystem including thermal and chemical pollutant retention (Davis and others 2009; Dietz and Clausen 2006; Hunt and others 2012). How infiltration-based SCMs affect the downstream flow regime, water quality and, subsequently, the aquatic community remains unclear.

In this dissertation, I examine the hydro-ecological effects of urbanization and infiltration-based SCM implementation in the Annapolis region of Maryland, USA. My research focuses on regenerative stormwater conveyances (RSC), which are an emerging type of infiltration-based SCMs currently being employed in the Mid-Atlantic region. The objectives of this dissertation were to 1) understand how urbanization and SCM implementation affect the hydrologic behavior and flow regime of headwater streams; 2) quantify the downstream changes in water quality and insect biodiversity resulting from urbanization and SCM implementation; and 3) identify factors controlling the hydrological functioning of these infiltration-based SCMs. The following is a brief description of the three research chapters addressing these objectives:

For Chapter 1, I focus on documenting the differences in hydrological behavior across 11 headwater catchments that span an urbanization–restoration gradient (4 forested; 4 urban-degraded, 3 urban-restored) to evaluate changes in watershed hydrologic function from urbanization and watershed restoration implementation. Both discrete discharge and high-frequency, continuous stage-rainfall monitoring were conducted in each catchment to develop six hydrologic metrics describing changes in watershed storage, flowpath connectivity, or the resultant stream flow regime. The hydrological effects of urbanization were clearly observed in all metrics, but only one of the three restored catchments exhibited partially restored hydrologic function. In this catchment, enhanced infiltration of stormwater runoff within the restoration structure increased storage of small rainfall events. However, baseflow in the stream draining the restored catchment remained low compared to the forested reference streams. This suggests that enhanced infiltration of stormwater runoff did not recharge the subsurface storage zones contributing to stream baseflow.

For Chapter 2, I conducted monthly baseflow water quality sampling, high-frequency temperature monitoring, and quantified aquatic insect community composition across the same 11 headwater catchments. I focused the water quality monitoring on three parameters known to be stressors for aquatic organisms, if outside the range of historic variability: 1) water temperature; 2) dissolved oxygen; and 3) conductivity. Both baseflow water quality and insect biodiversity remained poor in all urban streams, regardless of whether the watershed was implemented with an SCM. Stream conductivity and maximum summer daily water temperatures increased with the level of urbanization. Surprisingly, both dissolved oxygen and daily maximum summer temperature were greatest in the urban-restored streams, suggesting that infiltration-based SCMs placed close to the channel heads (as RSCs typically are) may exacerbate poor water quality in some instances. I did observe reductions in stream temperature surges in one restored stream, indicating that the infiltration-based SCMs have the ability to infiltrate runoff and partially mitigate thermal effects of stormwater runoff in certain circumstances.

Finally, in Chapter 3, I used a paired-catchment approach and conducted high-frequency rainfall-stage monitoring for a 1-year period at two adjacent catchments: one that was implemented with an infiltration-based SCM, and one without any stormwater management treatment. During this time period, I also conducted high-frequency groundwater monitoring within the infiltration media (i.e., seepage bed) of the infiltration-based SCM itself to quantify changes in subsurface storage during rainfall events. During the monitoring period, I observed that the SCM effectively captured all runoff for rainfall events less than 5 mm in size, and intercepted all runoff for some rainfall events up to 30 mm in size. Overall stream responses in both catchments were most significantly correlated with rainfall intensity. Structural equation modeling suggested that the additional storage provided by the SCM modulated the effect of rainfall event size on stream response, but did not mitigate against the effect of rainfall intensity. Analysis of individual rainfall events revealed that the SCM conveyed untreated runoff to the downstream channel during high-intensity rainfall events, even when subsurface storage was available in the seepage bed.

These results overall suggest the infiltration-based SCMs have the potential to mitigate some hydrological effects of urbanization, but that the complete mitigation of the effects of urbanization may not be feasible through this approach due to its centralized design. Despite some hydrological benefits reported by this work, there were not significant improvements in water quality or insect biodiversity. Efforts to mimic the distribution of the hydrological processes lost through urbanization are likely the best approach to restoring urban watershed hydrological function.

Chapter 1: Evaluating the effectiveness of infiltration-based stormwater management to restore hydrologic function in urban headwater catchments

Introduction

Storage of water is a key watershed hydrological function (Black 1997). A watershed's characteristic storage function controls the flow regime in its stream channel, which in turn shapes the structure and function of downstream aquatic ecosystems (Bunn and Arthington 2002; Poff and others 1997). Watershed storage, broadly, is the volume of water that can be retained within its boundaries (McNamara and others 2011), and a suite of hydrological processes support this storage function, including: temporary retention in surface storage zones (Phillips and others 2011); infiltration and temporary retention in the vadose zone (Rimon and others 2007); and the removal of water from these storage zones by deep percolation or evapotranspiration. The type, size, and spatial distribution of storage zones across a watershed control the magnitude of rainfall partitioning into runoff, as well as spatial and temporal patterns of runoff delivery to streams (Wagener and others 2007). Many factors control watershed storage, such as geology and topography (Sayama and others 2011), vegetation and climate (Christensen and others 2008), and antecedent moisture conditions (Tromp-van Meerveld and McDonnell 2006). When storage becomes depleted, catchment flowpaths (both surface and subsurface) become hydrologically connected and accelerate the delivery of water into the stream network (Detty and McGuire 2010; Hopp and McDonnell 2009; Phillips and others 2011).

Landscape disturbances, such as urbanization, can profoundly alter watershed storage. During urbanization, impervious cover replaces permeable soil and vegetation, thereby reducing infiltration into subsurface storage zones (Gregory and others 2006). The construction of storm sewer networks directly connects sources of stormwater runoff to stream channels, bypassing natural storage zones in the watershed (Leopold 1968). As a result, urban streams experience more frequent high flows with greater rainfall-runoff ratios, altered groundwater recharge rates, and more synchronous flowpaths delivering runoff to the stream channel (Rose and Peters 2001; Walsh and others 2012). Altered flow regimes in urban streams can increase sediment transport, degrade water quality, and reduce aquatic biodiversity, a globally documented phenomenon known as the "urban stream syndrome" (Booth and Jackson 1997; Paul and Meyer 2001; Walsh and others 2005b). Questions remain, however, about whether watershed management practices implemented in urban landscapes sufficiently recover lost watershed hydrologic functions, such as storage (Bernhardt and Palmer 2007; Palmer and Bernhardt 2006; Shuster and Rhea 2013; Walsh and others 2005a).

Urban watershed management practices vary greatly in design, but most share the goal of mitigating the effects of stormwater runoff on streams (MDE 2009). First generation watershed management practices, such as wet or dry ponds, were designed to reduce peak flows by temporarily retaining runoff generated in the catchment and releasing it slowly over time (Burns and others 2012). These practices, however, historically perform poorly at mitigating the effects of stormwater runoff (Hancock and others 2010), and stormwater runoff remains a major stressor to urban stream ecosystems (NRC 2001). In response to continued urban stream ecosystem degradation, stream ecologists have argued for watershed management to focus on restoring the entire flow regime in order to recover stream ecosystem function (Walsh and others 2016; Walsh and others 2005a). To achieve this, watershed management projects should enhance watershed storage and minimize surface hydrologic connectivity between impervious surfaces and stream ecosystems (Walsh and others 2009). Newer watershed management approaches that emphasize infiltration, evapotranspiration, and distributed storage may have the greatest potential for restoring streamflow patterns (Holman-Dodds and others 2003; Hood and others 2007). For example, site-scale studies on individual bioretention facilities demonstrated their effectiveness for both runoff reduction and pollutant retention (Davis and others 2009; Hunt and others 2006). Low impact development has also shown to reduce surface runoff volumes and increase lag-times at a site scale (Hood and others 2007; Wilson and others 2015). Other infiltration-based stormwater control measures, such as permeable pavement (Brattebo and Booth 2003), green roofs (VanWoert and others 2005), and, most recently, regenerative stormwater conveyances (Cizek 2014; Palmer and others 2014a) have shown the potential for reducing runoff and improving water quality.

Many of these studies of individual watershed management practices are conducted as case studies and often lack reference sites, making it difficult to identify factors beyond the site that may affect performance. For example, a recent synthesis highlighted the important role local hydrological conditions play in the effectiveness of watershed management practices to reduce nitrogen loading (Koch and others 2014). Moreover, site-based studies on watershed management practices rarely follow their effects down to the stream ecosystem, where many biological indices of the integrity of the ecosystem are monitored. In contrast, comparative hydrological studies across known environmental gradients are powerful for identifying factors that might be affecting watershed management performance. Moreover, comparative studies at the watershed scale are important for improving our understanding of the heterogeneity in watershed hydrologic responses to disturbances, such as urbanization (Hopkins and others 2015; McDonnell and others 2007; Nagy and others 2012). Urban catchment hydrology remains rich with opportunities to support basic discovery about watershed hydrological processes in disturbed landscapes (Burt and McDonnell 2015), while simultaneously addressing urgent watershed management issues if these are explicitly included in the study design.

The objective for this study was to understand how watershed management practices modulate the hydrological effects of urbanization at a watershed scale. Specifically, I sought to 1) quantify the changes in watershed hydrologic function due to both urbanization and watershed management activities; 2) identify catchment characteristics that influence the hydrologic functioning of watershed management practices within the catchment; and 3) identify key hydrologic metrics that can be applied in future urban hydrology field studies to assess watershed hydrologic function. I quantified hydrologic metrics to describe watershed storage, flowpath connectivity, and the resulting stream flow regime in 11 headwater catchments spanning an urbanization-restoration gradient. I used regenerative stormwater conveyance systems (RSCs) as an example watershed management practice for this study, which are an emerging approach being widely adopted in the mid-Atlantic region to restore urban watersheds. RSCs are centralized structures designed to serve as infiltration hotspots in the landscape, and therefore have the potential of restoring storage functionality to urban watersheds (for more detail on design, see (MDE 2009; Palmer and others 2014a).

Methods

Study site description

I conducted this study in the greater Annapolis region, Maryland, USA (Figure 1.1). An urban-suburban area in the Coastal Plain Province, this region is underlain by poorly consolidated marine sediments, primarily sands, silts and clay of Paleocene age (Angier and others 2005). Precipitation falls mostly as rainfall with an annual average of 1120 mm, evenly distributed throughout the year. Urban development is primarily on flat hilltops and low gradient hillslopes. Storm sewer outfalls are usually placed along ephemeral first order streams that have migrated headward into steeper slopes in response to runoff generated first by agricultural practices and, more recently, urban development. Storm sewer networks have extended the natural stream drainage system, causing further headward channel migration and ephemeral gully formation (Brown and others 2010). Concerns over the poor performance of stormwater runoff management (CSN 2008; Hancock and others 2010) have led to new stormwater designs such as regenerative stormwater conveyance systems (RSCs), which are the focus of this study. Briefly, they are implemented in incised ephemeral gullies or at the heads of perennial first-order streams below storm sewer outfalls in small, urban watersheds (Palmer and others 2014a). Comprised of a series of connected infiltration pools, these structures theoretically enhance infiltration and transient storage of stormwater through a constructed sand seepage bed and surface pools; stabilize eroding banks with sandstone bedrock; and dissipate energy of stormwater with step-pool features (Flores and others 2009).

Seven of the 11 headwater (5.6 to 48.8 ha) catchments chosen for this study have varying amounts of urbanization (20 - 76.7 % impervious cover) and extensive storm sewer networks. The four remaining study catchments have less than 10% impervious cover and no stormwater infrastructure serve as “forested” reference sites (Figure 1.1; Table 1.1). All are drained by first-order, perennial streams. Three of the seven urban watersheds have been restored with a RSC structure between the main storm sewer outfall and the downstream drainage network (Figure 1.1 insets). For these structures to comply with state water quality regulations (MDE 2009), they must provide adequate storage for runoff generated from a 1-inch, 24-hour rainfall event; the storage is a combination of surface storage in pools and subsurface storage in the seepage bed.

Five of the eleven urbanized watersheds, including two of the three restored catchments contain additional stormwater management structures, the largest of which are first generation wet and dry basins, which have been show to perform poorly for reducing runoff (CSN 2008; Hancock and others 2010). Regardless, I adjusted catchment impervious cover values to reflect only impervious cover not treated by these upland BMPs. The total and untreated values of impervious surface percentages are provided in Table 1.1. See section A of the Supplemental Information (SI) for additional information on methods for calculating untreated impervious cover. Untreated impervious cover was used for all subsequent statistical analyses in this study.

Field data collection

To understand whether infiltration-based stormwater management improves hydrological function in urban catchments, I monitored precipitation, stream stage, channel morphology, and baseflow discharge. From these datasets, I developed a set of hydrologic metrics to compare across the study catchments. Hydrologic metrics derived from streamflow records have long been used to quantify changes in the flow regime from landscape disturbances (Richter and others 1996). Storm discharge, however, is difficult to measure in small streams, and stage-based monitoring has been used as an alternative to assess hydrological effects of land use/land cover changes (McMahon and others 2003; Roy and others 2005; Shuster and others 2008). There are significant trade-offs between using stage data or discharge data for characterizing hydrologic processes. Although discharge is required for quantifying runoff volumes, which is a quantitative assessment of watershed management hydrologic performance, developing a rating curve in urban headwater streams is inherently difficult because short-lived runoff peaks (minutes to hours) often hinder the full development of stage-discharges relationship, or yields one with high uncertainty (Harmel and others 2006). Stage-based metrics, on the other hand, coupled with high-frequency rainfall monitoring, can be used to quantify *relative* differences in hydrologic responses among many catchments with contrasting characteristics. For this study, since I was interested in the relative differences in hydrologic responses among adjacent catchments with contrasting land cover and management practices (forested, urban-degraded, urban-restored), I developed stage-based metrics as a research tool.

Gauging stations for continuous, high-frequency stage monitoring (June 2014-2015) were established at the stream outlets of the 11 catchments. Stage was recorded by unvented pressure transducers suspended by steel cables inside of perforated PVC pipes driven into the channel thalweg. Pressure transducers (corrected for barometric pressure) recorded pressure and temperature at 3-minute intervals from June 20 to August 14, 2014, and at 2-minute intervals from August 14, 2014 until June 20, 2015. Streambed aggradation and erosion was monitored monthly by measuring the vertical distance between the top of the PVC and the streambed height over time. Channel cross-sectional surveys were established in July 2014 at each gauging station, and measured again at the end of the monitoring period. Metrics describing channel morphology were used to normalize some stage-based hydrological metrics. Stream stage was used to quantify several hydrological metrics related to either watershed hydrologic function or the flow regime (see next section).

Discrete measurements of discharge at baseflow conditions were quantified during the monitoring period using the velocity-area method (Marsh McBirney electromagnetic current meter model 201D). Baseflow was defined to be at least 24-hours after a rainfall event, and stage hydrographs were analyzed for each discrete discharge measurement day to ensure measurements were not taken during unsteady hydrological conditions (e.g. receding limb from a previous storm). Two rain gauges (Onset Hobo model RG3-M) were deployed at the northern and southern ends of the study area where there was no overlying canopy (Figure 1.1); these recorded the timestamps of each 0.2 mm of rainfall. Daily rainfall totals were calculated and corroborated with a nearby CoCoRHAS rain gauge. Precipitation records were interpolated to 5-minute rainfall totals to quantify rainfall intensities, but raw timestamps were used for delineating individual rainfall events (see section B of Supporting Information).

Rainfall events were defined as a period of rainfall separated by at least a five-hour rain-free period (otherwise known as minimum inter-event time, MIT; (Dunkerley 2008; Dunkerley 2015) in order to examine rainfall-runoff responses under a wide range of antecedent conditions. Preliminary analyses suggested the median hydrograph recession time for the 11 streams to be 4 hours, indicating about half the hydrographs from the watersheds had returned to base stage conditions at this time. Only rainfall events with similar rainfall totals and cumulative rainfall patterns at both rain gauges were retained for the analysis to ensure complete and even coverage of the rainfall events across the entire study area. In total, 81 rainfall events were defined using these criteria, and were evenly distributed across the four seasons. Storm duration, total rainfall, average and maximum rainfall intensity, as well as a 24-hour antecedent precipitation index were quantified for each of the 81 rainfall events.

Hydrologic metric descriptions

I selected two types of hydrological metrics to assess hydrologic differences across the 11 catchments: metrics that describe a watershed hydrologic function, such as watershed storage or flowpath connectivity; and metrics that describe a component of the flow regime in the stream. We identified six metrics that fit these criteria: 1) Mean annual baseflow; 2) minimum runoff thresholds; 3) rainfall-runoff lag times; 4) duration of stormflow hydrographs; 5) runoff frequency; and 6) a flashiness index. The first two metrics describe watershed storage. *Mean annual baseflow* expresses long-term hydrologic storage of a catchment (Bhaskar and others 2016a; Roy and others 2005) and was calculated by averaging monthly baseflow discharge measurements normalized by catchment area. *Minimum runoff thresholds* for each catchment are used to identify the apparent storage capacity of the catchment during rainfall events (Ali and others 2015; Hood and others 2007; Loperfido and others 2014). Minimum runoff thresholds were quantified using a piecewise regression between rainfall depth and a stream response magnitude metric (e.g., change in stage during the rainfall event) for each of the 81 precipitation events. Breakpoints were identified between the first and second regression lines, and were used as estimates of the rainfall depth at which the storage capacity of the watershed is exceeded. Breakpoints were identified only if the slope of the first line was not significantly different than zero. I hypothesized that, if the restorations were effective at enhancing storage in the catchment, I would observe increases in mean annual baseflow and minimum runoff thresholds compared to the urban-degraded catchments.

Additional metrics based on hydrograph characteristics were used to quantify catchment flowpath connectivity. *Lag times* have often been used to describe changes in hydrological responses from land-use change (Hood and others 2007; Leopold and others 1964). Storm sewer networks and gullies increase surface flowpath connectivity and flow velocity, thereby reducing travel times to stream channels. Watershed restoration practices that infiltrate stormwater runoff may decrease surface flowpath connectivity by disconnecting the drainage network, thereby providing opportunities for diverting surface runoff into significantly slower porous media flow (Jarden and others 2016). Lag times for this study were calculated as the time between the center of rainfall mass (50th percent of the cumulative rainfall for the event; the hyetograph centroid) and the stream stage hydrograph peak (Hood and others 2007). Only rainfall events that generated a runoff response (operationally defined as a 1-cm rise in stage or greater) at all sites were used initially (n = 17 events). These events were further limited to those with simple rainfall patterns (single peak, shorter duration, etc.) that facilitated the lag time analysis. *Average hydrograph duration* expresses the average time a stream spends above base stage conditions during runoff events, with base stage defined as the mean stage for 1-hour prior to the start of each rainfall event. Runoff hydrograph durations in small catchments may reflect their connectivity in delivering runoff to the stream channel (Hopkins and others 2015). I hypothesize that, if the watershed restorations were performing their hydrological function, I would observe longer lag times and storm hydrograph durations.

Finally, two metrics were used to quantify changes in the flow regime in the downstream ecosystem. *Runoff frequency,* or the percentage of rainfall events that generated an observable runoff response at the stream outlets, describes the frequency of flow disturbances in downstream ecosystems (Konrad and others 2005; Poff and others 1997). In this study, a runoff response was operationally defined as a 1-cm or greater increase in stage during the rainfall event. McMahon and others (2003) used a similar stage-based metric to assess changes in hydrology to urbanization, and found the frequency of high stage conditions (e.g., runoff frequency) were highly correlated with percent of developed land in the catchment. A related metric is the *flashiness index*, which is a measure of the rate-of-change of streamflow (Baker and others 2004). Flow variability, or flashiness, describes how quickly stage or discharge changes during runoff events (Poff and others 1997). Flashy hydrographs can reduce streambank stability, which can cause bank erosion and channel widening (Konrad and others 2005), as well as impact biofilms and macroinvertebrate communities that lack access to flow refugia (Biggs and Close 1989; Lancaster and Hildrew 1993). Although flashiness is usually expressed with discharge data, McMahon and others (2003) also used stage to calculate a stage-based flashiness index. For this study, flashiness was defined as the rate-of-change in stage during the rising limb of each storm hydrograph, and was calculated for all sites during each rainfall event that generated an observable runoff response. Given that changes in stage are also controlled by the catchment area and the channel geometry of the monitoring site, I normalized each site's rate-of-change value for their channel's width-depth ratio at peak stage during each rainfall event.

Statistical analysis

All statistical analyses were conducted using R (version 3.2.0). Analysis of variance was conducted on select metrics to test for differences among catchment types (forest, urban-degraded and urban restored). Piecewise regression for defining minimum runoff response thresholds were computed using the segmented package in R. For each individual hydrological metric, I tested for the effects of percent impervious cover, restoration status (yes or no), and an interaction between impervious cover and restoration status using mixed effects linear regression. Catchment area was included in the regression models as an additional explanatory variable. I conducted a principal component analysis on the six metrics using the princomp function in R. I used the variable loadings to identify hydrologic metrics which explained the majority of the variability in the dataset, and to explore how the 11 watersheds are distributed in multivariate space. Inspection of residuals determined that two metrics were not normally distributed (flashiness index and minimum runoff thresholds), and these two metrics were natural-log transformed before performing any statistical analyses. The metrics were all centered (means were removed) and standardized (standard deviation was scaled to 1) for the principal components analysis. I also conducted a regression analysis between the scores of the first two principal components (PC1 and PC2) against the abovementioned catchment characteristics.

Results

Assessing the effects of urbanization

Mean annual baseflow declined with impervious cover (Table 1.2), and was almost three times higher in the forested catchments than in the urban-restored or urban-degraded catchments (Figure 1.2A; $p < 0.002$). Mean annual baseflow at each watershed was highly correlated with mean summer baseflow (summer = July, August, and September months; $R^2 = 0.96$; $p < 0.001$). Mean summer baseflow was also significantly greater in the forested catchments than the two urban watershed types ($p < 0.004$). Among the urban catchments, SALT1 (urban restored) and CC (urban degraded) had the greatest mean annual baseflow (0.053 and 0.052 l/s-ha, respectively), though both were still more than 25% lower than the forested catchment with the lowest mean annual baseflow (SW3, mean = 0.072 l/s-ha).

Breakpoints for identifying *minimum runoff thresholds* were identified in 9 of the 11 catchments, indicating threshold hydrologic behavior in response to rainfall (Figure 1.3A). The two catchments that did not exhibit runoff threshold behavior, CC and CH2, were both urban-degraded catchments. Their rainfall-stage responses exhibited log-linear increases in runoff as a function of rainfall depth. For these sites, we used the smallest rainfall event that generated a runoff response as the minimum runoff threshold. Therefore these two sites do not have confidence intervals associated with their runoff threshold (Figure 1.3A). Minimum runoff thresholds ranged from as high as 15 mm (CH1, urban-restored) to as low as 0.52 mm (CC). Across the sites, thresholds declined with increasing impervious cover (Table 1.2, Figure 1.3A).

Centroid lag-to-peak times were significantly correlated with catchment area ($p < 0.002$); since lag time is well-known to increase with basin area (e.g. Dunne and Leopold), lag times were normalized by catchment area for the final regression analyses. Catchment area-normalized lag times declined with increases in impervious area (Table 1.2, Figure 1.2B), which is consistent with other studies that examined lag times and urbanization (Hood and others 2007). The shortest lag times were observed at CH2 (urban-degraded; mean lag time = 1.6 min/ha, or 20.4 minutes whereas the longest area-normalized lag times were measured at SALT1 (urban-restored; mean lag time = 5.8 min/ha, or 44.7 mins for the area –normalized or raw mean lag time, respectively). Even after normalizing lag times for catchment area, it was still a significant predicator in the model (Table 2) largely due to the leverage of one site (SALT1).

Average hydrograph duration ranged from 10 to 19 hours. The shortest duration was observed at ML, an urban degraded site and the longest was observed at CH1, an urban restored site. Forested catchments had significantly longer runoff durations than the urban degraded catchments as a group ($p < 0.02$). *Runoff frequency* was greater with increasing impervious cover (Table 1.2). The percentage of rainfall events that generated a runoff response in the study catchments ranged from 33% to 90% (27 to 69 of the 81 events during the one-year period; Figure 1.3B), which translates into flow disturbances occurring as frequently as every 5 days (CC, urban- degraded) to as infrequently as every 13 days (CH1, urban-restored). Catchment area was significantly correlated with the *flashiness index* ($p < 0.01$), therefore each flashiness index value was normalized by site catchment area. I observed that area-normalized flashiness indices increased with an increase in impervious cover across the study area ($p < 0.001$; figure 1.2C) with the highest mean flashiness index observed at CC (urban degraded) and the lowest index at SW3 (forested).

Assessing the effects of watershed restoration

I observed significant effects of watershed restoration implementation in three of the hydrologic metrics in the study: 1) minimum runoff thresholds, 2) runoff frequency, and 3) storm runoff duration. For these three metrics, the restoration model coefficient indicated a partial reversal of the urbanization effect (Table 1.2). For example, minimum rainfall thresholds declined with increasing impervious cover, but sites implemented with restorations had, on average, higher thresholds than expected for their impervious surface area (Table 1.2). Similarly, my results indicate that restoration activities may lower runoff frequency and lengthen hydrograph durations. I observed a significant interaction between impervious cover and restoration status for all three metrics. In each case, the interaction sign was in the same direction as the urbanization coefficient (and opposite of the restoration coefficient) suggesting that the restoration effect diminished with increasing impervious cover. Figure 1.3 illustrates that the significant restoration effect observed in minimum runoff thresholds, frequency of runoff responses, and event hydrograph duration is likely driven by only one watershed (CH1), which has 22.2% impervious cover, significantly less than the other two restored catchments.

Correlation among hydrologic metrics

Not surprisingly, several of the hydrologic metrics were correlated with one another (Table 1.3). Minimum runoff thresholds were highly correlated with runoff frequency, because streams draining catchments with lower runoff thresholds will exceed their storage capacity and respond to rainfall events more frequently. The strong correlation between runoff event duration and the flashiness index may reflect the shifts in runoff delivery processes within the watersheds as they are urbanized. Storm sewers systems extend the drainage network upstream into the catchment above stream channels, and the low roughness of pipes creates short travel times within these efficient drainage networks. The resulting hydrographs are short in duration, with steep rising limbs indicating much greater rates of change (and, presumably, greater peak discharges). These four metrics (minimum runoff thresholds, runoff duration, runoff frequency, and flashiness index) collectively describe changes in available hydrologic storage and the resulting changes in the runoff hydrograph. Interestingly, none of these storm-event based metrics were significantly correlated with mean annual baseflow, which suggests the storm hydrograph metrics and the baseflow metric may be capturing different processes by which urbanization affects watershed hydrologic function.

Principal components analysis

A principal component analysis was conducted to identify the metrics that describe the majority of the variance among the catchments and to visualize how the metrics collectively describe the overall differences in hydrologic behavior among the 11 catchments. The first two components of the PCA explained approximately 87% of the total variability in the overall dataset (Figure 1.4). The first principal component (PC1) explained 60% of the variance, and the four storm-event based metrics discussed above were all highly loaded onto this component (Table 4). Runoff frequency explained the greatest amount of variance within this component. Principal component 2 (PC2) explained 27% of the overall variance, and mean annual baseflow and lag times loaded highest on this component (Table 1.4). In general, the forested catchments clustered on the low end of PC1 and PC2, while the urban degraded catchments clustered at the upper end of PC1 and PC2 (Figure 1.4). The urban restored catchments varied widely in their overall placement along PC1 and PC2, indicating variable effects of watershed restoration on the hydrological metrics across the three sites. In the regression analysis, I observed impervious cover to be a significant predictor of both PC1 and PC2. I also observed a positive restoration effect (as seen in thresholds and runoff frequency), as well as a significant interactive effect for PC1 scores. Impervious cover and catchment area were significant predictors for PC2 scores.

Discussion

As expected, the hydrological effects of urbanization were clearly detected in all six hydrologic metrics. Mean annual baseflow, minimum runoff thresholds, lag times, and runoff event duration all decreased with increasing impervious cover. Although changes in baseflow in response to urbanization are complex (Bhaskar and others 2016a; Price 2011), lower mean annual baseflow observed in the urban streams is consistent with other studies conducted in the humid Eastern United States (Rose and Peters 2001). Minimum runoff thresholds quantified in this study are within the range of thresholds observed in other urban catchments. Loperfido and others (2014) identified thresholds for urbanized mid-Atlantic watersheds ranging from 7.5-11 mm, though these watersheds were larger in size (110-700 ha). In smaller urban catchments, Hood and others (2007) measured minimum runoff thresholds ranging from 0.9 mm to 6 mm. Declining runoff hydrograph durations with increasing urbanization has also been documented in several U.S. metropolitan regions (Hopkins and others 2015). I also documented increased runoff frequency and flashiness indices with impervious cover. Several other studies have quantified similar increases in high-flow frequency (Hopkins and others 2015; Roy and others 2005), and our flashiness results agree with other studies along urbanization gradients (Nagy and others 2012; Schoonover and others 2006).

Although I observed a significant restoration effect in three of the six hydrological metrics (runoff frequency, minimum runoff thresholds, and runoff hydrograph duration; Table 1.2), these effects were largely driven by only one restored catchment (CH1). These results suggest that only the restoration in the CH1 catchment may be effectively infiltrating stormwater runoff. The patterns at CH1 are consistent with other studies on infiltration-based stormwater management. For example, Loperfido and others (2014) found minimum runoff thresholds were greater in a watershed with distributed, infiltration-based stormwater management than at a similar watershed with centralized, detention-based stormwater management. Hood and others (2007) also observed a larger runoff threshold in a watershed implemented with low-impact development than at a watershed with conventional development. The significant interaction between restoration activity and impervious cover for these three metrics (Table 2) suggests the relative benefits of watershed restoration declines with increasing impervious cover.

Disparate hydrological responses among the three restored urban watersheds

The principle components analysis indicated general clustering among the forested catchments and the urban-degraded catchments (Figure 1.4), and also illustrated the range of hydrologic behaviors among the three urban restored catchments. First, the RR restored catchment clustered closely to all the other urban degraded watersheds (CC, CH2, SALT2 and ML; Figure 1.4), suggesting the restoration in this catchment is not significantly modifying any hydrological processes. In contrast, the restored catchment with the lowest level of impervious cover (CH1, 22.2% untreated impervious cover) exhibited a high minimum runoff threshold, suggesting that this restoration structure supports the infiltration and storage of stormwater runoff for rain events up to 15 mm (Figure 1.3A). The average storm hydrograph duration was also longer in CH1 than any other watershed as well. Collectively, these results suggest that the CH1 restoration is effectively enhancing stormwater infiltration and subsurface storage, and decreasing flowpath connectivity by serving as a sink for runoff.

Despite enhanced runoff infiltration in the CH1 watershed, mean annual baseflow in its stream channel is significantly lower than the forested reference catchments (Figure 1.2A), suggesting that the infiltrated runoff is not recharging longer-term storage zones that supply baseflow to the stream. One of the urban-degraded catchments (CH2) can be used as a paired, non-restored urban reference catchment for CH1 because they are located adjacent to one another and are very similar in terms of impervious cover, age of development, geology, and topography (Figure 1.1). A student's t-test of monthly baseflow measurements between CH1 and CH2 show no difference between the two sites ($p = 0.83$), suggesting this difference may not be due to inherent variability among headwater catchments in the region. Although persistently low baseflow in CH1 suggests that recharge to groundwater have not been restored by the project, there is a possibility is that enhanced recharge from infiltrated runoff could be elevating baseflow downstream of the monitoring stream reach. However, the long hydrograph durations at this catchment may indicate the short-term release of the infiltrated runoff.

Finally, the restored catchment with the greatest impervious cover (SALT1) showed little difference in the metrics related to storm responses when compared with non-restored catchments (e.g., flashiness, runoff frequency, runoff thresholds). I did observe, however, higher baseflow and longer lag times at this site. Low minimum runoff thresholds and a high runoff frequency suggest little or no infiltration of runoff is occurring in this catchment. The location of the restoration within the stream floodplain may limit infiltration opportunities, given that lowland Coastal Plain streams are often groundwater discharge zones (Bachman and others 1998), and as such water tables are typically shallow in the floodplain. Elevated baseflow observed in SALT1 could be due to the catchment's larger contributing area (48.8 ha); catchment area was indeed a significant predictor in the regression model (Table 1.2). This is consistent with the longer lag times observed for this watershed, which is also influenced by catchment area. Alternatively, higher baseflow and lag times could indicate enhanced surface detention as a result of the restoration practice. A similar pattern was observed in a suburban watershed, where a wetland retaining stormwater runoff delayed storm peaks (Burns and others 2005).

Catchment controls on hydrologic behavior of watershed restoration projects

Why did the three urban restored watersheds exhibit such disparate hydrologic behavior? Catchment size, impervious surface area, and other characteristics of the catchment may influence the types of hydrological processes that each restoration project supports, which in turn controls the fate of the incoming stormwater runoff. In general, the effectiveness of a watershed restoration (or any stormwater control measure) may be constrained by characteristics of both the natural environment (soils, topography, geology) and built environment (development age and intensity, storm sewer configuration), a concept known as watershed capacitance (Miles and Band 2015). I selected watersheds within close proximity to each other (within a 5-mile radius) for this study to minimize differences in the natural environment. However, I allowed for patterns in urban development to vary within the restored catchments to explore how similar restoration practices work within different configurations of the built environment.

Impervious cover may be a major factor controlling the effectiveness of these watershed restoration projects. The restoration design is explicitly tied to the impervious cover and catchment area of the watershed (MDE 2009), so all of these restorations should be able to accommodate the volume of runoff generated in its catchment from a 1-inch, 24-hour event. Yet, I observed very different minimum runoff thresholds across the three RSCs in this study and additional RSCs monitored elsewhere. A recent study in North Carolina documented the hydrologic performance of an RSC in a Coastal Plain watershed similar in size to CH1 (5.2 ha) but with about half the impervious cover (12.3%). Runoff frequency at the outlet from this site was very low (2 occurrences of runoff conveyance during their two-year monitoring period) and the restoration retained runoff generated from rainfall events as large as 45 mm (Cizek 2014). CH1, in comparison, infiltrated runoff completely for events as large as 15 mm. The RR catchment, with twice the impervious area as CH1 (40% impervious), only retained runoff completely for events smaller than 1.2 mm on average (Figure 1.3A), and SALT1 (50.7% impervious) performed even worse for retaining runoff. This clearly shows that greater impervious cover limits the ability of these restoration projects to completely capture stormwater runoff for even small rainfall events well within the design criteria.

Restoration effectiveness could also be influenced by catchment area and characteristics of the storm sewer network. Besides being more developed, the RR catchment also had a larger contributing area (Table 1.1), and a much more connected storm sewer network than CH1 (Figure 1.1 inset). In this watershed, the storm sewer network may be delivering runoff at such a great rate into the restoration that infiltration opportunities are minimal unless the rainfall event is very small. The centralized position of these restoration practices (placed in topographic depressions between the main watershed storm sewer outfall and the perennial stream channel) allows these structures to serve as a type of gatekeeper node within the larger drainage network (Phillips and others 2011). When the storage capacity of the restoration structure is exceeded, the upland drainage network is hydrologically connected to the stream channel. A highly connected storm sewer network homogenizes flowpath lengths within the catchment, and delivers the bulk of runoff to the restoration structure within a short period of time. This may overwhelm the restoration structure even at low rainfall depths, and allow untreated runoff to be conveyed into the downstream channel more frequently than at a watershed with lower storm sewer connectivity. Larger contributing areas draining to the restoration structures can exacerbate this issue by generating a larger volume of runoff as well.

Watershed restoration design and placement is constrained by surrounding landscape characteristics, which in turn may control the types of hydrological processes the practice can support. For example, in SALT1 much of the watershed has been developed, and likely constrained watershed restoration activities to its riparian zone and floodplain (Figure 1.1 inset). As a result, the SALT1 design relies on lateral surface storage in the floodplain rather than the upland vadose zone as with the CH1 and RR restoration projects. The depth of the water table may control the partitioning of the two primary ways this restoration design enhances watershed storage: through surface storage in large pools (which enhances surface detention) or through subsurface storage in the seepage bed and surrounding vadose zone (which enhances infiltration and potentially groundwater recharge). With limited infiltration potential, surface detention may be the only hydrological process the restoration practice at SALT1can support. Indeed, seasonally elevated groundwater levels have been documented within the seepage bed of another RSC in North Carolina (Cizek 2014) so it is possible this process may be occurring at our sites as well. In contrast, CH1 and RR have subsurface storage zones that are presumably well above the groundwater table, which may support more infiltration/percolation and subsurface storage opportunities.

Implications for improving water quality with infiltration-based watershed restoration

Poor water quality in downstream aquatic ecosystems is often the primary motivation for implementing management and restoration projects in urban watersheds. However, improved water quality may not be achieved if watershed restoration practices do not enhance or restore hydrological processes in the watershed. Although this current study did not focus on water quality, ongoing research in the same region show mixed results for reducing sediment loads from this type of infiltration-based watershed restoration design. In one study, storm sampling of total suspended sediments (TSS) at the CH1 and CH2 catchments show lower TSS loads exported from CH1 compared to CH2 (negative control), although this difference decreased during larger rainfall events (Palmer and others 2014a). Filoso and others (2015) also examined TSS loads into and out of two urban stream reaches (impervious cover for both watersheds > 40%) that had been restored using a similar design as SALT1. They found exported TSS loads were not significantly lower than the incoming TSS load from the watershed, and the net exported TSS load increased with increasing storm magnitude. RSCs may be promising for reducing TSS loads at low to moderate levels of urbanization simply because the design stabilizes eroding banks and channels. However, these studies provide further evidence of the limitations of RSCs in highly urbanized regions. Moreover, these results show that restoration effectiveness will vary with changing hydroclimatic conditions as well (e.g., storm magnitude).

Applications for future studies on watershed management and restoration

One of my research goals was to identify metrics that could be easily measured at many watersheds (including populations of reference catchments) to increase our understanding of how urbanization and watershed restoration activities manipulate the routing, storage, and release of runoff from catchments. Based on our analysis, I believe two of the six hydrologic metrics used in this study capture the fundamental changes in hydrologic behavior across our study catchments, and therefore have the potential to be useful in future urban hydrology field studies. The first metric, runoff frequency, describes the resultant change in runoff delivery to the stream channel from decreased watershed storage. Runoff frequency was highly correlated with three other metrics, including minimum runoff thresholds, which is a more robust metric for quantifying watershed storage (Table 1.3). The runoff frequency metric captures changes in rainfall-runoff partitioning from both urbanization as well as restoration, but could be measured over a shorter period of time than this study (3-6 months rather than the 1-year period used in this study). Runoff frequency is ecologically significant, as more frequent high flows in headwater streams is linked to lower biodiversity (Roy and others 2005; Walsh and others 2016).

The second metric, mean annual baseflow, captures the level to which rainfall is partitioned into longer-term storage, beyond any short-term storage that may occur immediately following a runoff event. I suggest that these two metrics in tandem are appropriate for assessing the effectiveness of watershed management for restoring watershed hydrologic function (Figure 1.5). If the processes supporting watershed hydrologic function were fully restored, one would observe both decreased runoff frequency from enhanced infiltration of runoff, as well as increased baseflow from percolation of that infiltrated runoff into long-term subsurface storage. I did not observe these combined processes occurring in any of the restored watersheds (Figure

5), suggesting that this particular design, which concentrates the infiltration of runoff above or adjacent to the stream channel, does not effectively reproduce the hydrological processes lost through urbanization. Alternative stormwater management approaches that emphasize de-centralized infiltration throughout the catchment, such as green infrastructure (Jarden and others 2016), may be better at restoring overall watershed hydrologic function because this approach more closely mimics the natural distribution of storage zones in undisturbed, forested landscapes.

Conclusions

I used a suite of hydrological metrics to evaluate changes in watershed hydrologic processes due to urbanization and subsequent watershed restoration practices. This multi-metric analysis, which leveraged both discrete discharge and continuous stage-rainfall monitoring data, revealed lower watershed storage, short duration hydrographs, flashier flow regimes, and greater runoff frequency with increasing urbanization. Infiltration-based watershed restorations showed limited success in modulating the hydrological effects of urbanization. Although one restored catchment demonstrated significantly enhanced infiltration of stormwater runoff, its mean annual baseflow remained low, indicating that enhancing infiltration and storage proximal to the channel head does not restore long-term storage and stream baseflow. Variable hydrological responses among the three restored catchments were likely influenced by catchment characteristics, including level of imperviousness, catchment size, and extent of the storm sewer network. I identified two metrics in particular that are easily quantified in many (>5) catchments over a relatively short period of time: 1) runoff frequency, which captures rainfall-runoff dynamics; and 2) baseflow discharge, which quantifies release of water from long term storage. Restoration actions designed to restore watershed hydrologic processes should ideally be addressing both short-term and long-term storage of rainfall, and these two metrics seem to capture these hydrological processes. This approach could be used by resource managers to gain a better understanding of how management practices can affect watershed function.

Tables and Figures

Table 1.1:. Catchment characteristics for the 11 watersheds in the study area. Watersheds are in order of their impervious cover.

Site ID	Watershed type	Catch area, ha	Impervious area, %	Adjusted (untreated) Impervious area, %[c]
SW3	Forest	6.9	1.0	1.0
SW1	Forest	13.1	1.9	1.9
SW2	Forest	8.5	4.9	4.9
SALT3	Forest	13.9	8.6	8.5
SALT2	Urban-degraded	18.6	22.0	21.5
CH1[a]	Urban-restored	5.4	22.2	22.2
ML	Urban-degraded	8.0	22.5	22.5
CH2	Urban-degraded	5.6	23.2	23.2
RR[a]	Urban-restored	11.4	43.0	40.0
SALT1[b]	Urban-restored	48.8	59.0	50.7
CC	Urban-degraded	33.5	76.7	65.9

[a] Implemented with vertical storage RSC watershed restoration (see text for details)
[b] Implemented with lateral storage RSC watershed restoration
[c] Impervious cover was adjusted for existing urban BMPs in SALT1, SALT2, SALT3, CC, and RR watersheds. See text and SI for additional information.

Table 1.2: Linear regression analysis to quantify the effect of impervious cover and restoration on the 6 hydrologic metrics and the first two principal components from the PCA. Model coefficients in bold indicate a significance, given α=0.05. Asterisks indicate size of the p-value.

	Estimates for model predictors				Model fit	
Response variable	**Imp**	**Rest**	**Area**	**Imp*Rest**	**adjR^2**	**p-value**
Baseflow	**-0.002***	0.01	**0.003***	-0.001	0.50	0.08
Threshold	**-0.08***	**29.9*****	**0.14***	**-0.68*****	0.92	0.0004
Frequency	**0.81****	**-67.3****	-0.37	**1.45***	0.87	0.002
Lagtime	**-0.06***	1.48	**0.12****	-0.02	0.68	0.02
Flashiness	**0.04****	-1.37	0.01	0.03	0.87	0.002
Duration	**-0.19****	**14.5***	0.20	**-0.33***	0.67	0.03
PC1	**0.09*****	**-7.9****	-0.06	**0.18****	0.88	0.002
PC2	**0.06****	0.92	**-0.11***	-0.02	0.72	0.02

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Table 1.3: Correlation matrix for the six hydrologic metrics. Correlation coefficients in bold indicate a significance, given α=0.05. Asterisks indicate size of the p-value.

	Baseflow	Threshold	Frequency	Lagtime	Flashiness
Threshold	-0.05				
Frequency	-0.26	**-0.90*** **			
Lagtime	0.47	-0.18	0.07		
Flashiness	-0.23	**-0.82** **	**0.92*** **	0.21	
Duration	0.50	**0.71* **	**-0.91*** **	0.11	**-0.77** **

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Table 1.4: Loadings for the first two principal components. Variables in bold have loadings greater than 0.5.

Metric name	**PC1**	**PC2**
Baseflow	-0.17	**-0.66**
Threshold	-0.47	0.24
Frequency	**0.52**	-0.03
Lag times	0.03	**-0.68**
Flashiness	0.49	-0.12
Duration	-0.49	-0.19

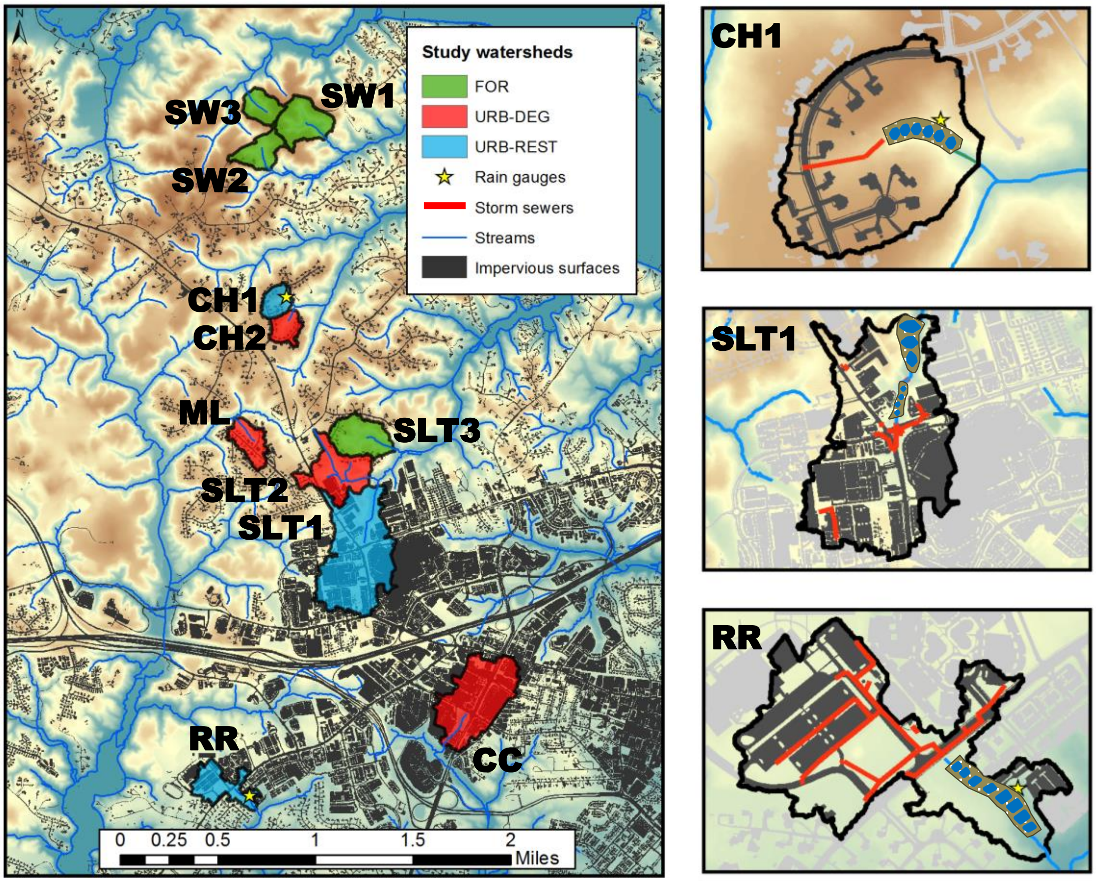

Figure 1.1: Site map of the 11 watersheds and locations of rain gages within the study area (left), and additional site details of the three restored watersheds (right three panels) , including storm sewer networks and location of the watershed restoration practice (depicted by blue pools).

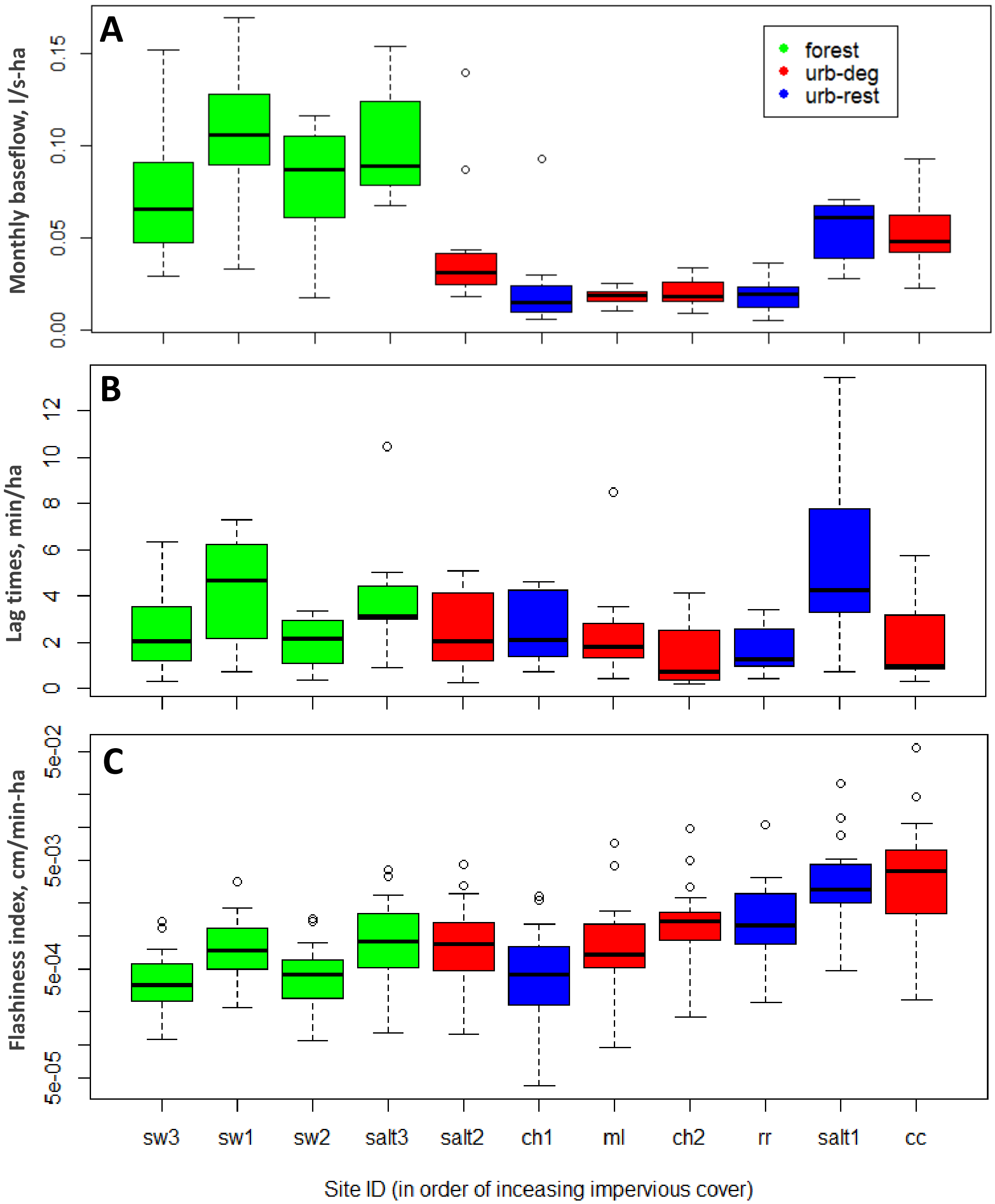

Figure 1.2: Boxplots for (**A**) baseflow normalized by catchment area (l/s-ha; n=12), (**B**) centroid to peak lag time normalized by catchment area (min/ha; n=7 events), and (**C**) flashiness index (cm/min-ha; n =17) for the 11 watersheds. Flashiness index (cm/min-ha) is the mean rate of change (cm/min) for a hydrograph rising limb, normalized by the stream channel width-depth ratio and catchment area (ha). Sites are ordered from lowest to highest impervious cover and shaded by watershed type. See Table 1 for catchment impervious cover percentages.

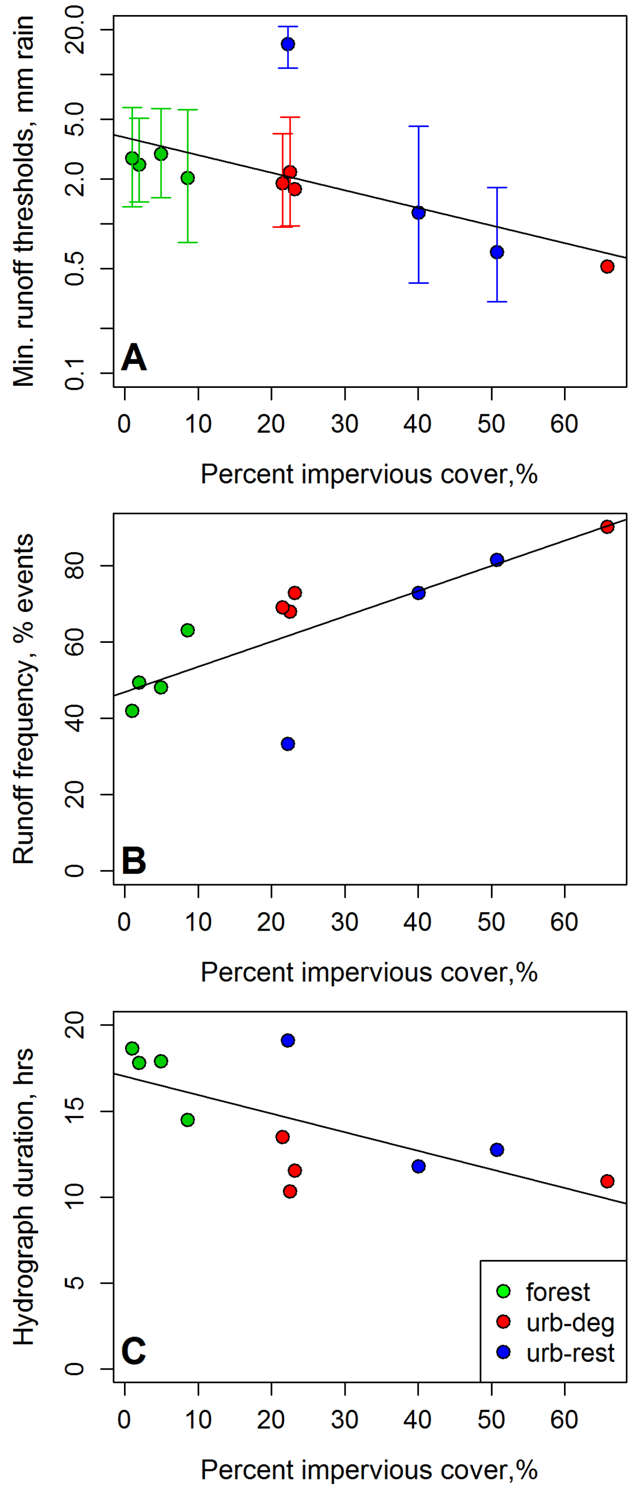

Figure 1.3: Percent impervious cover vs (A) minimum runoff thresholds (mm rain), (B) runoff frequency (percentage of rainfall events), and (C) mean duration of runoff events (hrs). For minimum runoff thresholds, whiskers indicate 5^{th} and 95^{th} percentile confidence intervals for thresholds identified through a breakpoint analysis.

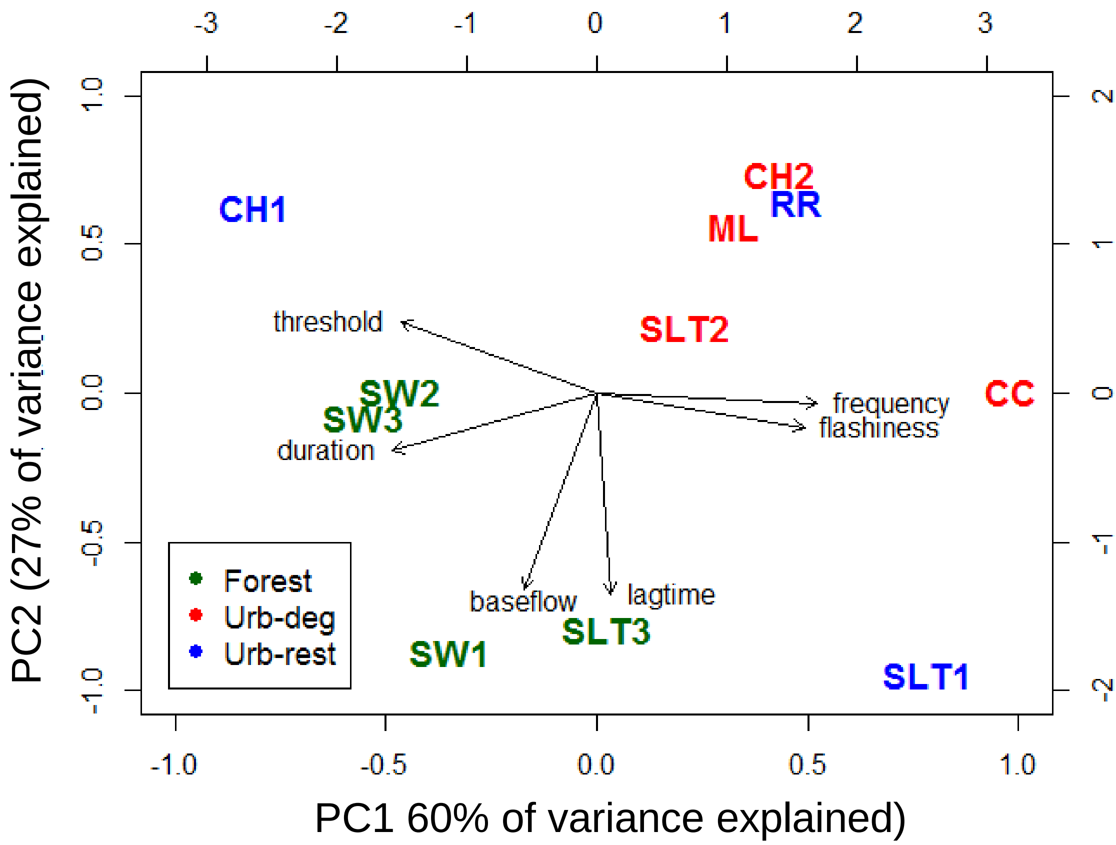

Figure 1.4: Results of a principal component analysis on the 6 hydrological metrics. Black arrows indicate which variables load most heavily on PC1 (x axis) and PC2 (y axis). Individual sites are indicated by their site ID (Table 1).

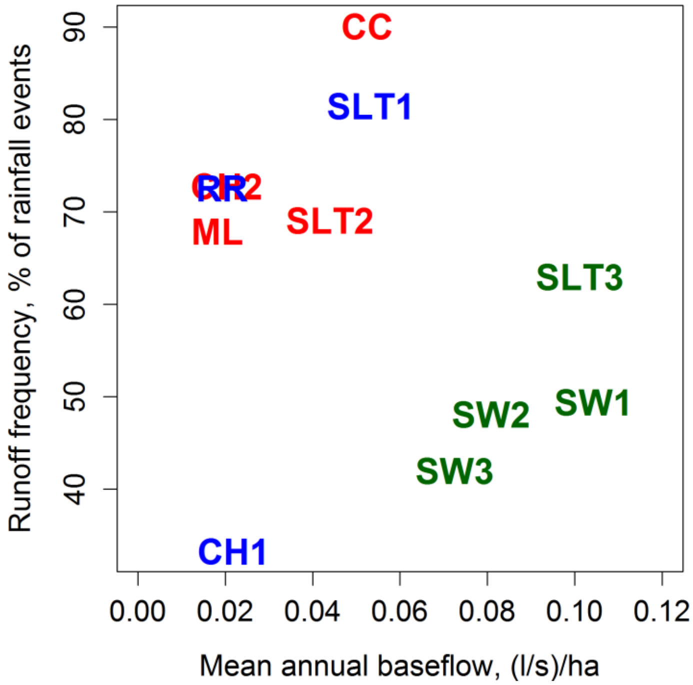

Figure 1.5: Relationship between area-normalized baseflow and runoff frequency.

Supplemental information

A. Accounting for upland BMPs in the study area

Five watersheds in the southern portion of the study area had been implemented with upland urban best management practices (BMPs). One was a forested catchment (SALT3); two were within the urban-degraded group (SALT2 and CC) and the remaining two were within the urban-restored group (RR and SALT1). Given that the focus of my study was to examine the effects of the RSC on hydrologic responses, there was a need to account for the impervious cover treated by these upland BMPs. To remove the effect of these BMPs, I adjusted the percent impervious cover (IC), the metric we used to describe the magnitude of urbanization, to account for IC that theoretically had been treated by a BMP up-gradient of the inlets to the RSCs, which were the focus of this study.

I leveraged a recently developed database from Ann Arundel County which catalogued each BMP with descriptions/types, ages, as well as spatial location within county boundaries. Using ArcGIS, we selected the BMPs which fell within the study watershed boundaries (Table SI-1.1). I then applied estimated runoff reductions based on the age and type of BMP, given that different types of BMPs (e.g., bioretention vs wet ponds) exhibit different hydrologic behavior, and that BMPs built in recent year adhere to stricter design standards. Runoff reduction estimates calculated by the Chesapeake Stormwater Network (CSN 2008) were based on a synthesis they had conducted on BMP effectiveness for runoff reduction and nutrient retention, and provided two tiers, or levels, of runoff reduction for variability in design. For BMPs built prior to 2000, I used the tier 1 runoff reduction estimates; tier 2 runoff reductions were applied to BMPs built later than 2000, which assumed a more advanced design (and therefore efficacy).

To account for the IC treated by upland BMPs, I calculated a treated drainage area for each BMP using individual BMP drainage areas and its assumed runoff reduction. For example, an infiltration trench in CC watershed (drainage area = 1.0 acres) was assumed to have a runoff reduction between 50-90%. This BMP was built after 2000, so I used the 90% runoff reduction estimate to calculate the drainage area treated by the BMP. To quantify the impervious cover area treated by the BMPs, I summed the total area treated by BMPs for each watershed, then multiplied that area by the watershed-averaged percent impervious (Table SI-1.2). Finally, I subtracted those IC acres treated by the upland BMPs from the overall watershed, and calculated an adjusted percent impervious cover that reflects the untreated IC in the watershed. Impervious cover in only three watersheds changed more than 1% after accounting for the upland BMPs: RR (3% reduction), SALT1 (9% reduction) and CC (11% reduction; Figure SI-1.1). This is a somewhat optimistic approach to addressing the BMPs, and using total impervious cover vs. untreated impervious cover did not change the overall results of the statistical analyses.

B. Precipitation event definition and characterization

Many of the metrics quantified in this study describe a component of stage hydrographs during rainfall events, and so I developed a standardized method to define rainfall events across the period of record (POR; June 2014 to June 2015). In rainfall-runoff studies, both streamflow records and precipitation records have been used to define high-flow, or runoff, events. For example, Ali and others (2010) used precipitation records to examine runoff responses in a stream, and applied a minimum inter-event (MIT) duration of 48 hours to their study. Conversely, Schuster and others (2008) defined a runoff event as a period of time bounded between adjacent local minima of water level (stage). Tromp-van Meerveld and McDonnell

(2006) used minimum changes in streamflow (either a rise of 0.4 l/s rise in streamflow within 3 hours or 30% rise in streamflow within 3 hours) to define a runoff event. For this study, I used the precipitation record to define our rainfall and runoff events, because we are interested in defining periods of time during which there is the *potential* of a runoff response in the study watersheds. This method enables us to not only simultaneously compare the runoff responses among the 11 watersheds, but also identify rainfall events that did *not* elicit a runoff response from the streams, in an effort to assess watershed storage. Both types of analyses are informative for understanding how urbanization and management modify watershed hydrologic function.

I used a minimum inter-event time (MIT) to define our rainfall events (Dunkerley 2015). The MIT defines the duration of the minimum duration of a rainless period of time, so rainfall events are therefore defined as the time between two rainless periods. I initially varied the MIT from 3 hours to 60 hours to explore the effect of MIT length on the number of rainfall events derived from the rainfall time series (Figure SI-1.2a). As expected, the number of rainfall events defined for the POR declined with increasing MIT length, because smaller rainfall events were aggregated together into single events as the length of the MIT increased. This aggregation led to a shift in the characteristics of the resulting rainfall events (Figure SI-1.2b), with a decline in the fraction of small rainfall events (0-10 mm), and an increase in the fraction of moderate sized rainfall events rainfall events (20 mm and larger). Given that I was interested in understanding watershed hydrologic behavior during small but frequent rainfall events, I chose a 5-hour MIT, which is similar to the widely adopted 6-hour MIT (Dunkerley 2008).

The application of a 5-hour MIT yielded approximately 118 raw rainfall events at the CH rain gauge (north end of study area) and 101 raw rainfall events at the RR rain gage (south end of study area) during the POR. Because I was analyzing runoff responses in all 11 streams during these events, Iwanted to retain only events that occurred across the entire study area. To ensure complete areal coverage during rainfall events, I then compared both rainfall event records from the two gages for their timing (start and end time), magnitude (rainfall total) as well as general rainfall patterns (intensity over the event) to identify rainfall events that 1) occurred at both gages and 2) were similar in rainfall characteristics. After filtering for comparable events, snowfall events were also removed. The final record was comprised of 81 rainfall events, evenly distributed across the four seasons. Rainfall totals for the two rain gages are therefore highly correlated for these 81 rainfall events (Figure SI-1.3). Rainfall totals ranged from 0.2 mm to 95 mm during the POR, but the majority of (60%) of rainfall events were less than 10 mm (Figure SI-1.4a and SI-1.4b).

Table SI-1.1: Detailed descriptions of the upland BMPs present in the study area (n=49), including BMP type, date built, drainage area, its assumed runoff reduction, and the estimated treated drainage area. Upland BMPs built before 2000 were assumed to have tier 1 (lower) runoff reductions, and those built after 2000 were assumed to have tier 2 (higher) runoff reductions. Estimated runoff reductions derived from Chesapeake Stormwater Network (2004).

SITE	BMP description	Date built	Drainage area, acres	Runoff reduction, %	Drainage area treated by BMP, acres
CC	Permeable pavement	5/20/2012	0.03	75	0.02
CC	Bioretention	2/2/2011	0.30	80	0.24
CC	Bioretention	2/20/2011	4.78	80	3.82
CC	Infiltration basin	5/9/2000	0.53	50	0.27
CC	Infiltration basin	3/10/2006	1.00	90	0.90
CC	Infiltration basin	1/6/2003	6.73	90	6.06
CC	submerged gravel wetland	5/20/2012	1.94	0	0.00
CC	Extended detention, wet	1/3/1990	1.13	0	0.00
CC	Step pool stormwater conveyance	5/20/2012	0.21	90	0.19
CC	Step pool stormwater conveyance	5/20/2012	0.26	90	0.23
CC	Oil and grit separator	2/20/2011	0.53	0	0.00
RR	Sand filter	7/8/2011	0.25	0	0.00
RR	Sand filter	7/8/2011	0.25	0	0.00
RR	Sand filter	7/8/2011	0.50	0	0.00
RR	Sand filter	7/8/2011	1.00	0	0.00
RR	Sand filter	7/8/2011	1.00	0	0.00
RR	Infiltration basin	7/8/2011	1.00	90	0.90
RR	Dry swale	7/8/2011	1.70	60	1.02
RR	Oil and grit separator	6/22/1998	0.91	0	0.00
SALT1	Bioretention	7/29/2005	0.00	80	0.00
SALT1	Bioretention	8/5/2008	1.16	80	0.93
SALT1	Infiltration basin	8/27/1987	4.70	50	2.35
SALT1	Infiltration basin	8/16/2006	4.80	90	4.32
SALT1	Infiltration basin	3/18/1991	0.27	50	0.14
SALT1	Infiltration basin	10/14/1992	0.60	50	0.30
SALT1	Infiltration basin	7/29/2005	0.77	90	0.69
SALT1	Infiltration basin	8/29/2000	0.86	50	0.43
SALT1	Infiltration basin	1/17/2007	0.93	90	0.84
SALT1	Infiltration basin	4/30/1987	1.28	50	0.64
SALT1	Infiltration basin	8/13/2014	1.84	90	1.66
SALT1	Infiltration basin	7/25/2001	2.30	90	2.07
SALT1	Infiltration basin	11/29/2004	2.45	90	2.21
SALT1	Landscape infiltration	1/15/2015	0.13	90	0.12
SALT1	Dry swale	5/14/1996	0.50	40	0.20

SALT1	Extended detention, wet	4/25/1996	14.31	0	0.00
SALT1	Extended detention pond, dry	5/6/1986	5.50	0	0.00
SALT1	Extended detention pond, dry	1/24/1995	25.70	0	0.00
SALT2	Infiltration basin	10/14/1992	0.60	50	0.30
SALT2	Infiltration basin	10/14/1992	0.60	50	0.30
SALT2	Infiltration basin	9/19/1990	0.90	50	0.45
SALT2	Extended detention, wet	5/11/2001	3.87	0	0.00
SALT2	Extended detention, wet	6/30/1988	18.40	0	0.00
SALT3	Infiltration basin	2/13/1991	0.05	50	0.03
SALT3	Infiltration basin	8/20/1999	0.05	50	0.03
SALT3	Infiltration basin	7/23/1993	0.09	50	0.05
SALT3	Infiltration basin	12/28/1999	0.10	50	0.05
SALT3	Infiltration basin	1/10/2000	0.11	50	0.05
SALT3	Infiltration basin	12/28/1999	0.11	50	0.05

Table SI-1.2: Summary table describing BMP implementation in each of the 11 watersheds. Area treated by upland BMPs were calculated by applying each BMPs runoff reduction to their drainage area and summing up the treated areas for all BMPs within a watershed.

Site ID	TYPE	Watershed area, acres	Total impervious cover, %	# BMPs	Area treated by upland BMPs, acres	Adjusted (untreated) impervious cover, %
SW3	FOR	17.1	1.0	0	0	1.0
SW1	FOR	32.4	1.9	0	0	1.9
SW2	FOR	21.0	4.9	0	0	4.9
SALT3	FOR	34.3	8.6	6	0.3	8.5
SALT2	U-DEG	46.0	22.0	5	1.5	21.5
CH1	U-REST	13.3	22.2	0	0	22.2
ML	U-DEG	19.8	22.5	0	0	22.5
CH2	U-DEG	13.8	23.2	0	0	23.2
RR	U-REST	28.2	43.0	8	1.6	40.1
SALT1	U-REST	120.6	59.0	19	15.6	50.7
CC	U-DEG	82.8	76.7	11	6.1	65.9

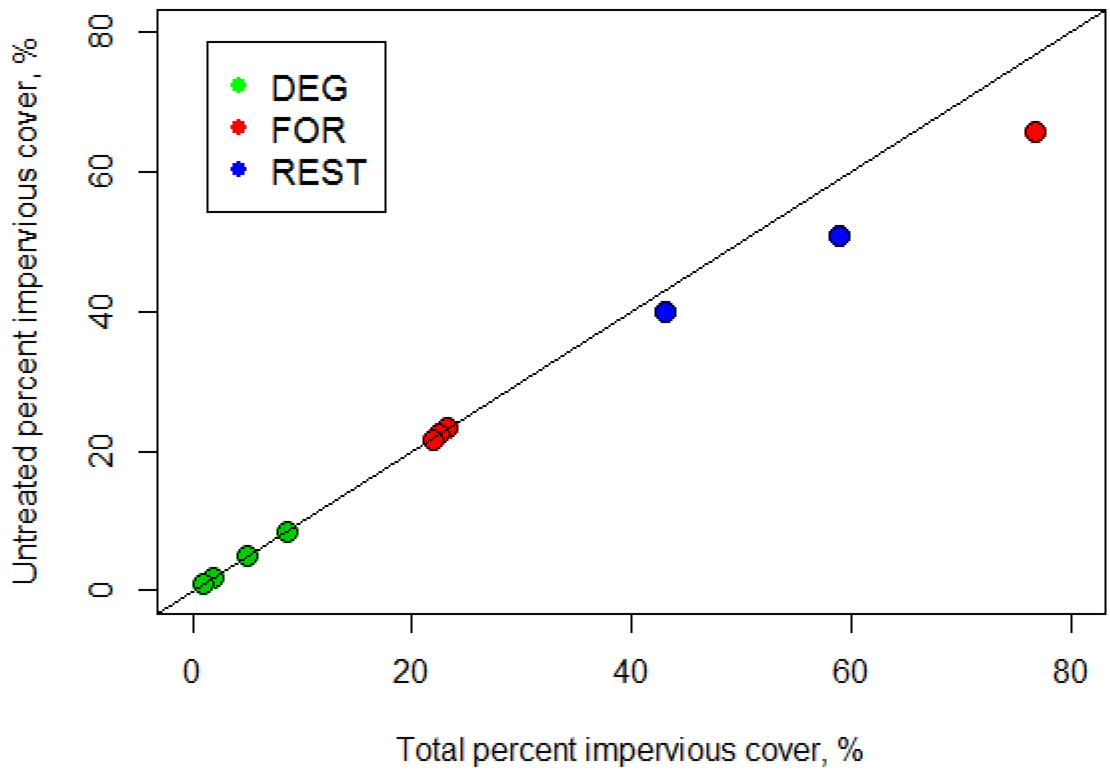

Figure SI-1. 1: Total percent impervious cover vs untreated percent impervious cover for the 11 watersheds, after accounting for upland BMPs. Black line denotes a 1:1 relationship.

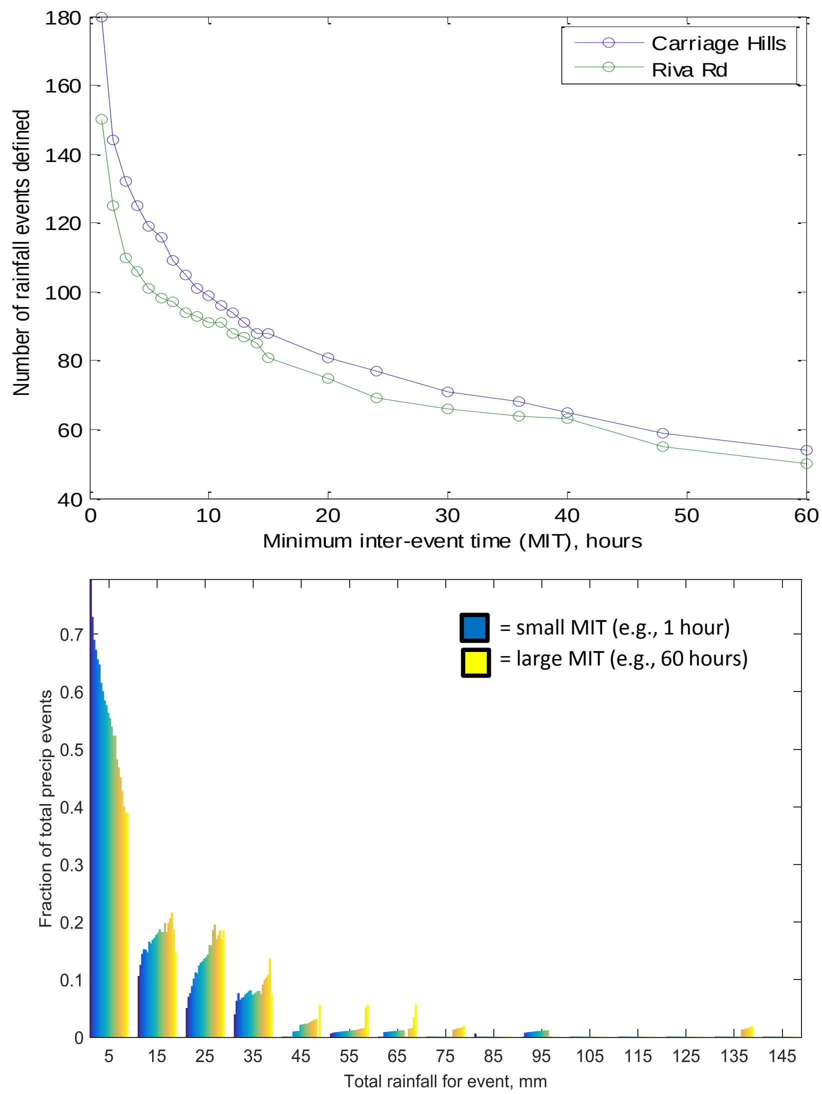

Figure SI-1.2: Effect of variable minimum inter-event time (MIT) on the characteristics of the population of events defined by the MIT. A: Number of rainfall events defined for the two rain gages vs length of MIT, hrs. B: Frequency histograms of rainfall events for variable MITs.

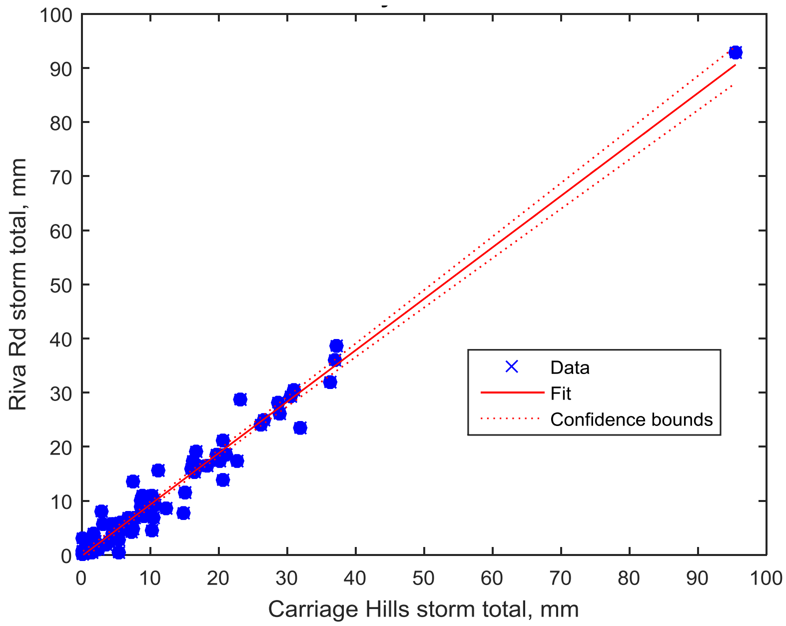

Figure SI-1.3: Rainfall totals for the final 81 events defined by the 5-hour MIT for the Carriage Hills and Riva Rd rain gages. $R^2 = 0.97$; $p < 0.001$.

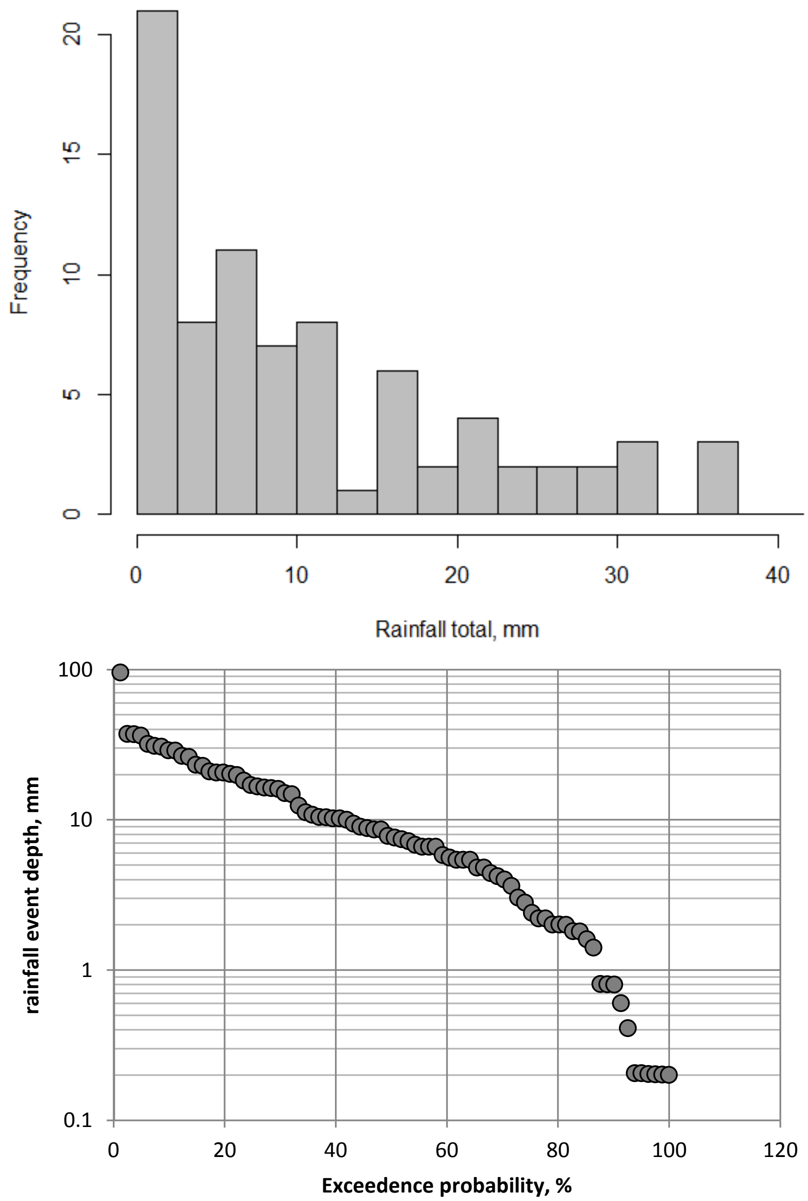

Figure SI-1.4: Rainfall characteristics of the rainfall events used for the hydrometric analysis in the study. (A) Frequency histogram showing the distribution of different sized rainfall events. Note that the x-axis is not fully extended to show the largest rainfall event during the POR (95 mm, August 13, 2014); (B) An exceedance probability plot for the 81 rainfall events according to their size.

Chapter 2: Downstream changes in water quality and insect community composition from urbanization and infiltration-based watershed restoration

Introduction

Headwater streams are vital for sustaining the health and integrity of downstream aquatic ecosystems (Alexander and others 2007; Clarke and others 2008; Freeman and others 2007). They are, however, at high risk of degradation from development. The detrimental effects of urbanization on aquatic ecosystems are a global phenomenon (Cuffney and others 2010; Kim and others 2016; Paul and Meyer 2001; Walsh and others 2005b), with corresponding losses of sensitive benthic organisms occurring at remarkably low levels of urbanization (Hilderbrand and others 2010; King and others 2011). The drivers of species loss from urbanization are numerous: Altered hydrology from unmanaged stormwater runoff modifies a stream's flow regime (Bunn and Arthington 2002; Utz and Hilderbrand 2011), and degrades physical stream habitat (Booth and Jackson 1997; Vietz and others 2016). Urban landscapes act as pollutant sources, including fertilizers (Groffman and others 2004), trace heavy metals (Gobel and others 2007), de-icing chemicals (Corsi and others 2015), and heat (Somers and others 2016). Storm sewer networks facilitate the delivery of these contaminants to stream ecosystems (Leopold and others 1964; Paul and Meyer 2001).

To ameliorate these effects, resource managers have implemented a wide variety of management interventions. Conventional stormwater management (SWM), focused on flood mitigation, has been insufficient at reducing stormflow volumes and restoring flow patterns in urban streams (Burns and others 2012). In-channel stream restoration is a popular management intervention in the U.S., Europe, and beyond (Bernhardt and Palmer 2007; Nakamura and others

2006; Palmer and others 2014b; Spanhoff and Arle 2007; Woolsey and others 2007). In the eastern U.S., stream restoration often entails bank stabilization and channel re-configuration (Castillo and others 2016; Laub and others 2013; Thompson and others 2016). Although these practices can potentially to increase stream habitat heterogeneity and locally suppress sediment erosion rates, most projects fail to improve biodiversity because of persistent stressors (e.g. altered flow regimes and poor water quality) not addressed by the intervention itself (Booth and others 2016; Louhi and others 2011; Palmer and others 2010; Violin and others 2011). Techniques for managing stormwater have advanced in recent years, and now emphasize the infiltration, rather than temporary retention, of stormwater runoff (Bhaskar and others 2016b; Jarden and others 2016; MDE 2009). Infiltration-based, rather than retention-based, SWM systems may have greater potential than previous interventions for recovering urban stream biodiversity because infiltration enhancement could improve stream baseflow and decrease hydrographic peaks, thus improving hydrological processes affected by urbanization (Bernhardt and Palmer 2011; Bhaskar and others 2016b; Jarden and others 2016; Walsh and others 2009).

In this study, we ask if a relatively new infiltration-based SWM approach growing in use is effective in mitigating impacts on the stream benthic community. Regenerative stormwater conveyances (RSCs) are designed to serve as infiltration "hotspots" in the urban landscape (Flores and others 2009; Palmer and others 2014a). Placed between storm sewer outfalls and the perennial stream network, RSCs may disconnect upland urban runoff networks from downstream stream networks, and thus may mitigate these watershed-scale stressors. If so, the downstream stream reaches could then be re-colonized by a more diverse benthic community. While previous work on RSCs has documented mixed results for naturalizing streamflow regimes and small or no reductions in reducing nutrient and sediment loads (Filoso and Palmer 2011; Filoso and others 2015; Palmer and others 2014a), the response of benthic organisms to these practices has not yet been investigated. This study documents benthic response and identifies watershed-scale stressors may or may not be mitigated through the implementation of this SWM design.

In Chapter 1, I examined 11 small catchments along an urbanization-restoration gradient to quantify changes in watershed hydrological function and the resulting downstream flow regime from urbanization and subsequent RSC implementation. Here I focus on patterns in water quality and aquatic insect community composition across the same urbanization-restoration gradient. I ask: 1) How does the aquatic insect community vary across the gradient? 2) How do water quality conditions vary across the gradient? And, 3) Do any of these water quality factors (or hydrological metrics previous quantified) explain differences among the insect communities? I use the aquatic insect community as the lens through which I examine the ecological effects of RSCs because aquatic organisms are integrators of the stream ecosystem health (Karr 1999).

Methods

Study area

I conducted this work in 11 first-order streams in the Coastal Plain, near Annapolis, MD, USA (Figure 2.1). The western Maryland Coastal Plain is characterized by gentle topography underlain by unconsolidated material with bands of alternating high and low permeability sediments (Bachman and others 1998). Urban development is primarily on hilltops; hillslopes bordering stream valleys can be fairly steep, which has allowed riparian forests to remain fairly intact even in urbanized regions. Coastal Plain streams are predominantly sand-bottom streams with no bedrock grade controls. Headwater stream channels have relatively low gradients (1-

3%), with coarse woody debris (tree falls, large limbs, etc.) providing much of the geomorphic channel complexity (Laub and others 2012). Soils in this region range from well drained to poorly drained, depending on the topographic position and depth to a low-permeability stratigraphic lens. All sites within the study are located within a 5-mile radius of each other, and drain to either the Severn or South Rivers. The study sites span a gradient of urbanization and subsequent restoration implementation, and include four forested streams, four urban-degraded streams, and three urban-restored streams (Table 2.1). The urban catchments have greater than 20% impervious cover, while the forested catchments have less than 10% impervious cover. The urban-restored catchments have been implemented with RSC systems, the infiltration-based stormwater control measure, which is the focus of this study.

Aquatic insect sampling

The aquatic insect community at 10 of the 11 study streams was sampled between April 21, 2014 and April 28, 2014. The remaining site, CH2 (an urban-degraded catchment) was not sampled due to a severe disturbance to the channel from a failed check dam in its watershed in the week prior to the sampling campaign. However, I report water quality monitoring results from CH2 catchment in this study. With the exception of sampling a shorter stream reach (50 m in length), I followed the Maryland Biological Stream Survey (MBSS) protocols to allow for comparison of results to extensive regional monitoring efforts (Klauda and others 1998). All water quality and aquatic community sampling was conducted in the reach downstream of the RSCs. I sampled 20ft^2 of habitat at each site, per the MBSS suggested methods. The goal of this methodology is to assess the community composition and relative abundance and diversity in favorable habitat types, so specific habitat types were targeted during the sampling, rather than using a gridded or random sampling design (DNR 2010). Habitats were sampled using a D-net with a 500-µm mesh opening, and included riffles, leaf packs, root wads, and coarse woody debris. The streambed was manually disturbed, allowing organisms to flow into the D-net. Habitat samples were composited into a single sample per site; organisms were extracted with forceps and fixed in 95% ethanol. Organisms were identified to Family, and standard aquatic community descriptors were calculated for each of the 10 insect communities, including taxon richness, Shannon diversity, Simpson's dominance, overall insect abundance, and number and percent of sensitive taxa (e.g. orders Ephemeroptera, Plecoptera, and Trichoptera; EPT).

Water quality monitoring and sampling

I conducted monthly, discrete baseflow water quality sampling at the midpoint of each 50-m benthic sampling reach from March 2014 and April 2015. Sampling took place on days that were preceded by at least a following a 24-hour or longer rain-free period. In-situ water quality parameters (water temperature, specific conductance, pH, and dissolved oxygen) were measured with a YSI multi-meter probe. A single, 300-ml water sample was collected from each reach for analysis of major anions, including chloride. Samples were chilled immediately after collection, and filtered and frozen within a 24-hour period after sampling. Chloride concentrations for each baseflow sample were analyzed using a Dionex 1000 ion chromatograph. I monitored continuous stream stage and water temperature for a 1-year period (June 2014-June 2015) at the midpoint of each 50-m benthic sampling reach using Hobo water level/ temperature loggers (models U20 and U20L). The temperature sensors for both models have a documented accuracy of 0.44 ^{o}C and a 0.1 ^{o}C resolution (http://www.onsetcomp.com/products/data-loggers/u20l-04). The loggers were housed in a perforated 1-inch PCV pipe embedded into the streambed, and their sensors places at the streambed elevation to ensure submergence throughout the monitoring period. Stream temperature and stage were logged at 3-minute intervals from June 2014 to August 2014, and at 2-minute intervals from August 2014 to June 2015.

Characterizing the stream environment

Six metrics were used to describe environmental conditions across the study streams; each describes a stressor known to affect aquatic organisms, such as flow patterns, water quality, and stream temperature (Bunn and Arthington 2002; Hester and Doyle 2011; Lenat and Crawford 1994). Two hydrological metrics were previously identified that characterized catchment responses during low flow and stormflow [Chapter 1]. The first metric is *mean annual baseflow*, the average of monthly discharge measurements taken at each site normalized for catchment area. To capture changes in stormflow patterns, I used the percentage of rainfall events that generated an observable runoff response during the 1-year stage monitoring period. This metric describes the *runoff frequency* each stream experiences due to incoming stormwater runoff. These two hydrological metrics captured the greatest variability among the study streams in a previous study [Chapter 1].

Dissolved oxygen (DO) and *conductivity* in streams are potential water quality stressors (Merritt and others 2008). Many aquatic organisms are sensitive to dissolved oxygen concentrations, which can be depressed by high water temperatures and/or chemical or biological oxygen demands (Connolly and others 2004). Likewise, many aquatic taxa have evolved to live in specific ranges of conductivity, thereby making them sensitive to anthrophonic changes in stream conductivity (Azrina and others 2006; Garcia-Criado and others 1999). Because aquatic organisms are most sensitive to extreme values in water quality, I used the mean of the lowest three baseflow DO measurements, and the mean of the three highest conductivity measurements taken during the monitoring period to reflect extreme changes in water quality that could affect aquatic insect community composition.

Finally, we used *maximum daily summer stream temperatures (Tmax)* and stream *temperature surges during runoff events (Tsurge)* to describe changes in the stream thermal regime across the study sites. Stream temperatures have increased from both urbanization and recent climate change (Kaushal and others 2010), and high water temperatures in the summer time can induce stress in intolerant benthic organisms (Winterbottom and others 1997). Abrupt increases in stream temperature in urban streams have been documented elsewhere (Nelson and Palmer 2007) and have the potential to initiate drift of intolerant organisms (Hester and Doyle 2011). Temperature surges during runoff events were quantified only for summer events (June 1 to September 30) during which an observable runoff response occurred. For those events, pre-event temperature (measured as the mean temperature of the stream 1-hour prior to the start of the event) was subtracted from the maximum stream temperature observed during the runoff event. The rainfall events analyzed were short in duration and often occurred in the later-afternoon or evening hours, so daily fluctuations in stream temperature was presumed to have no effect on the Tsurge metric.

Regional distributions of aquatic insects

I accessed benthic community data for Maryland Western shore Coastal Plain streams from the MBSS program, which included samples taken from tributaries in the Severn, South, Magothy, West and Lower Pautuxent river basins. To understand how water quality potentially controls taxa distributions regionally, I also requested water chemistry information taken from these sites. I focus this analysis on conductivity alone because dissolved oxygen was not available in the MBSS datasets. To compare our results to the greater Coastal Plain region, I used an average baseflow conductivity value during typical spring season MBSS sampling months (March-May 2014). Finally, I used both insect abundance and conductivity to identify conductivity limitations for sensitive taxa across my sites and within the greater region.

Data analysis

I used analysis of variance (ANOVA) or analysis of covariance (ANCOVA) to test for the differences in insect community metrics and the stream environmental metrics (e.g., the response variables) among the three watershed types (forest, urban-degraded, and urban-restored; Table 2.1). I initially included percent impervious cover as a covariate in each model, and only removed it when the response variable was clearly not correlated with it (covariate p-value > 0.1). For those response variables, ANOVA was performed to test for overall differences among the watershed types. Log transformation of the explanatory variable (impervious cover) was performed when model residuals indicated the need. Tukey's Honestly Significant Difference test (HSD) was used to assess differences among individual watershed types (e.g, urban-degraded vs urban-restored). To examine overall insect community similarity, a non-metric, multidimensional scaling (NMDS) analysis was conducted using the vegan package in R. I used Bray-Curtis distances to quantify community dissimilarity, because this distance metric works well with zero data (e.g., absences) and is often used in describing ecological data (Leslie and others 2012; Stranko and others 2012; Violin and others 2011). I fitted the environmental metrics to the NMDS ordination and applied a permutation test to determine which environmental metrics correlated best with the NMDS ordination.

Results

Aquatic insect community metrics

I characterized the aquatic insect communities in the 10 streams using metrics to describe community structure (Table 2.2; Figure 2.2). Forested streams, on average, had significantly greater insect abundance, overall taxa richness, Shannon diversity, number of EPT taxa, and percent EPT taxa than the urban stream communities. For example, we identified 13 different taxa in the forested streams on average, and only 4 different taxa in the urban-degraded and urban-restored streams, respectively (Table 2.2). I found no significant differences in insect abundance, overall taxa richness, Shannon diversity, number of EPT taxa, and percent EPT taxa between the urban-degraded streams and urban-restored streams. Forested streams also had a lower Simpson's dominance than the urban degraded streams, indicating a greater level of evenness among forested stream communities. Aquatic insect abundance, overall taxa richness, and Shannon diversity all declined with impervious cover, while Simpson's dominance increased with greater impervious cover (Figure 2.2; Table 2.2).

Forested stream insect communities were composed of 41% EPT taxa on average, and included *Leptophlebiidae* (order: Ephemeroptera), *Leuctriidae* (Order: Plecoptera), and *Limnephilidae* (Order: Trichoptera). *Corydalidae* and *Sialidae* (Order: Megaloptera) larvae were found only in the forested streams as well. The only sensitive insect found in an urban watershed was *Polycentropodidae* (Order: Trichoptera). Two individuals of this family were found in CH1, an urban-restored stream with moderate impervious cover (22%). *Polycentropodidae* was also found in three of the four forested streams (SW1, SW3, and SALT3). No EPT taxa were found in sites with 25% or greater impervious cover. Urban stream insect communities (at both degraded and restored sites) were instead dominated by *Dytiscidae* and *Hydrophilidae* (Order: Coleoptera), *Tipulidae* and *Ptychopteridae* (Order: Diptera), and *Coenagrionidae* (Order: Odonata).

Stream environment metrics

As reported earlier [Chapter 1], urbanization significantly altered both storm runoff responses and baseflow discharge across the study streams. Mean annual baseflow per basin area in forested streams was twice high as in urban streams (forest streams group average = 0.10 l/s-ha; Figure 2.3B, Table 2.3), and there was no difference in mean annual baseflow between the urban-degraded streams and urban-restored streams (group means = 0.04 and 0.05 l/s-ha, respectively). During the monitoring period, forested streams responded to far fewer rainfall events (51% on average) than either the urban-degraded or urban-restored streams (75% and 63%, respectively). I observed a large range in runoff frequency among the three urban-restored streams. One urban-restored stream (CH1) exhibited a lower runoff frequency than any other site in the study (responding to only 33% of the rainfall events), whereas the remaining two urban restored streams had much higher runoff frequencies (RR =73%; SALT1 = 81%; Figure 2.3A). Overall, runoff frequency increased with increasing impervious cover ($p < 0.005$).

Maximum daily summer temperatures were lowest in the forested streams (average 18.4 ^{o}C) and greatest in the urban-restored streams (average 20.5 ^{o}C; Table 2.3). In general, summer daily maximum temperature increased with increasing impervious cover ($p < 0.002$; Figure 2.3C), and was greatest in CC (urban-degraded stream; 22.3 ^{o}C on average) and SALT1 (urban-restored stream; 21.2^{o}C on average). Temperature surges during summer runoff events were greatest in the urban-degraded streams (average = 2.1 ^{o}C), and smallest in the forested streams (average = 0.4 ^{o}C; Table 2.3). The greatest average runoff temperature surges were observed in urban-degraded streams with moderate impervious cover: ML (2.8 ^{o}C) and SALT2 (2.1 ^{o}C; Figure 2.3D). The three highest temperature surges for an individual summer runoff event were recorded at SALT2 (6.0 ^{o}C), ML (5.5 ^{o}C), and RR (urban-restored stream; 4.0 ^{o}C).

Maximum conductivity concentrations were an order of magnitude lower in the forested streams (average = 165 uS/cm) than in the urban streams, and I observed no difference between the urban-degraded and urban-restored streams (group averages = 1,481 and 1,224 uS/cm, respectively; Table 2.3). During my synoptic sampling, I observed the highest conductivity at CC, an urban-degraded stream (maximum conductivity = 8,187 uS/cm) and at RR, an urban-restored stream (maximum conductivity = 2,380 uS/cm), In general, maximum conductivity was highly correlated with impervious cover ($p < 0.001$; Figure 2.3E). Minimum dissolved oxygen concentrations were generally greatest in the forested streams (group average = 5.9 mg/l), and were significantly lower in the urban-restored streams (group average = 3.8 mg/l; Table 2.3). I observed the lowest baseflow DO concentrations in RR (urban-restored stream; average = 2.1 mg/l) and ML (urban-degraded stream; average = 2.2 mg/l; Figure 2.3F).

Multivariate analysis of the insect communities and stream environmental metrics

The non-metric multidimensional scaling (NMDS) ordination for all 10 streams defined a stable, 2-dimensional solution with a final stress of 8.19%. Forest streams clustered at the high end of the first NMDS axis, indicating similar community structure (Figure 2. 4A). There was no clear clustering among the other two urban watershed groups (urban-degraded or urban-restored). Rather, site coordinates along the first NMDS axis were highly correlated with impervious cover (R^2= 0.98, $p < 0.001$), such that urban streams with the greatest impervious cover exhibited the greatest dissimilarity from the forested stream communities. The permutation test indicated that maximum conductivity, runoff frequency, and daily summer maximum temperatures significantly correlated with the overall NMDS ordination (R^2 values = 0.90, 0.64 and 0.69; p-values < 0.003, < 0.03, and < 0.02 respectively). NMDS ordination conducted with streams whose watersheds have less than 30% impervious cover defined a stable, 2-dimensional solution with a final stress of 4.54% (n = 7 sites). Clustering occurred between two of the forested sites (SW2 and SW3; Figure 2.4B). Neither NMDS axis correlated with impervious cover, but the second NMDS axis did correlate with catchment area (larger sites at the lower end; $R^2 = 0.67$, $p < 0.02$). A permutation test identified minimum dissolved oxygen as the only metric to significantly correlate with this NMDS ordination ($R^2 = 0.71$, $p > 0.06$).

Local and regional patterns in taxa abundance and stream water conductivity

Across the study streams, stream conductivity varied by treatment and by season; it ranged from 21 uS/cm (SW1; forested stream) and to 10,430 uS/cm (CC, urban-degraded stream in winter; Figure 2.5). Forested streams exhibited the least variability in seasonal conductivity; stark increases in stream conductivity were observed in most urban streams during winter and early spring. Regionally, conductivity measured during the spring sampling months ranged from as low as 40 uS/cm to as high as 8,500 uS/cm. I examined regional abundance-conductivity relationships for four taxa considered to be sensitive to conductivity: *Leuctriidae* (Order: Plecoptera), *Leptophlebiidae* (Order: Ephemeroptera), *Limnephilidae* (Order: Trichoptera), and *Polycentropodidae* (Order: Trichoptera). Throughout the region, neither *Leuctriidae* nor *Leptophlebiidae* have been documented where conductivity values exceeded ~500 uS/cm, suggesting a conductivity limit (Figure 2.6). Both taxa seemed to favor conditions where conductivity was less than 200 uS/cm. *Limnephilidae and Polycentropodidae* were both generally absent above 500 uS/cm as well, but seemed to tolerate slightly higher conductivities than *Leuctriidae* and *Leptophlebiidae* (Figure 2.6).

Discussion

I documented severe reductions in the abundance and diversity of aquatic insects across the urbanization gradient, consistent with many similar studies conducted within the mid-Atlantic region (Cuffney and others 2010; King and others 2011; Moore and Palmer 2005; Utz and others 2009) and across North America (Cuffney and others 2010; Wallace and others 2013). I found no difference in insect diversity between urban-degraded streams and urban streams whose watersheds were restored with infiltration-based SWM. This lack of restoration effectiveness for increasing biodiversity aligns with other studies examining the effects of in-stream restoration practices (Ernst and others 2012; Smucker and Detenbeck 2014; Stranko and others 2012), but somewhat contrasts with the findings of Smucker and others (2014), who found that out-of-channel stormwater management can improve stream biodiversity. The unique design of these restoration projects makes it difficult to clearly categorize them as either an in-stream practice or out-of-channel practice, especially since they are being implemented in both settings.

Regardless, all these studies highlight that watershed-scale stressors not addressed by the management or restoration practice may inhibit the recovery of stream biodiversity. Since urbanization affects the ecosystem in many ways, identifying the primary drivers of biodiversity loss in urban stream ecosystems is difficult (Paul and Meyer 2001). However, poor water quality has been suggested as a major factor affecting stream organisms. For example, many freshwater benthic taxa are intolerant to elevated stream conductivity (Cormier and others 2013; Pond 2010). A recent study conducted in West Virginia streams defined the 95% extirpation limits for

Leptophlebiidae as 200-500 uS/cm, depending on the family (EPA 2011) and my regional analysis of abundance vs conductivity confirms this limitation (Figure 2.6). Road salt applications have been linked to increased salinization of urban streams in temperate cities (Corsi and others 2015; Kaushal and others 2005), and recent studies have revealed shifts in community structure as a result of changes in conductivity and chloride (Wallace and Biastoch 2016). Indeed, stream conductivity was highly correlated with chloride concentrations at the study sites ($R^2 = 0.94$, $p < 0.001$; Figure SI-2.2), suggesting road salt is contributing to persistent water quality issues.

This study showed stream conductivity values in excess of 500 uS/cm for long periods of time (e.g. weeks) in many of the urban streams, regardless of restoration status (Figure 2.5). The lack of a difference between urban-restored and urban-degraded streams is likely due to both the type of restoration implemented and the solubility of chloride. The RSC projects are emplaced upstream or adjacent to the streams, which means that infiltrated, chloride-rich runoff will continue to drain to the first-order streams. Previous hydrological analyses indicated that the two restored watersheds with the highest impervious cover (SALT1 and RR) did not show significant restoration of hydrological flow regimes [Chapter 1; Figure 2.3A and 2.3B]. Although the RSC in the other urban-restored stream (CH1) was effectively infiltrating urban runoff, this urban-restored stream still had higher conductivity than forested reference streams (Figure 2.5). Infiltration-based stormwater management may actually increase the loading of chloride into the shallow groundwater in winter months. This could potentially shift the chloride "disturbance" from a pulse to a press regime, with chronically high chloride concentrations across all months, as seen in some recent floodplain studies (Cooper and others 2014; Ledford and others 2016). Moreover, de-icing salts may also compromise the ability of porous media used in infiltration-based SWM to retain nutrients and metals (Kakuturu and Clark 2015). Due to the solubility of chloride and its behavior as a conservative tracer, chloride can only be managed by limiting roadway applications or using deicing alternatives (Corsi and others 2015).

Urbanization also modifies the stream thermal regime, and our findings are consistent with other studies documenting this (Nelson and Palmer 2007; Somers and others 2016). Shifts in benthic communities in cold water environments have been attributed to temperature changes from urbanization (Wang and Kanehl 2003). I found daily maximum stream temperatures were correlated with the overall NMDS plot, likely because of the strong impervious gradient expressed through the benthic community (Figure 2.4A). The effects of restoration on stream temperature were complex; for example, daily maximum stream temperatures were the highest in the urban-restored streams. This could be due to the loss of riparian vegetation near or along the stream channel heads during restoration implementation, a mechanism documented by others (Sudduth and others 2011). The urban-degraded streams in the study area have fairly intact riparian cover, which could provide more shade than the more recently disturbed urban-restored streams. Somers and others (2013) found that, for explaining patterns in daily maximum temperature, reach-scale characteristics were better predictors than watershed-scale characteristics, which supports this hypothesis.

However, elevated baseflow temperatures in the urban-restored streams could also be due to the retention of warm water infiltrated during summer storms by the RSCs. These streams are groundwater-fed, making them more susceptible to changes in temperature of baseflow sources (Herb and others 2008). Other factors, such as the aspect of the catchment (and therefore incident radiation), may also play a role in defining the baseflow thermal regime of headwater streams (Dick and others 2015). Although further investigation is needed to understand the mechanisms driving these patterns, these results clearly demonstrate that the restorations do not mitigate the impacts of urbanization on baseflow stream temperature.

In contrast with their effects on baseflow temperature, the restoration projects, on average, seemed to mitigate temperature surges during runoff events (Figure 2.3D). Infiltration of warm stormwater into the seepage beds of the restoration structures could raise local groundwater tables, thereby increasing groundwater outflow from depth in these systems. This mixing of new stormwater runoff with older, retained water may reduce the temperature of the summer stormwater runoff overall. Other stormwater SCMs that promote infiltration have been shown to reduce peak thermal loads from impervious surfaces, including vegetated filter strips (Winston and others 2011) and bioretention basins (Long and Dymond 2014). Thermal load reductions also occur during events when stormwater runoff is completely infiltrated within the restoration, which is likely occurring only at one of the urban-restored streams (CH1). This analysis did not include these types of rainfall events, but is instead expressed implicitly in the runoff frequency metric (Figure 2.3A). CH1 responded to far fewer rainfall events across the year, including events in the summer season that conveyed hot runoff into the other urban streams in the study. In fact, during the event in which the greatest temperature surge was observed (6^{o}C in SALT2) the runoff generated from that event in the CH1 catchment was completely infiltrated by the restoration.

Finally, I also documented an unintended water quality consequence of restoration. I observed the lowest minimum DO levels in the urban-restored streams, with all three restored streams consistently at or below the state of Maryland numeric criteria for protecting aquatic organisms (5 mg/l). Low dissolved oxygen has been documented in several other streams in the mid-Atlantic region that have been implemented with the same restoration design (Williams and

Filoso in review), and may be related to the construction of the restoration itself. One of the construction materials used to build RSCs in the mid-Atlantic is iron ferrocite, which is sandstone with iron (Fe) matrix. The addition of these materials to the watershed may increase the soluble Fe load downstream and increase the chemical or biological oxygen demand in the downstream channel, depending on several factors, including the availability of dissolved organic carbon, nutrients, and bacterial community composition (Volkmar and Dahlgren 2006; Williams and others 2016). Indeed, DO saturation was consistently lower in the urban-restored streams as well, suggesting there are factors other than temperature driving these patterns (Figure SI-2.1). Dissolved oxygen was significantly correlated with the NMDS axes for sites with less than 30% impervious cover, suggesting it may be a factor controlling insect community composition at moderate levels of impervious cover (Figure 2.4B).

In addition to poor water quality, the general failure of these restorations to mitigate the altered hydrology of their watersheds may also contribute to low stream biodiversity. These restorations are designed to serve as infiltration hotspots to disconnect upland sources of stormwater runoff, reducing the runoff frequency to the streams channels. However, only one of three restorations in this study effectively reduced this hydrological connectivity (Figure 2.3A). Stream flow regimes, including the frequency of high flows, shape the aquatic community in general (Bunn and Arthington 2002). Previous work at these study sites demonstrated that restoration did not modulate the effects of urbanization on the timing, duration, and rate-of-change of storm responses [Chapter 1]. Elevated peak flows can lead to streambed instability, increased sedimentation of the streambed, and lowered geomorphic complexity, among other things (Booth 2005; Paul and Meyer 2001; Utz and Hilderbrand 2011). Recent studies show that habitat restorations may not be effectively recovering biota because of the persistent watershed-scale stressors of altered flow and water quality (Ernst and others 2012; Palmer and others 2010), which were clearly documented in this study. Dispersal limitations may be another factor potentially affecting the recovery of the biological community in this region. Smith and others (2015) found that urbanization can affect local processes, such as habitat suitability, as well as regional processes, such as dispersal from of the metacommunity. Given that these are headwater steams, dispersal of sensitive taxa back into these streams is more limited since they have no upstream source of organisms. Finally, persistently low baseflow in the restored streams may lead to greater intermittency in these streams, which has been linked to loss of EPT taxa (King and others 2016).

I observed one sensitive insect, *Polycentropodidae*, at the least urbanized of the three urban-restored streams (CH1). Although in the immediate region, this insect seemed to be limited to streams with conductivity < 500 uS/cm, in West Virginia it had an observed 95% extirpation rate at 1,400 uS/cm (EPA 2011), suggesting that it may be able to tolerate higher conductivity values. The spatial distribution of this organism compared to another sensitive taxa (*Leptophlebiidae*) illustrates the importance of dispersal distances for recolonizing restored stream reaches. MBSS documented several *Polycentropodidae* individuals at a site less than 1 km downstream of CH1's outlet (MBSS site ID: SEVE-110-R-2008; Figure 2.7), as well as at several other sites nearby, indicating the presence of a viable dispersal pool for this particular taxa. In contrast, *Leptophlebiidae* was only found in our three northern forested sites, and a handful of sites well north of the study area (Figure 2.7).

In general, I found that these restoration practices are insufficient for fully mitigating stormwater runoff, improving water quality, and recovering biodiversity, echoing the conclusions of other studies (Smucker and Detenbeck 2014; Stranko and others 2012). The lack of any effective change in hydrology and water quality from restoration efforts emphasizes the difficulty in reversing the detrimental impacts of urbanization on stream ecosystems, especially since past land use has been shown to control present day insect communities (Brown and others 2009; Harding and others 1998). Continuing to innovate and improve stormwater management is critical, although infiltration-based strategies implemented throughout the watershed, rather than at the channel head only, are likely to be more effective. In addition, there should be a high priority for the preservation of forested, minimally-disturbed streams, to serve as both dispersal sources of high quality taxa to downstream reaches, and to connect patches of habitat to one another (Palmer and others 1997; Thieme and others 2016). Watershed-scale, process-based restoration should be emphasized, and only after the alleviation of these stressors has been documented, should in-stream habitat restoration be conducted if the goal is improved biodiversity (Palmer et al., 2010). Finally, continued improvements for managing stormwater runoff should continue in heavily urbanized streams to reduce downstream impacts, although there should not be an expectation of recoverable biodiversity in these reaches.

Conclusions

I observed persistently poor water quality and altered hydrology as a result of urbanization, and infiltration-based watershed restoration did little to mitigate these watershed stressors. As a result, aquatic insect communities in the urban streams (regardless of being restored) were less diverse and had far fewer organisms than in undisturbed, forested streams. The restorations were unable to recover the biological community for several reasons: 1) they, in large part, did not perform their hydrologic function to infiltrate runoff and modify flow patterns; 2) the water quality issues identified in the downstream reaches either cannot be explicitly addressed by this restoration design (in the case of chloride) or has been exacerbated by the implementation of the restoration (in the case of dissolved oxygen and daily maximum temperatures); and 3) poor dispersal opportunities may prohibit rare taxa from re-colonizing restored stream. This study underscores the importance of identifying the key sources of stream ecosystem degradation to ensure the restoration activity will actually address it. Finally, I emphasize the need for prioritizing the preservation of undisturbed streams in the region to support a diverse metacommunity, which serves as an investment into the future success of restoration efforts in the region.

Tables and Figures

Table 2.1: Site characteristics (watershed type, catchment area and untreated impervious cover) for the 11 streams used for the study, in order of increasing impervious cover. Note that no biological monitoring occurred at CH2 and therefore is omitted from those analyses.

Site name	Watershed type	Catchment area, ha	Impervious cover, %*
SW3	Forest	6.9	1.0
SW1	Forest	13.1	1.9
SW2	Forest	8.5	4.9
SALT3	Forest	13.9	8.5
SALT2	Urban-degraded	18.6	21.5
CH1	Urban-restored	5.4	22.2
ML	Urban-degraded	8	22.5
CH2	Urban-degraded	5.6	23.2
RR	Urban-restored	11.4	40.0
SALT1	Urban-restored	48.8	50.7
CC	Urban-degraded	33.5	65.9

* Impervious cover provided by AA county

Table 2.2: Group means and standard deviations for a variety of aquatic insect community metrics for the 10 watersheds. Letters indicate group differences as indicated by Tukey's Honestly Significant Difference test ($\alpha = 0.05$).

	Un-adjusted means (SDs) for each watershed type			Covariate p-value	Overall model
Metric	**Forest**	**Urban-degraded**	**Urban-restored**	**$\log_{10}$(% imp)**	**adj. R^2**
Insect abundance	93 (27.5) a	22 (11.9) b	15 (15.5) b	0.0002	0.98***
Taxa richness	13 (2.6) a	4 (1.7) b	4 (2.3) b	0.003	0.95***
Simpson's dominance	32 (11.2) a	68 (24.6) b	47 (9.9) ab	0.10	0.59*
Shannon's diversity	2.1 (0.3) a	0.9 (0.5) b	1.2 (0.3) b	0.06	0.76**
Number EPT taxa	4 (1.3) a	0 (0) b	0.3 (0.6) b	ns	0.81**
Percent EPT taxa	41 (19.1) a	0 (0) b	2.0 (3.5) b	ns	0.71**

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Table 2.3: Results from ANOVA and ANCOVA analysis between the six environmental variables and watershed characteristics (watershed type and percent impervious cover as a covariate). Letters denote mean pair-wise differences among the groups as reported with Tukey's HSD test (α=0.1). Only the 10 study sites with complete environmental and community datasets were included in the analysis.

	Unadjusted means (SDs) for each watershed type			Covariate	
Metric	**Forest**	**Urban-degraded**	**Urban-restored**	**p-value (% imp cover)**	**Model adj. R^2**
Baseflow, l/s-ha	0.10 (0.02) a	0.04 (0.02) b	0.05 (0.03) b	ns	0.71**
Runoff frequency, %	50.6 (8.8) a	75.2 (12.5) b	62.6 (25.7) b	0.02	0.66*
Max conductivity, uS/cm	165 (134) a	1481 (612) b	1224 (1907) b	0.0008	0.86**
Min DO, mg/l	5.9 (0.06) a	4.5 (1.5) ab	3.8 (1.0) b	ns	$0.46^{\#}$
Tmax, ^{o}C	18.4 (1.1) a	19.5 (2.5) ab	20.5 (1.0) b	0.003	0.73**
Tsurge, ^{o}C	0.4 (0.2) a	2.1 (0.8) b	1.1 (0.7) ab	0.07	0.74*

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

$^{\#}$ $p < 0.06$

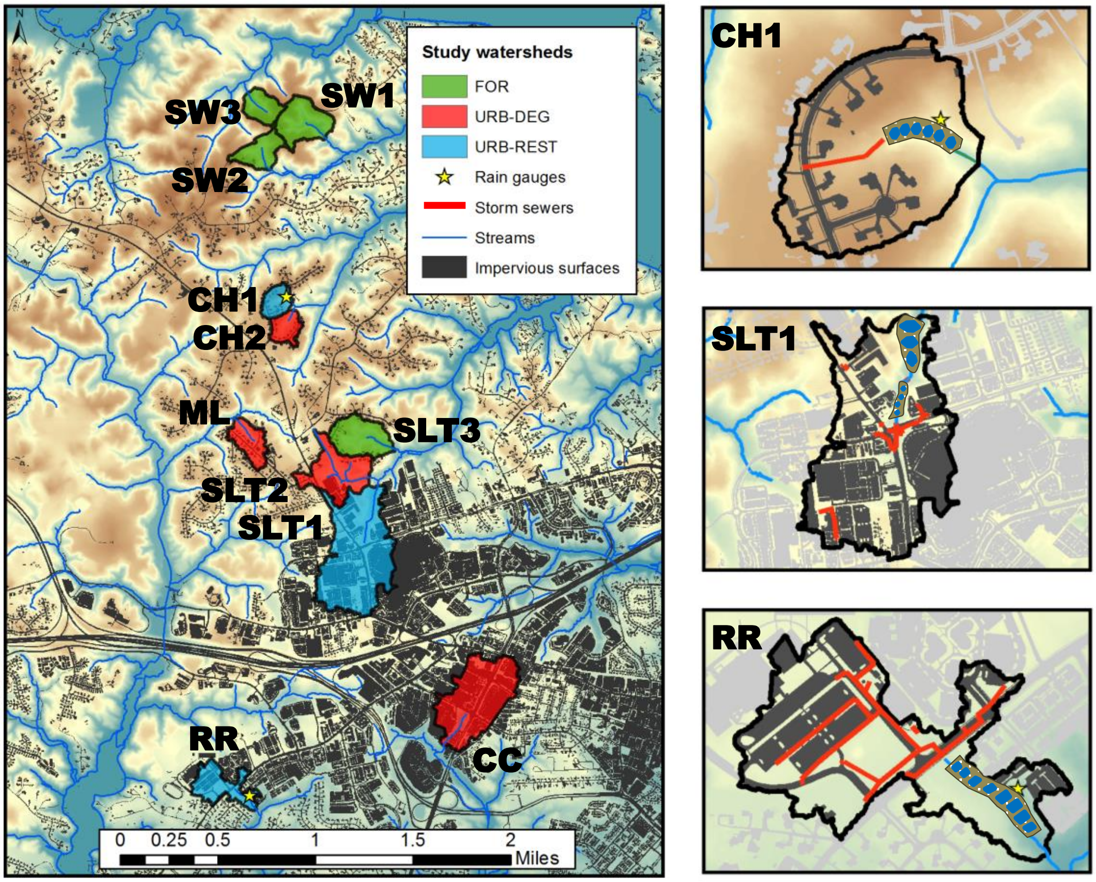

Figure 2.1: (Left) The 11 study watersheds included in the study. (Right) Detailed site maps of the three urban-restored watersheds (CH1, SALT1, and RR). All water quality, hydrological and biological monitoring took place at the watershed outlets, downstream of the RSC structures.

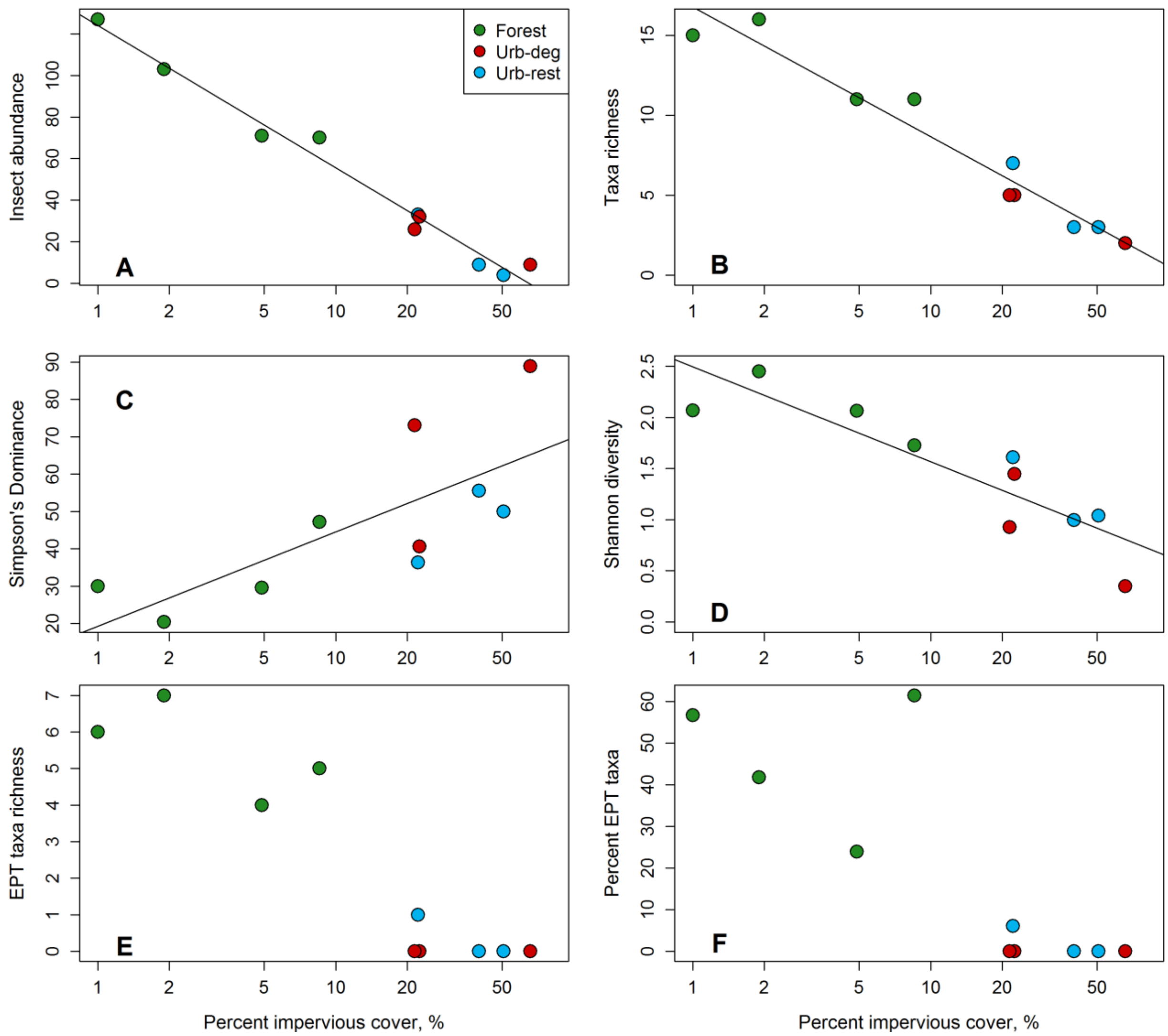

Figure 2.2: Aquatic insect community metrics for the 10 study streams: (A) Insect abundance, (B) Taxa richness (taxa= Family), (C) Simpson's dominance index, (D) Shannon's diversity index, (E) Number of EPT taxa found in sample, and (F) Percent of EPT organisms in the sample.

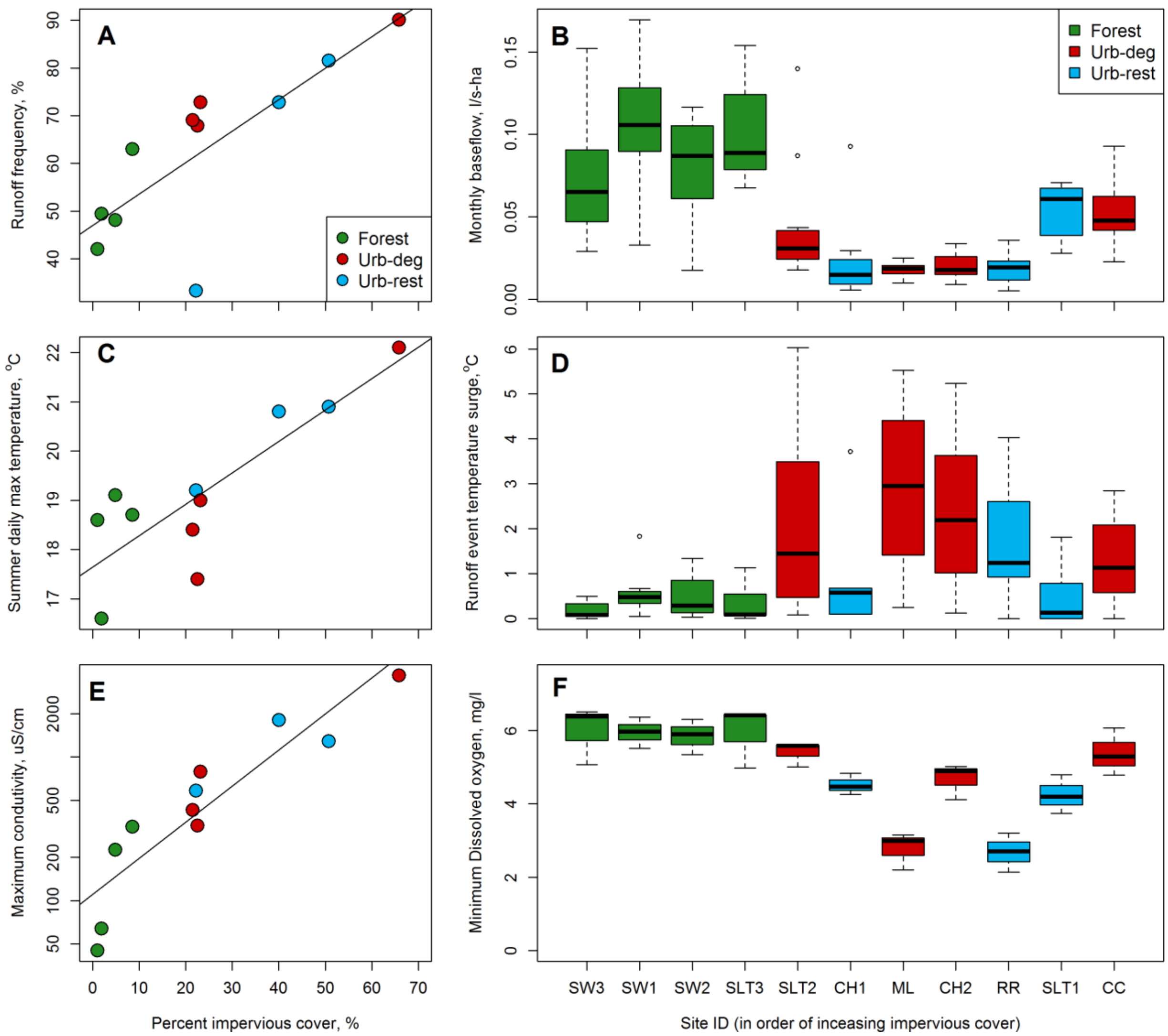

Figure 2.3: Results from the environmental monitoring at the 11study streams (includes CH2): (A) Impervious cover vs. runoff frequency; (B) Boxplot of monthly baseflow (n=12); (C) Impervious cover vs. mean summer daily maximum temperatures, (D) Stream temperature surges during runoff events (n = 9,9,9,15,16,5,13,16,15,18,20); (E) Impervious cover vs. mean of the three highest conductivity measurements during the monitoring period; and (F) Minimum dissolved oxygen concentrations (n=3). In Figures 2.3B, 2.3D, and 2.3F, watersheds are ordered by their impervious cover.

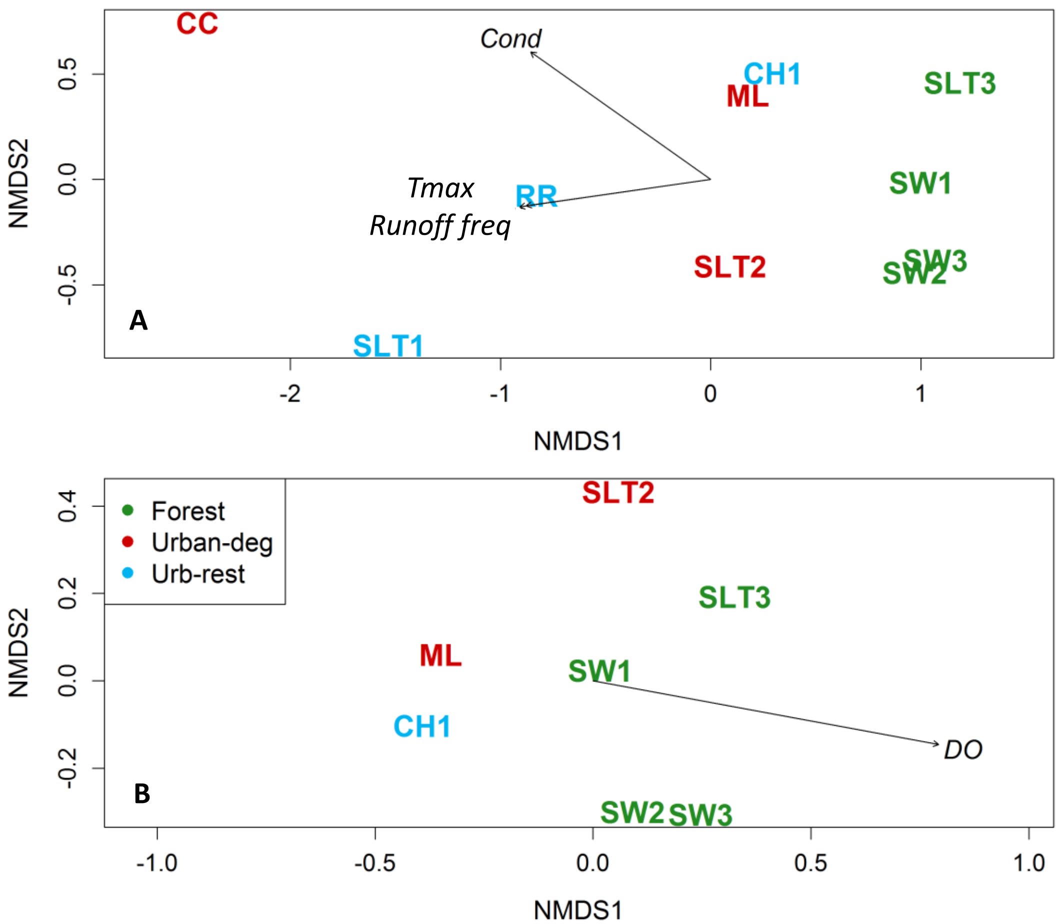

Figure 2.4: (A) NMDS plot with all study streams included in the insect community analysis (n=10) and environmental variables which were significantly correlated with the NMDS axes. (B) NMDS plot with only sites with 30% impervious cover or less included in the analysis (n=7 sites). Arrows indicate a significant correlation between the environmental variables and NMDS axes ($p < 0.10$).

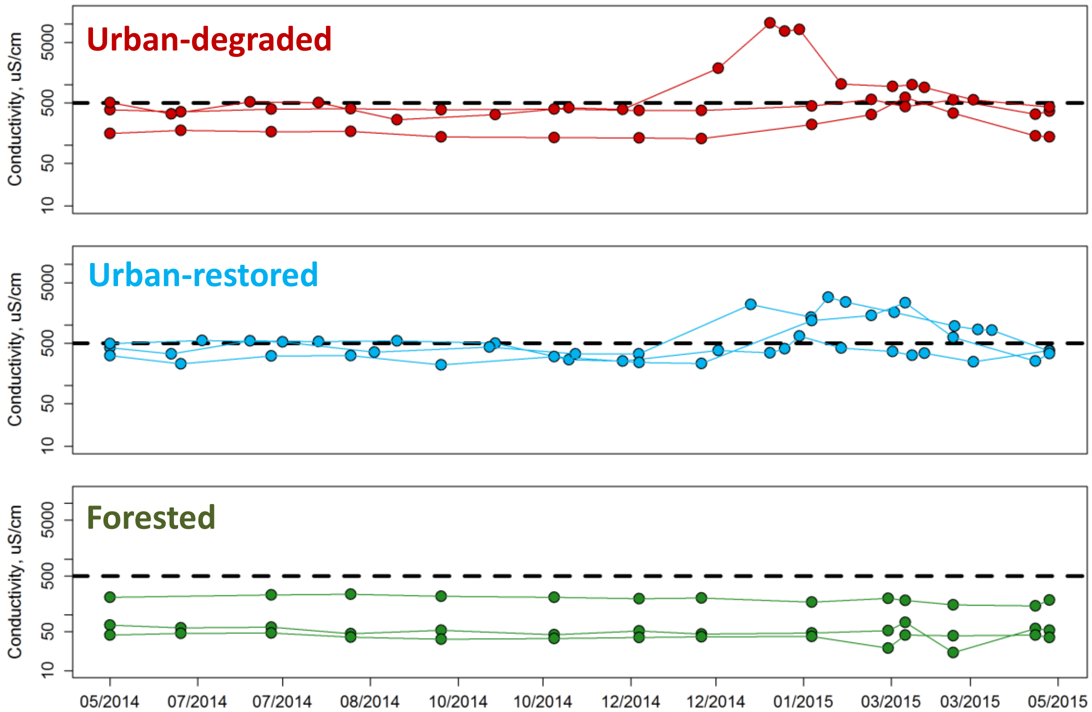

Figure 2.5: Annual baseflow time series of conductivity for the 10 study streams, reported as specific conductance (uS/cm) for the three types of watersheds (urban-degraded, urban-restored, forested). Note the y-axes in log10 scale.

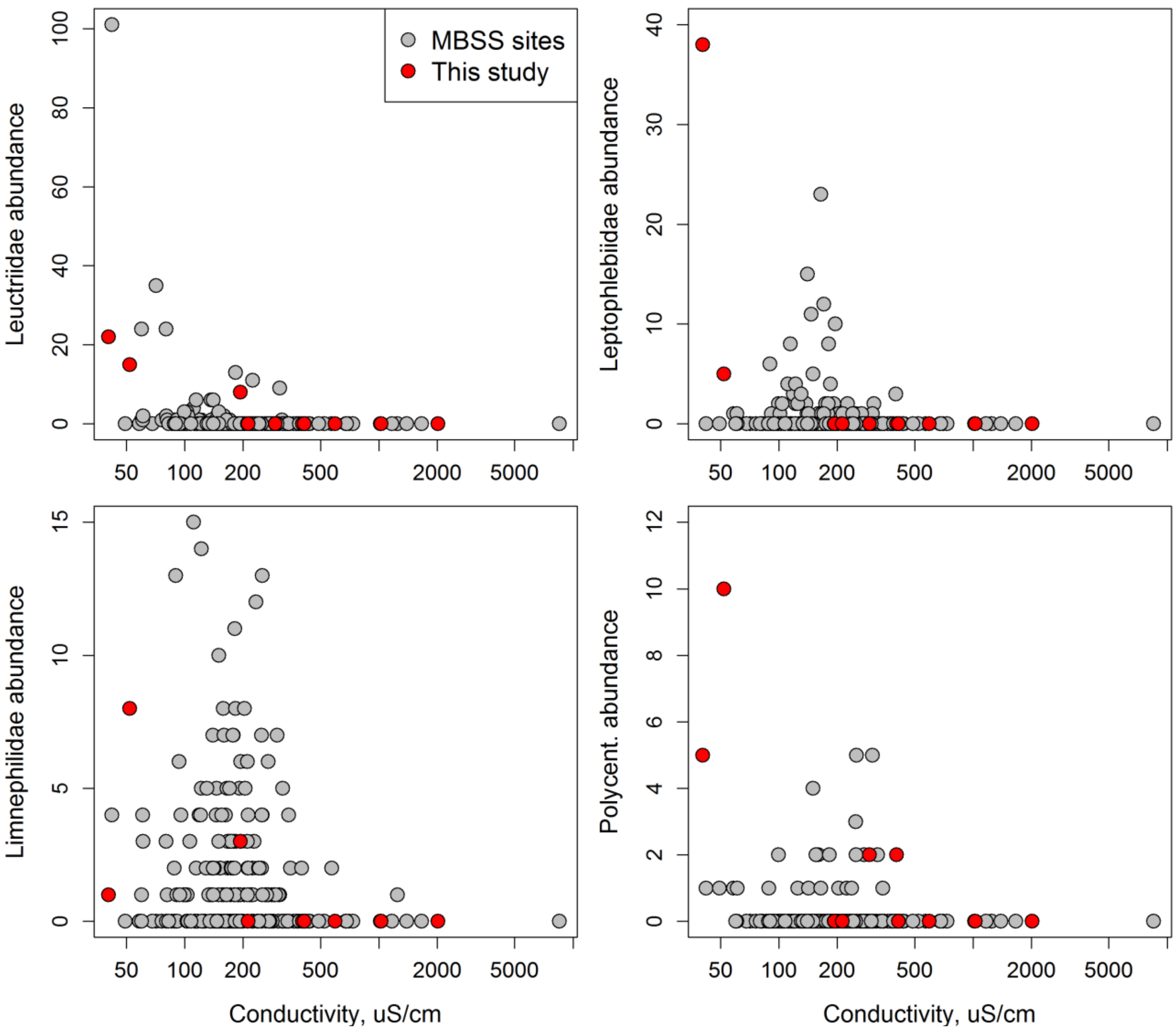

Figure 2.6: Relationships between sensitive taxa abundance and conductivity for the region (MBSS sites in gray) and for this study (in red).

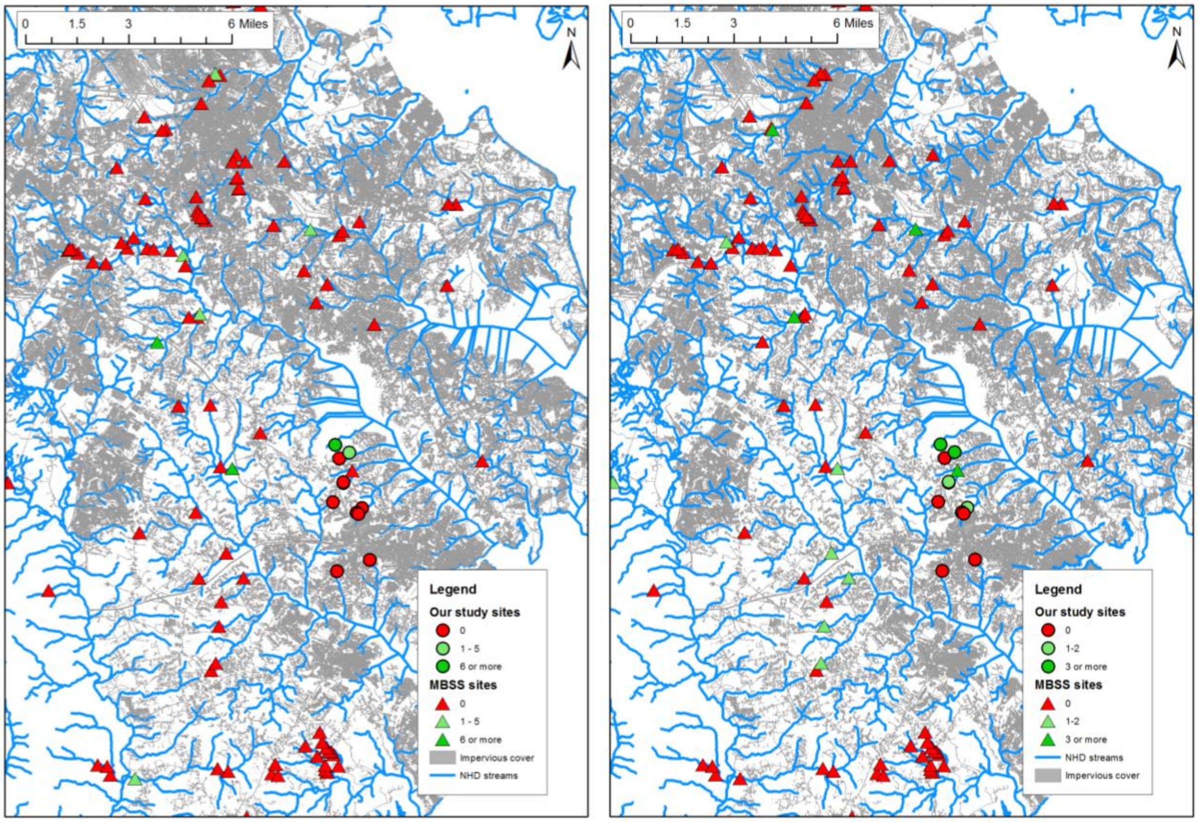

Figure 2.7: Presence and abundance of two EPT taxa fund at the western shore Coastal Plain region: (**left**) *Leptophlebiidae* (Order: Ephemeroptera) and (**right**) *Polycentropodidae* (Order: Trichoptera). Sites sampled for this study are represented by circles; sites sampled by MBSS represented by triangles.

Supplemental Information

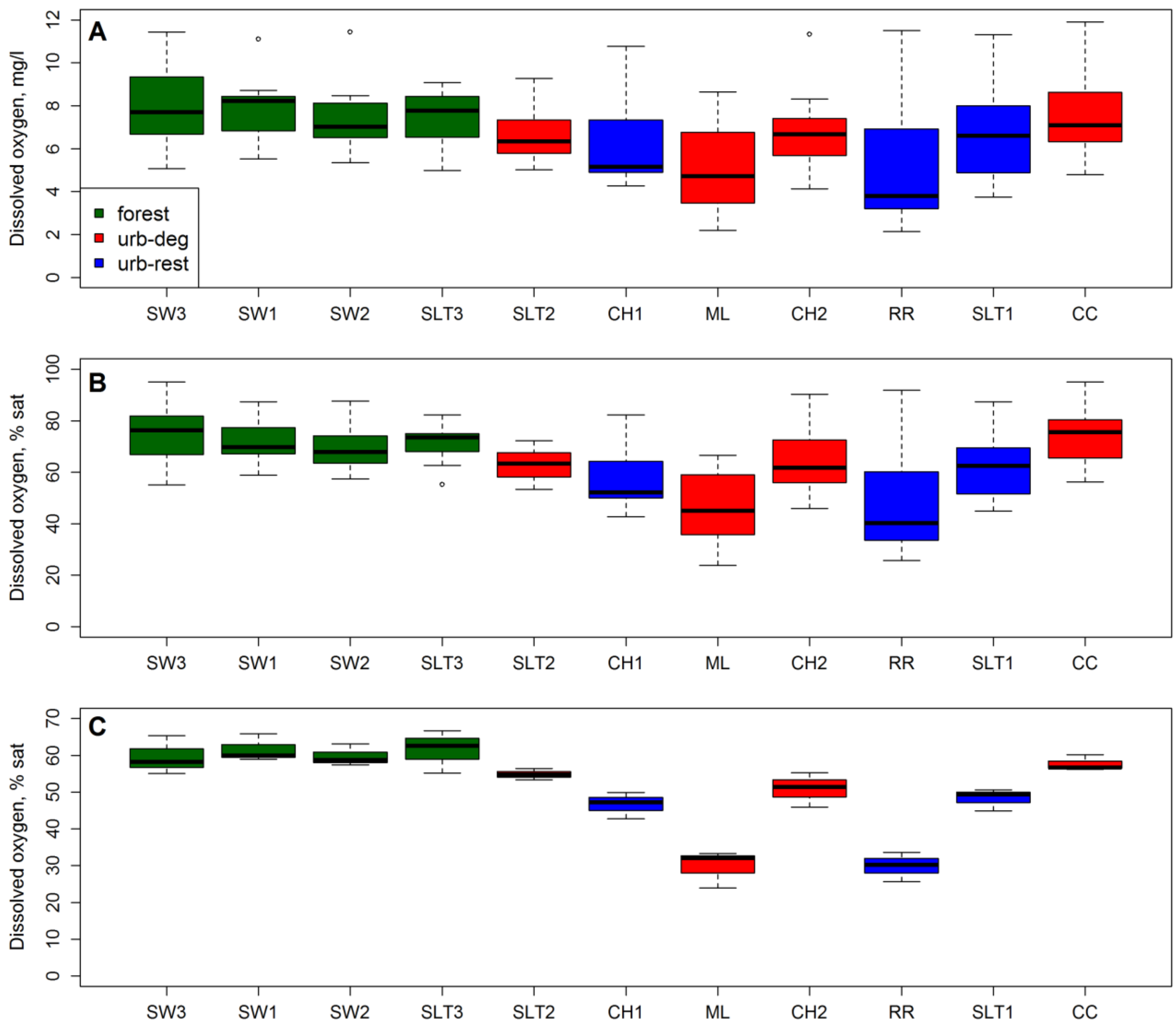

Figure SI-2.1: (A) Dissolved oxygen concentrations for the entire monitoring period (n = 11). (B) Dissolved oxygen (percent saturation) for the entire monitoring period (n = 11). (C) Dissolved oxygen (percent saturation) for the lowest three DO measurements during the monitoring period (n=3).

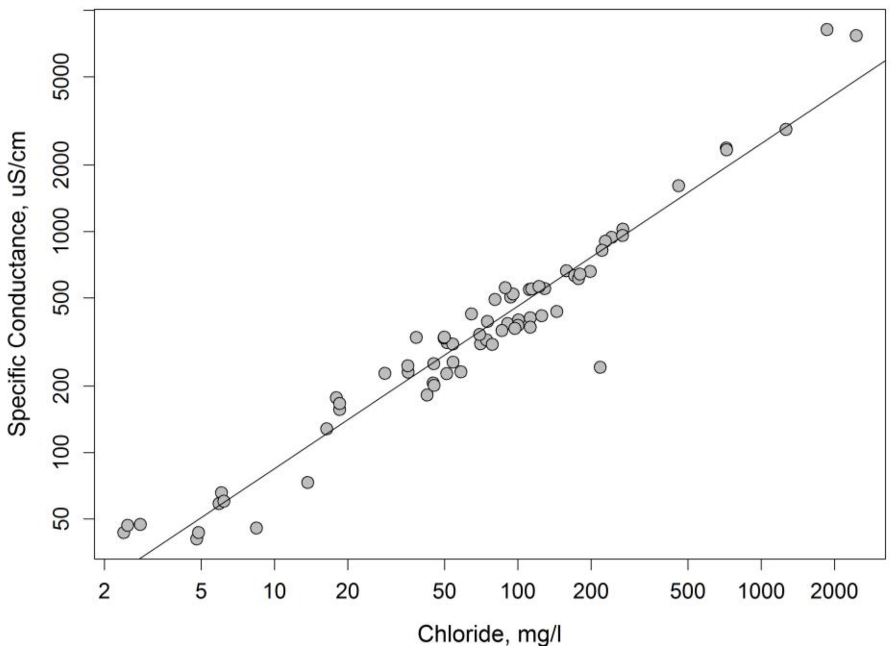

Figure SI-2.2: Relationship between chloride concentrations and conductivity (e.g., specific conductance) at the study sites (n= 72). Pooled data include samples taken from all 11 sites over 14 different sample dates.

Chapter 3: Climate and landscape controls on the hydrologic function of an infiltration-based stormwater control measure

Introduction

Given the continued degradation of stream ecosystems in urban landscapes (Paul and Meyer 2001), stormwater management goals are slowly shifting from that of mainly flood protection (Burns and others 2012; Hale 2016) to the goal of restoring hydro-ecological processes within urbanized streams and watersheds (Bernhardt and Palmer 2007; Walsh and others 2016). Watershed managers and stakeholders are faced with the dilemma of how to minimize the impacts of urbanization on aquatic ecosystems (Kaplowitz and Lupi 2012). Many of the newer approaches managers are adopting include infiltration-based stormwater control measures (SCMs) aimed to mimic pre-development hydrology in urbanized watersheds (Davis and others 2009; Dietz 2007). Because of their design, the effectiveness of infiltration-based SCMs is modulated by both the surrounding built environment and the natural environment, a phenomenon recently defined as "watershed capacitance" (Miles and Band 2015). Watershed capacitance describes the ability of an area to infiltrate, store, and slowly release runoff either as evapotranspiration or recharge. This interaction between an infiltration-based SCM and its surrounding landscape controls how well the SCM can perform its function and, ultimately, how successful these SCMs will be in restoring the water cycle to an urban landscape (Bhaskar and others 2016b; DeBusk and others 2011).

Bioretention basins are a widely utilized type of infiltration-based SCM (Davis and others 2001), and the effect of the surrounding environment on their hydrologic function has been well documented. For example, the effectiveness for runoff reduction in studied bioretention facilities studied by Winston and others (2016b) strongly depended on the drainage characteristics of the surrounding soil. Seasonal reductions in hydrologic performance can also occur when high groundwater tables reduce exfiltration rates (Hunt and others 2006). These seasonal effects can also be exacerbated by low rates of evapotranspiration, or ET (Hunt and others 2006), or temperature-induced reductions in hydraulic conductivity in the media within the bioretention basin itself (Emerson and Traver 2008). Studies have also shown that slow dissipation of groundwater mounding beneath bioretention basins (Endreny and Collins 2009) and infiltration trenches (Locatelli and others 2015) can reduce storage volumes. While bioretention and infiltration trenches treat concentrated runoff, even SCMs designed to reduce runoff generation are affected by the surrounding landscape. For example, Winston and others (2016a) observed poor runoff reductions from permeable pavement underlain by heavy clay soil.

Regenerative stormwater conveyances (RSCs) are another type of infiltration-based SCM whose design borrows concepts from bioretention technology (Brown and others 2010; Cizek 2014). Bioretention and RSCs both treat concentrated runoff through the combination of surface and subsurface storage in porous media (a "seepage bed" within the RSC), and both rely on ET and deep percolation as pathways for removing infiltrated runoff (Davis and others 2009; MDE 2009). However, bioretention basins are typically sited in upland topographic positions close to runoff sources, whereas RSCs are often placed in topographic depressions between the main storm sewer outfalls and the stream drainage network in ephemeral, intermittent, and perennial stream reaches (MDE 2009; Palmer and others 2014a). Because RSCs are often placed in these topographic convergence zones, they are closer by analogy to ephemeral losing streams (Winter and others 1998). The contrasting placement strategies of different infiltration-based SCM types may have important consequences for their ability to effectively infiltrate and store stormwater runoff, given the strong control the surrounding landscape plays on their hydrologic function. Indeed, recent studies on RSCs have shown quite a range in hydrologic function (Cizek 2014);[Chapter 1].

In addition to landscape factors, rainfall characteristics can clearly affect the hydrologic function of infiltration-based SCMs. For example, on individual event scales, the hydrologic performance of bioretention (Li and others 2009) and RSCs (Palmer and others 2014a) declined with increasing rainfall total. In the state of Maryland, SCMs are designed with a standard storage volume to capture runoff generated for a 1-inch rainfall event (MDE 2009). However, SCM design criteria may not explicitly incorporate rainfall intensity, which strongly controls runoff generation rates in urbanized landscapes (Dooge 1957), and therefore may affect SCM function. For example, Alfredo and others (2010) showed decreased green roof interception with increasing rainfall intensity, and peak flow reductions decreased during high-intensity rainfall events in a bioretention facility as well (Winston and others 2016b). More concerning is how these SCMs will perform over time as shifts in rainfall characteristics are expected (Vasiljevic and others 2012). Indeed, Hathaway and others (2014) found the frequency and magnitude of untreated runoff exported from a bioretention facility increases under future climate scenarios.

Given how the hydrologic function of infiltration-based SCMs is tightly coupled to local hydrology and climate (Forsee and Ahmad 2011; Koch and others 2014), there is an urgent need for more empirical studies on emerging approaches to infiltration-based stormwater management, such as RSCs. To address this, I sought to identify hydrologic processes responsible for affecting the hydrologic function of an RSC, and how that ultimately influences patterns in the conveyance of untreated stormwater runoff into the stream channel downstream of the system. I used a paired catchment approach to understand how the RSC modulates runoff inputs from the watershed. Specifically, the objectives of this study were to: 1) quantify spatial and temporal changes in subsurface storage within an RSC; 2) Understand the role subsurface storage dynamics plays in explaining responses in the stream channel; and 3) explore how SCM design itself interacts with climate and landscape factors in controlling its hydrologic function.

Methods

Study area and site descriptions

This research took place in the Coastal Plain Physiographic province near Annapolis, Maryland, USA (Figure 3.1A). The Coastal Plain is underlain by unconsolidated permeable sands and silts, interspersed with lower hydraulic conductivity soils (Bachman and others 1998). Topography is gently rolling in this region, with urban development occurring mostly on hilltops and low-gradient regions. Secondary forests in this region have remained intact on steeper hillslopes and in many riparian zones. The mid-Atlantic has a humid temperate climate with approximately 45 inches of rainfall each year, distributed evenly across the seasons. Rainfall intensity rates, however, vary widely between seasons, with lower intensity rainfall events largely in the fall and winter seasons, and convective thunderstorms in the spring and summer months are typically associated with higher intensity rainfall rates. Snowfall occasionally occurs in the late winter months, with a regional average of ~15 inches annually.

This research was conducted in two first-order, urban catchments that have been the focus of previous research on the effects of watershed restoration on headwater stream ecosystems [Chapters 1 and 2]. The two study catchments, hereby referred to as CH1 and CH2, are located adjacent to one another and have similar catchment areas, topography, and percent impervious cover (Table 3.1; Figure 3.1). The CH1 catchment, however, has been implemented with a RSC type of infiltration-based SCM, whereas the CH2 catchment contains no stormwater management facilities and serves as a negative control site. The infiltration-based SCM implemented in the CH1 catchment is located between the main storm-sewer outfall and the perennial stream channel, and is comprised of a vertical sand seepage bed with 16 infiltration pools to stabilize the structure on top (Figure 3.1B and 3.1C). These two catchments were part of larger studies to understand the effects of urbanization and SCM implementation on stream flow patterns [Chapter 1], as well as downstream water quality and aquatic insect biodiversity [Chapter 2]. In these previous studies, striking differences in water quality and hydrology had been documented between the two catchments, as well as between the CH1 catchment and two other urban watersheds that have also been implemented with an RSC type of infiltration-based SCM. These results were the motivation for additional hydrometric and groundwater monitoring at the CH1 watershed to understand the factors influencing storage capacity of the SCM.

Field data collection

I conducted high-frequency hydrometric monitoring in the two catchments and groundwater monitoring within the SCM seepage bed at CH1 for a one-year period between September 2015 and September 2016. At the catchment outlets, I maintained stage monitoring stations to record stream responses to rainfall events (orange arrows in Figure 3.1B). Hobo water level loggers were installed in perforated PVC pipes secured in the thalweg of each perennial stream channel and an additional logger was deployed out of the water on site to record barometric pressure. During this time period, I also monitored stage at the storm-sewer outlet into the top of the SCM within the CH1 catchment (green arrow on Figure 3.1B). This station is useful for understanding runoff behavior of the watershed above the SCM. All pressure loggers measured pressure and temperature between 2-and 5-minute intervals. High-frequency rainfall for the two catchments was measured with a Hobo tipping bucket rain gauge located in a clearing near the outlet of the CH1 catchment (yellow star on Figure 3.1B). The tipping bucket rain gauge measures the time stamp for each 0.2 mm of rainfall. Stage and rainfall datasets were used to quantify watershed and stream responses to individual rainfall events.

To understand changes in subsurface storage within the SCM during and following rainfall events, I conducted high-frequency groundwater monitoring within the SCM sand seepage bed (Figure 3.1C). Groundwater wells were installed within seven of the 16 infiltration pools, and were evenly distributed across the SCM. Wells were constructed out of perforated schedule 40 PVC pipe wrapped with mesh screen to prevent clogging. Each well was installed to a depth of a confining layer below and span the entire depth of the seepage bed section of the structure, and sealed with bentonite at the surface. In each of the wells, a hobo water level logger was suspended from a nylon rope and logged water levels at 2- or 5-minute intervals. Select sediment samples were also collected to characterize the grain size distribution of the seepage bed material.

Data analysis

To identify and characterize a population of rainfall events, I used a 5-hour minimum inter-event time (MIT) period to delineate individual rainfall events (Dunkerley 2015). Events with a MIT of 5 hours or greater were retained in the rain event dataset, and events with a MIT of less than 5 hours were aggregated until the rain-free periods around the rainfall periods met the 5-hour MIT criterion. The raw rainfall data was interpolated into 5-minute rainfall totals and both raw rain data and 5-minute interpolated data were used to characterize each rainfall event, including the total, duration, average rainfall intensity, maximum 5-minute rainfall intensity, and 7-day antecedent precipitation index. Finally, I screened the rainfall events and removed any events that were clearly influenced by snowfall, or indicated any malfunctioning of the rain gauge itself. I confirmed daily rainfall totals of select days with nearby CoCoRHAS rain gauges (gauge IDs MD-AA-4 and MD-AA-53).

To quantify the magnitude of the runoff response from the two study watersheds, I used the peak change in stage recorded at each of the two catchment outlets during individual rainfall events. Given that stage changes in a stream channel are dependent on the channel dimensions, I did not compare absolute stage changes between the two study catchments. Rather, I used the high-frequency stage hydrographs to 1) identify rainfall events during which there was an observable runoff response (vs. no runoff response) at each monitoring site; and 2) compared the relative differences in stream responses (defined by stage) during across different rainfall events with variable characteristics; this analysis was done separately for each of the two study watersheds. Stage changes during rainfall events were quantified by taking the average stage measured 1-hour prior to the beginning of the rainfall event and subtracting it from the maximum stage measured during the rainfall event. I used the stage changes at the three monitoring stations to quantify runoff frequencies for rainfall events with variable size, as well as the annual runoff frequency, which is the percent of rainfall events that generated an observable runoff response. For this metric, I defined an observable runoff response as a 3-cm change in stage or greater during the rainfall event. Runoff frequency defined by stage changes has been used in previous studies to document differences in catchment hydrologic behavior (McMahon and others 2003).

Water table fluctuations, sediment analyses, and detailed areal measurements of the infiltration pools were used to quantify changes in subsurface storage within the SCM. Given that this analysis was primarily focused on the relative changes in the 'state" of the subsurface storage between the rainfall events, I assume that the water table fluctuations observed in the 7 monitored infiltration pools are representative of the entire sand seepage bed. I estimated available subsurface storage in the unsaturated zone within the seven monitored infiltration pools using the following equation:

$$\text{Subsurface storage, } m^3 = [(GSE - WTE) * SA] / (n - Sr),$$

Where GSE = ground surface elevation (m), WTE = water table elevation (m), SA= infiltration pool surface area (m^2), n = estimated porosity of the sediments, and Sr = estimated specific retention of the sediments (Fetter 2001). Porosity and specific retention were estimated using the D_{50} grain size measured from several surficial sediment samples extracted in 3 of the infiltration pools (Beard and Weyl 1973). The surface area of each infiltration pool was directly measured using measuring tapes. The subsurface storage volumes beneath each of the seven monitored infiltration pools were summed to estimate a total monitored storage volume at each 2- or 5-minute time step. Finally, I scaled the total observed storage volume to the highest observed storage during the monitoring period to estimate the fraction of total potential subsurface storage within the SCM. From this scaled subsurface storage time series, I then quantified two metrics to describe the storage "state" within the seepage bed in the SCM: 1) the *initial storage* for a rainfall event, which is the average of the scaled subsurface storage for 1-

hour prior to the beginning of the rainfall event, and 2) the *minimum storage,* which is the smallest scaled storage value recorded during the rainfall event itself.

Analysis of individual rainfall events

Rainfall events with complete subsurface storage data were used to explore the relationships between rainfall characteristics, subsurface storage dynamics, and responses in the CH1 and CH2 catchment outlets. Simple and multiple linear regressions were used to initially explore the dataset and develop a conceptual model of how the response variables (CH1 and CH2 stream response) and explanatory variables (rainfall characteristics for both sites; storage metrics for CH1 only) were related. I then used confirmatory factor analysis (CFA) to test whether our conceptual model of how climate and storage characteristics controlled patterns in stream response at CH1 was supported by the observations. CFA is a type of structural equation modeling (SEM) that can be used to test a hypothesized set of relationships between multiple explanatory and response variables (Hall and others 2009; Somers and others 2013). The lavaan package in R was used to conduct the CFA analysis. I also used time series of these datasets to visualize the relationships between these factors and for understanding potential mechanisms controlling the hydrologic function of the SCM.

Results

Catchment responses to rainfall events

A total of 66 rainfall events were delineated for the monitoring period, evenly distributed across the summer, fall, winter, and early spring months. There was a 2-month gap from late April to late June 2016 with no rain event data due to malfunctioning of the rain gauge. Rainfall events during the monitoring period ranged from 0.4 mm to 50.8 mm in size (mean = 11.6 mm); the mean event duration was 8.5 hours, and the mean event average and 5-minute maximum rainfall intensities were 2.2 mm/hr and 23 mm/hr, respectively. The hydrologic responses of the two study catchments to rainfall events were quite variable (Figure 3.2). No runoff responses were observed at the CH1 catchment outlet for rainfall events less than 5 mm in size (Table 3.2). Only 23% of events between 5-9 mm in size generated a substantial runoff response, despite runoff being conveyed into the SCM during the vast majority of these events (92%; Table 3.2).

In contrast, runoff responses were observed at the CH2 catchment outlet during about half (43%) of rainfall events between 2-4 mm in size, and a majority (77%) of events 5-9 mm in size. This contrasting behavior was observed for larger events as well. Runoff responses were observed at the CH2 catchment outlet during all (100%) rain events between 10-24 mm, but only during half (50%) at the CH1 catchment outlet. Both study catchments consistently responded to rainfall events 30 mm or greater in size. On an annual basis, the watershed above the SCM at CH1 generated runoff during 74% of the rainfall events, while runoff responses at the CH1 catchment outlet were observed only during 30% of the rainfall events. The CH2 catchment, in contrast, responded to 65% of the rainfall events.

Water table and subsurface storage dynamics beneath the SCM at CH1

As expected, water table depth in the SCM seepage bed varied widely over time and space (Figure 3.3, Table 3.3). Water table depths were, in general, deeper beneath the upper infiltration pools (pools 16, 14, 13) and shallow beneath the lower infiltration pools (pools 3 and 1; Figure 2). The water tables beneath three pools (pools 16, 14, and 8) never reached the ground surface during the monitoring period (Table 3.3). Two pools had water tables which rarely reached or exceeded the ground surface (pool 13 =7.7 days and pool 6 = 3.1 days) and the water tables beneath the two remaining pools reached or exceeded the ground surface for a significant portion of time (pool 3 = 72.3 days; pool 1 = 86.2 days). Water tables in the SCM responded rapidly to rainfall events, presumably from infiltration of surface runoff conveyed into the SCM. The water tables beneath the upper pools (pools 16, 14, and 13) show the most frequent responses to rainfall events, whereas the lower pools (pools 6, 3, and 1), indicated longer time periods between responses (Figure 3.3).

The seepage bed beneath the monitored infiltration pools exhibited different types of drainage behavior: most pools (16, 14, 13, 8, and 6) exhibited sharp rises from head-driven, wetting front propagation and slower, gravity-driven drainage behavior after rain events. At pool 3, however, peaks in water table elevation during and after rainfall events were much broader than at the other sites (Figure 3.3; Figure SI-3.1). The water table beneath pool 1 also behaved differently than the other sites: although it exhibited sharp rises in response to rainfall events, water table recessions were slow and shallow. The water table at pool 1 remained at or near the ground surface elevation for most of the monitoring period, except for two instances: 1) at the beginning of the monitoring period, the water table at pool 1 was below the bottom of the well prior to a large rainfall event in late September 2015. The preceding summer had been unusually dry, and the water tables at all sites were beneath the bottom of their wells at this time; and 2) during a drier period in late June 2016, when the water table drew down significantly (Figure 3.3). Other than at pool 1, however, I did not observe any significant seasonal shifts in the water table elevation within the SCM.

Given the high variability in water table fluctuations in the seepage bed within the SCM, daily average subsurface storage volumes across the SCM were also quite variable (Figure 3.4).

Pools 16 and 8 had deep vadose zones and large surface areas, thereby providing the largest portion of the total monitored available subsurface storage (Table 3.3). Pools 14, 13, and 6 have intermediate storage volumes, and pools 3 and 1 both provide little to no additional storage to the overall SCM, given their relatively shallow water tables and small surface areas. The variability of subsurface storage across the seven sites was greater than the variability across seasons (Figure 3.4). Only slight seasonal differences in daily average storage volumes were observed in pool 1 (slightly greater in spring and summer), and some instances where extreme low daily average values were lower in the winter and fall seasons (pools 16 and 6; Figure 3.4).

Relationships between rainfall characteristics, subsurface storage, and stream responses

Of the 66 rainfall events delineated for the monitoring period, 43 of them had complete groundwater data that were used to quantify initial and minimum storage values in the seepage bed within the SCM at CH1. Initial storage availability in the seepage bed immediately prior to the 43 rainfall events ranged from 99% to 16% of the total potential storage observed during the monitoring period. However, the mean initial storage across events was relatively high (74%), suggesting that subsurface storage was largely available at the beginning of most rainfall events. Initial storage declined with increasing 7-day antecedent precipitation index (API), indicating the effect of recent rainfall on antecedent moisture within the seepage bed (Figure 3.5). Two events were an exception to this pattern: rainfall events # 14 and 15 occurred in early February 2016, after a significant snowfall event in late January (approx. 24 inches of snowfall fell on January 23-24). The 7-day API did not capture this precipitation event, as it was stored for a longer time period as snowpack. Minimum storage observed in the seepage bed during the 43 rainfall events ranged from 99% to 6% of the total potential storage during the time period, with an average event-based minimum storage of 53%. Minimum storage declined with lower initial storage values, though there was some scatter in the relationship (Figure 3.5). Minimum storage observed during the rainfall events also declined with increasing rainfall totals (Figure 3.5).

For this subset of rainfall events, stream responses at the CH1 outlet to rainfall totals follow similar patterns as the full dataset (Figure 3.6A; Figure 3.2). Correlations between rain event characteristics and stream responses followed a somewhat similar pattern at the two catchment outlets. Stream responses at both catchment outlets had the greatest correlation with rainfall intensity (CH1 $p < 0.001$, $R^2 = 0.76$; CH2 $p < 0.001$, $R^2 = 0.75$; Figures 3.6B and 3.6E, respectively), followed by rainfall total (CH1 $p < 0.001$, $R^2 = 0.62$; CH2 $p < 0.001$, $R^2 = 0.51$; Figures 3.6A and 3.6D, respectively). Neither CH1 nor CH2 stream responses were significantly correlated to rainfall event duration or 7-day API. Stream responses at the CH1 stream outlet declined with decreasing minimum storage in the seepage bed ($p < 0.004$; $R^2 = 0.44$).

I used SEM to explore intermediate relationships between some of these explanatory and response variables. For example, I hypothesized that the relationship between stream response and rainfall total is modulated by the subsurface storage provided by the SCM, given the significant correlation between minimum storage and rainfall total (Figure 5; $p < 0.001$; $R^2 = 0.56$). Our SEM model converged after 47 iterations (chi-squared = 0.50; DF = 7) and explained 77% of the variance observed in log-stream response at the CH1 outlet. The analysis confirmed that both storage and climate do control patterns in stream responses at the CH1 outlet, but that rainfall intensity exerted the strongest direct control on stream response (Figure 3.7). The final model did not include any direct relationship between rainfall total and stream response, suggesting that relationship was indeed modulated by the presence of additional subsurface storage in the SCM. However, the SEM model did not indicate any significant “decoupling” between rainfall intensity and the stream response at the catchment outlet from SCM implementation.

Classifying rainfall events based on hydrologic responses of the SCM and the watersheds

To explore these relationships empirically, I classified the rainfall events for which there was at least one runoff response observed at the two study catchments. I classified the events based on the presence/absence of runoff responses at the CH1 catchment; the relative timing of the runoff peaks at the two catchments; the state of the storage in the seepage bed; and whether there was a pulse of high-intensity rainfall during the event (Table 3.4). Of the 43 events included in the analysis, 31 rainfall events generated a runoff response in the CH2 catchment, but only 16 of those generated a significant runoff response at the CH1 catchment outlet. The remaining 15 events exhibited no exceedance of the SCM (Figure 3.8; Figure 3.9). Of those 16 events that did generate a runoff response at CH1, only 2 events exhibited clear evidence of storage exceedance, meaning the seepage bed was filled to capacity prior to the observed runoff response at the catchment outlet (Table 3.5, Figure 3.8). These events were long in duration with low-intensity rainfall rates and very low initial storage in the seepage bed (Figure 3.9, Table 3.5). In contrast, 8 rainfall events exhibited clear evidence of infiltration exceedance occurring in the SCM. These rainfall events typically had very high maximum rainfall rates, short durations, and moderate rainfall totals (Figure 3.9, Table 3.5). The remaining 6 events exhibited a mix of both storage and infiltration exceedance behavior in the SCM, and were characterized by larger rainfall totals, lower 7-day API, and a pulse of high intensity rainfall.

Discussion

Overall, the infiltration-based SCM at CH1 effectively reduced runoff frequency at the outlet compared to CH2, the negative control catchment. During 23 of the 66 rainfall events, the SCM completely infiltrated runoff conveyed from the upland storm sewer network, thereby disconnecting it from the downstream ecosystem. Reduced runoff frequency in streams can help maintain stable banks (Konrad and others 2005) and reduces the effect of flow disturbances on biofilms and benthic aquatic communities (Utz and Hilderbrand 2011). Moreover, the infiltration of runoff may reduce the water quality impacts from urbanization on the downstream ecosystem, including the conveyance of heat (Somers and others 2016) and other urban pollutants (Paul and Meyer 2001), especially if the SCM infiltrates the “first flush”, which often carries the bulk of containments (Schiff and others 2016). The infiltration-based SCM at CH1 has been shown to both reduce total suspended sediments exported at the catchment outlet compared to CH2 during smaller events (Palmer and others 2014a) and reduce temperature surges in stream temperature by intercepting warm runoff during summer events [Chapter 2].

The runoff frequencies documented in this study are consistent with runoff frequencies quantified during an earlier time period [Chapter 1]. In this earlier study, runoff frequencies were quantified for these two catchments and nine additional headwater catchments that varied in level of imperviousness and SCM implementation. Although two of these additional catchments had also been implemented with the same type of infiltration-based SCM (catchment IDs: RR and SALT1), I did not observe any significant reduction in runoff frequency compared to their urban counterparts, suggesting that effectiveness of these SCMs may be limited in certain circumstances. In light of these results, the primary motivation for this study was to identify the processes that may be controlling effectiveness of this nascent SCM design.

Processes affecting subsurface storage within the SCM seepage bed

The ability of infiltration-based SCMs to mitigate the impacts of stormwater runoff is partially controlled by the storage capacity within the structure itself. Potential subsurface storage in the unsaturated zone clearly varied across the SCM in our study, and was likely driven by the site's topography, the surrounding landscape, and the relative position of the infiltration pools. I observed several factors that could contribute to reduced storage in sections of the seepage bed within the SCM. First, the regional ground water table limited the storage volume in the seepage bed beneath pool 1 during most of the monitoring period. I observed only two time periods where the water table beneath pool 1 well dropped enough to provide significant storage in the seepage bed; both of these instances were in the summer months after a dry period. As discussed earlier, other studies on infiltration-based SCMs have also documented the influence of groundwater on infiltration-based SCMs (Hunt and others 2006; Locatelli and others 2015), including another study on a RSC (Cizek 2014). Secondly, there may be subsurface storm flow draining from the upper part of the SCM into the seepage bed beneath pool 3. The water table in the seepage bed here exhibits very broad peaks with maximum water table elevations occurring a day or two later than the other monitoring sites. Water table recession limbs are also concave in shape, rather than the exponential-decay type drainage observed in the other monitoring wells (Figure SI-3.1). The steep drop in elevation immediately up-gradient of pool 3 (Figure 3.1C) may concentrate subsurface stormflow in this region if flow is lateral through the seepage bed, rather than vertically into the surrounding soils (Hewlett 1961). This may occur if the surrounding soils of the original stream channel are composed of low hydraulic conductivity material, which has been observed in the channels downstream of the SCM.

Finally, low hydraulic conductivity sediments at the surface of the infiltration pools or within the seepage bed could also affect drainage rates and, therefore, the overall storage capacity of the system. Grain size analysis of surficial sediments from 3 infiltration pools suggest the sediments in the seepage bed largely adhere to the design standards of 0.5-1 mm diameter material (Table SI-3.1). However, samples did include finer grained sediments, which often controls the overall hydraulic conductivity of the material (Beard and Weyl 1973). Preliminary monitoring in four other infiltration pools in an earlier study (pools 9, 10, 11, and 12) revealed highly variable drainage rates, with pools 10 and 11 showing prolonged ponding of surface water extending well beyond the design standard of 24-hours after the event. Communications with the practitioner after construction suggested the mixing of native soil into the seepage bed to regulate the residence time of the system and support redox-sensitive biogeochemical processes (e.g., denitrification; (Hsieh and others 2007). This mixing of lower hydraulic conductivity sediment into the seepage bed, especially near the surface, could have inhibited infiltration rates, which reduces the SCM storage capacity.

Processes controlling the hydrologic function of the infiltration-based SCM

Despite these factors potentially constraining the subsurface storage availability in the SCM, there were only two rainfall events in the monitoring period during which the lack of subsurface storage was the sole factor contributing to runoff conveyance into the downstream channel (i.e., storage exceedance; Table 3.5; Figure 3.8). These two events were larger (rainfall total > 10 mm), longer duration events with unusually low initial storage from previous precipitation events (note that 7-day API for event #15 does not reflect the actual antecedent moisture, as a snowpack retained moisture in the watershed for longer than seven days). In contrast, an average of 43% of the potential subsurface storage in the SCM remained available during the other 14 rainfall events for which there was a substantial runoff response at the CH1 catchment outlet. These rainfall events all had a pulse of high-intensity rainfall which immediately preceded the rising limb of the storm hydrograph in the downstream channel (Figure 3.8). This suggest that high-intensity rainfall, and its associated runoff delivery rates into the SCM, may be controlling stormwater-stream connectivity in this system. The SEM results confirmed that this conceptual model fits our empirical results (Figure 3.7), and that the infiltration excess mechanism is more likely limiting the capacity of the SCM than storage limitations within the structure itself.

The storage exceedance and infiltration exceedance mechanisms observed in the SCM are comparable to two overland flow generation processes: saturation-excess overland flow and infiltration-excess overland flow. Saturation-excess overland flow occurs when subsurface stormflow or rising groundwater occupies the vadose zone, eventually reaching the surface (Dunne and Black 1970). The saturation exceedance mechanism in the SCM is different in that the majority of the inputs to the system are from the top (infiltrated surface runoff). However, given the topographic position of this structure in a convergence zone, some available space may be filled by subsurface inputs as well (e.g., groundwater, previously infiltrated runoff from upper pools). Infiltration exceedance is somewhat akin to infiltration-excess overland flow, which occurs when rainfall intensity rates exceed the infiltration rates of the sediments (Horton 1933). However, in this study, the infiltration rates of the seepage bed sediments "compete" with the runoff delivery rate from the catchment, rather than the rainfall intensity itself, although rainfall intensity will control the magnitude of runoff delivery rates.

Implications for SCM design to restore hydrologic function to urban catchments

In the broadest sense, the goal of infiltration-based SCMs is to maximize the partitioning of incoming runoff into subsurface seepage. This partitioning of runoff into the subsurface is similar to the process of hyporheic exchange in natural stream channels, especially streams that have "losing" conditions (Winter and others 1998). As I earlier noted, the amount of runoff infiltrated into the subsurface strongly depends on the hydraulic conductivity of the seepage bed material. Stream hyporheic exchange is also strongly controlled by the composition of streambed material, with lower exchange rates measured in streams with low-K sediments (Morrice and others 1997; Valett and others 1996; Wroblicky and others 1998). The proportion of incoming runoff infiltrated by the SCM is likely declining with increasing runoff delivery rates. Returning to the hyporheic zone analogy, the proportion of total streamflow entering the hyporheic zone has also been found to decline with increasing discharge rates (Wondzell 2011). Runoff delivery rates are controlled by the size of the contributing area, the impervious cover, the configuration of the storm sewer network, and rainfall intensity.

Beside runoff delivery rates, I propose that the surface area, or footprint, of the SCM may play an equally important role in determining how much incoming runoff is infiltrated by the SCM. The surface area of the seepage bed (relative to the runoff delivery rate) may constrain the partitioning of runoff into the subsurface, much like the wetted perimeter of a streambed partially constrains the fraction of streamflow that flows into the hyporheic zone. Streams with larger wetted perimeters tend to have greater exchange rates because more of the water column comes into contact with the streambed, which reduces velocities (Boulton and others 1998). Holding the total storage volume constant, a SCM with large watershed inputs and a small surface area will not be able to infiltrate as much runoff as one with a larger surface area. Indeed, this surface area control on runoff reduction has been documented for bioretention basins (Davis and others 2012; Li and others 2009).

Assuming permeability of the seepage bed or subsurface media adheres to design standards, I propose that the SCM surface area and the runoff delivery rates collectively control overall hydrologic function of infiltration-based SCMs (Figure 3.10). Ideally, SCM designs that utilize infiltration should maximize the ratio of surface area of permeable sediments to the runoff rate delivery from the catchment. SCM approaches that are most likely achieving this are practices that restore the infiltration capacity of the ground surface itself to prevent or reduce runoff generation (e.g., permeable pavement or soil restoration; (Olson and others 2013; Schwartz and Smith 2016). Bioretention that captures small volumes of runoff can also be relatively effective. RSCs, such as the one at the CH1 catchment with a relatively small catchment area and moderate impervious cover, are expected to be relatively effective at infiltrating concentrated runoff and decreasing runoff frequency. However, RSCs implemented in catchments with larger contributing areas and/or greater impervious cover have been shown to be less successful at infiltrating runoff and may act more like detention basins [Chapter 1].

Conclusions

The goal of this study was to identify processes affecting hydrological function of a nascent infiltration-based SCM design rapidly gaining popularity in the mid-Atlantic region. I found that processes constraining available subsurface storage were similar to those affecting storage in other types of infiltration-type SCMs (e.g., bioretention basins). Although evidence of groundwater intrusion at this site was documented, I did not observe any significant seasonal reductions in the seepage bed, because the majority of the SCM was located well above the regional groundwater table. Although sediments largely followed design specifications, there was some evidence of slightly lower hydraulic conductivity materials present at this site as well. When I examined storage dynamics during rainfall events, I found little evidence that storage limitations controlled its hydrologic function. However, there were two cases in which storage was completely filled and conveyed runoff to the downstream channel. The dominant mechanism controlling SCM function seemed to be infiltration exceedance, which occurs when high-intensity rainfall caused the conveyance of untreated runoff into the stream, regardless that storage in the seepage bed remained available. I found that the unique placement of RSCs within topographic convergence zones may limit their hydrologic functionality when compared to bioretention basins. Given their location between the storm sewer network and the perennial stream channel, concentrated runoff from the contributing area may bypass the system under certain circumstances because their relatively small footprint may limit runoff interception during high-intensity rainfall events. Finally, I suggest that infiltration-based SCMs should maximize the surface area/runoff delivery rate ratio will be most effective at intercepting runoff and restoring hydrologic function to urbanized landscapes.

Tables and Figures

Table 3.1: Characteristics for the three watersheds in the study.

Site ID	**Percent impervious cover, %**	**Area, ha**	**Watershed type**
CH1	22.2	5.4	urb-restored
CH2	23.2	5.6	urb-degaded

Table 3.2: Runoff frequency for different sized rainfall events during the monitoring period.

	Runoff frequency for specific size rainfall events, mm[a]								Annual runoff frequency
	< 2	**2-4**	**5-9**	**10-14**	**15-19**	**20-24**	**25-30**	**> 30**	
No. events	12	14	13	10	2	4	7	4	**--**
CH1-top	0.08	0.64	0.92	1.00	1.00	1.00	1.00	1.00	0.74
CH1	0.00	0.00	0.23	0.50	0.50	0.50	0.71	1.00	0.30
CH2	0.00	0.43	0.77	1.00	1.00	1.00	1.00	1.00	0.65

[a] Runoff frequency = number of rainfall events during which a stage change of 3 cm or more was observed.

Table 3.3: Characteristics of the seven monitoring groundwater wells within the CH1 SCM.

well ID	GSE, m	Depth, m	Days above GSE	Surface area, m^2	Water table depth, cm		
					min	mean	max
16	22.4	2.3	0	58	-234	-186	-37
14	19.6	2.2	0	42	-227	-127	-8
13	19.3	2.6	7.7	46	-157	-109	15
8	15.2	2.1	0	62	-209	-183	-2
6	13.8	1.4	3.1	60	-140	-120	20
3	11.4	1.1	72.3	30	-111	-62	50
1	11.3	0.8	86.2	25	-85	-10	56

GSE= Ground surface elevation

Table 3.4: Descriptions for classifying the rainfall events based on their rainfall characteristics, subsurface storage responses in the CH1 SCM and stream responses at the CH1 and CH2 watershed outlets (n = 31 rainfall events).

Response type	Response in CH2	Response in CH1	Peak stage timing	Storage response	High-intensity rainfall	No. events
No exceed. (NE)	YES	NO	---	----	---	15
Storage exceed. (SE)	YES	YES	Offset	Approaches zero prior to stream response	NO	2
Infiltration exceed. (IE)	YES	YES	Coincident	No decline until after stream response	YES	8
Both SE and IE (BOTH)	YES	YES	Offset	Declines before stream response	YES	6

Table 3.5: Rainfall and storage characteristics for the 16 rainfall events that generated a significant (>= 3 cm) runoff response at the CH1 catchment outlet.

Event ID	Event Start	Event duration, hours	Rainfall total, mm	Average rainfall, mm/hr	Max rainfall, mm/hr	Initial storage	Minimum storage	7-day API	Stage change, cm	Runoff response type[a]
8	11/19/2015	13.4	23.4	1.8	28.8	0.93	0.40	0.2	1.3	NE
4	10/1/2015	32.7	42.8	1.3	16.8	0.27	-0.01	69.4	9.1	SE
15	2/3/2016	10.9	13.6	1.2	16.8	0.45	0.10	4	7.6	SE
26	6/28/2016	0.7	11	15.3	55.2	0.62	0.55	33	9.4	IE
30	7/8/2016	4.4	8.6	1.9	57.6	0.74	0.44	45	11.7	IE
32	7/13/2016	2.3	13.6	5.9	38.4	0.72	0.46	14.4	13.5	IE
36	7/28/2016	12.7	40.6	3.2	108	0.95	0.45	0.2	10.7	IE
37	7/30/2016	8.2	28.2	3.4	62.4	0.68	0.34	41	21.7	IE
38	8/1/2016	3.5	9	2.6	64.8	0.59	0.34	69.4	10.2	IE
41	8/17/2016	8.8	7.4	0.8	40.8	0.75	0.39	18.6	8.1	IE
43	9/1/2016	7.5	11.8	1.6	52.8	0.95	0.71	0	7.9	IE
2	9/29/2015	7.5	50.8	6.7	67.2	0.99	0.35	1	14.9	BOTH
18	2/23/2016	40.3	36.8	0.9	48	0.80	0.13	41.4	10.9	BOTH
27	7/1/2016	4.1	19.6	4.7	60	0.73	0.55	11.4	8.5	BOTH
29	7/4/2016	16.3	23.8	1.5	38.4	0.72	0.34	32.4	5.6	BOTH
39	8/14/2016	1.4	12.2	8.6	62.4	0.96	0.72	0.2	6.1	BOTH
42	8/21/2016	10.6	23	2.2	50.4	0.79	0.25	26.2	15.2	BOTH

[a] NE = no exceedance; SE = storage exceedance; IE = infiltration exceedance; BOTH = both IE and SE

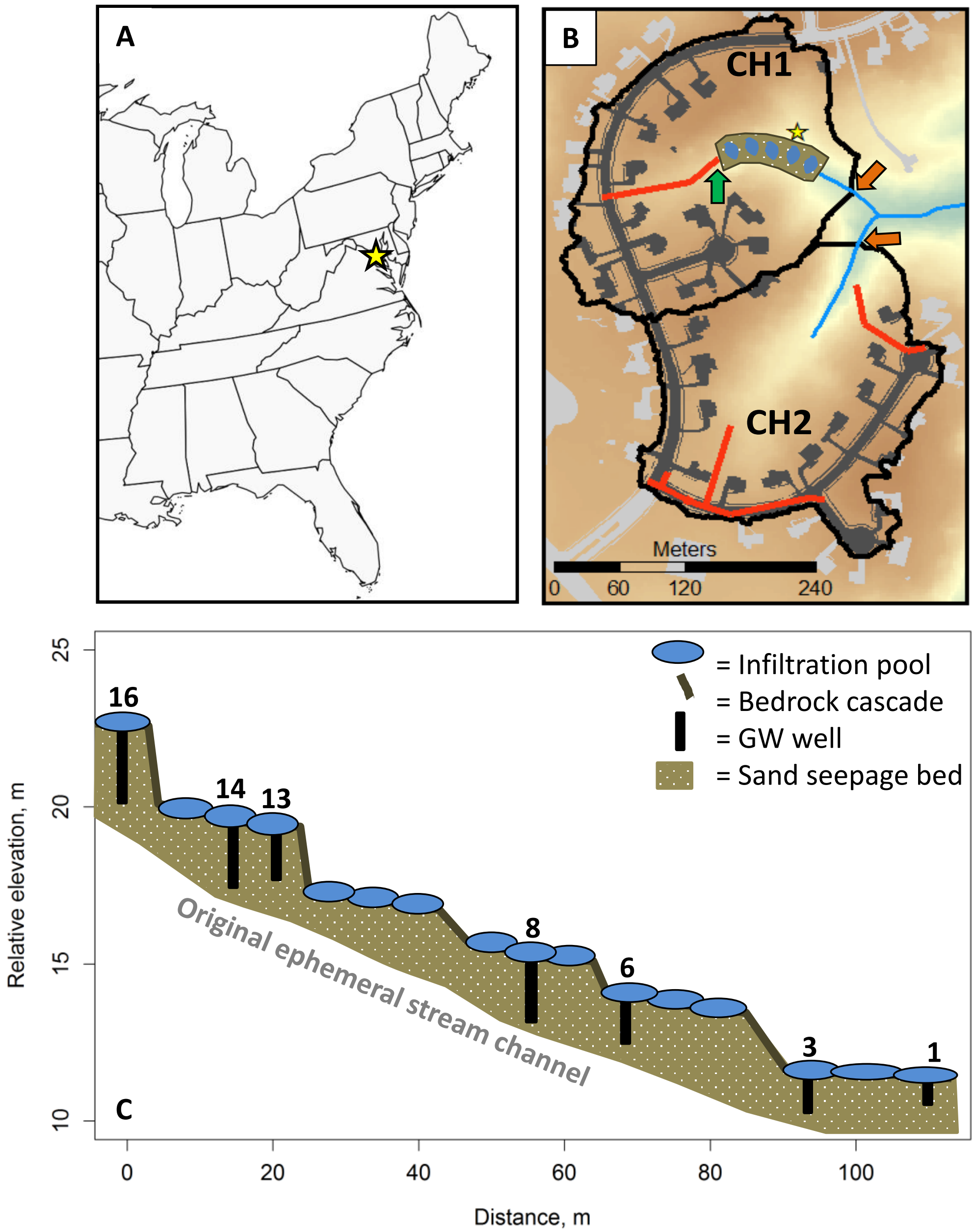

Figure 3.1: (A) Study area location within the Eastern United States. (B) Detailed site map for the two study catchments (CH1 and CH2). Location of the SCM in the CH1 catchment is denoted with the brown and blue structure between the storm sewer outfall (red lines) and the perennial stream (blue lines). Yellow star denotes location of tipping bucket rain gauges. Orange arrows indicate stage monitoring locations at the watershed outlets, and the green arrow indicates stage monitoring at the inlet above the SCM. (C) Detailed site map of the groundwater monitoring at the SCM within the CH1 catchment. Numbers denote the infiltration pool ID for each of the 7 monitoring wells.

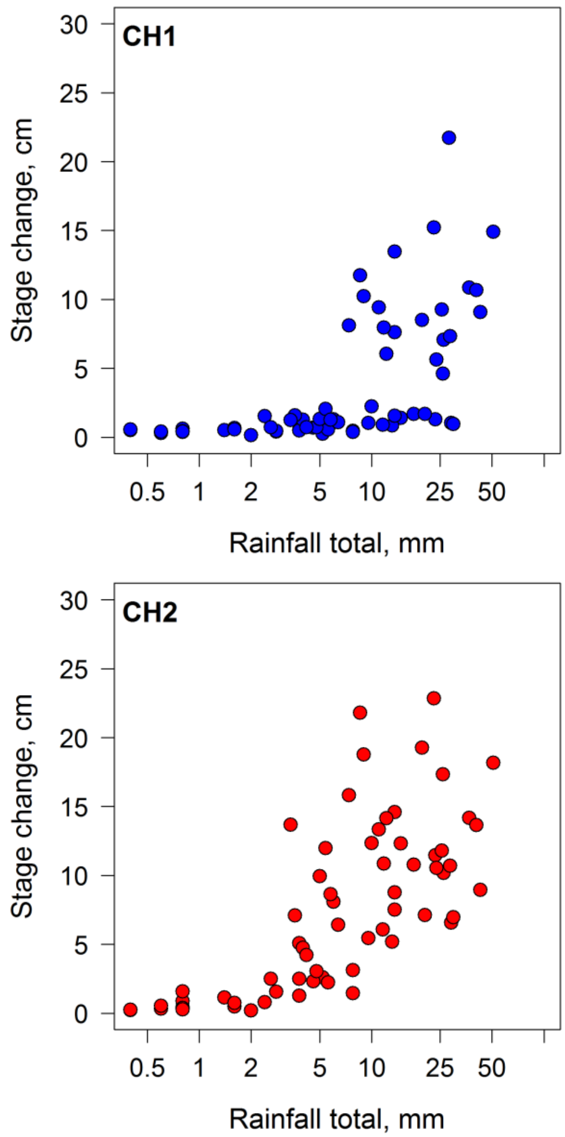

Figure 3.2: Relationship between stream response and rainfall event total (mm) at the urban restored catchment (CH1) and the negative control catchment (CH2) during the one-year monitoring period (n = 66 rainfall events). Stream responses are the change in stage (maximum stage during event - pre-event stage) observed at the outlets of the two study watersheds.

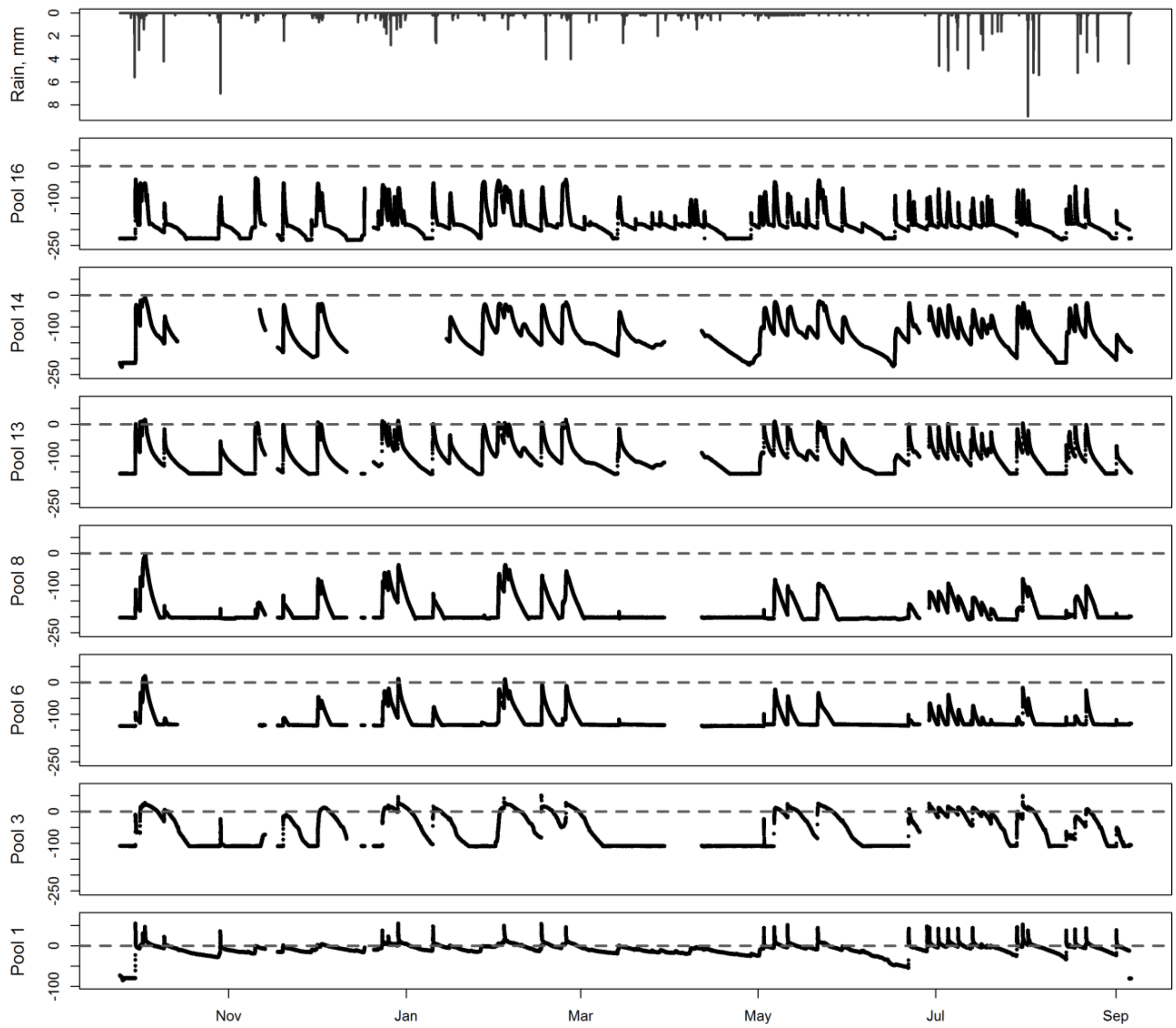

Figure 3.3: Patterns in precipitation and water table dynamics within the CH1 SCM during the monitoring period **Top panel**: Five-minute rainfall totals from the tipping bucket rain gauge. **Remaining panels**: Changes in water table depth beneath the ground surface (cm) over time at the seven groundwater wells (see figure 1c for a detailed site map). Dashed gray lines indicate the ground surface. Note the different scale for the Pool 1 water table elevations.

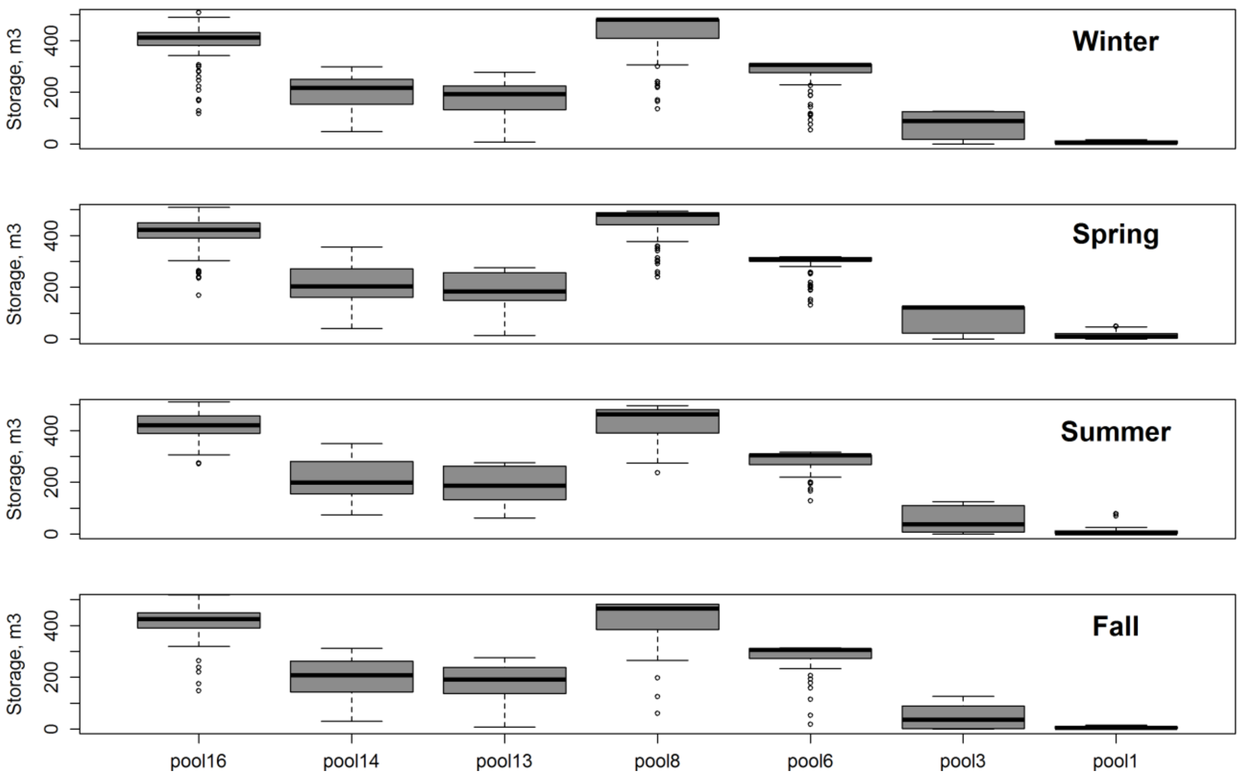

Figure 3.4: Mean daily subsurface storage (m^2) in the seepage bed beneath the seven monitored infiltration pools for each of the seasons (winter, spring, summer, fall). Sites are ordered such that the wells on the left are in the upper portion of the SCM (see Figure 1c for site configuration).

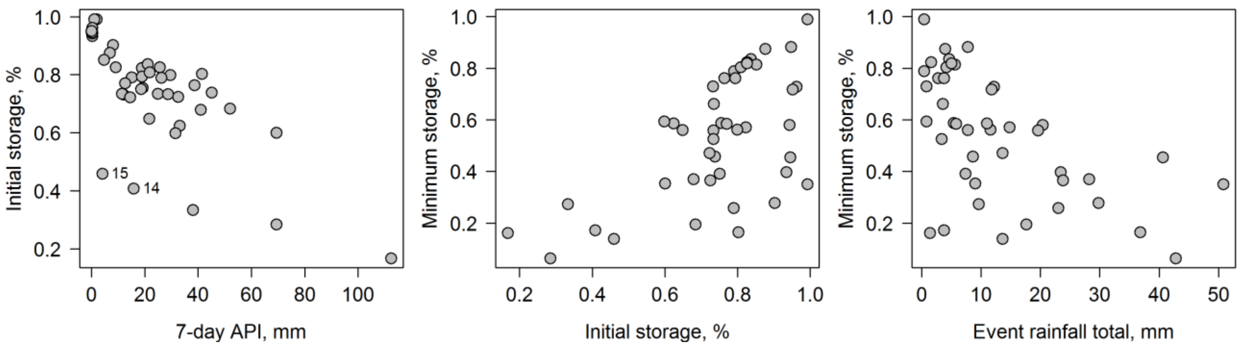

Figure 3.5: Relationships between rainfall and storage dynamics during a subset of rainfall events in the latter monitoring period with complete groundwater data (n=43 events). (**Left**) Initial subsurface storage in the SCM vs. 7-day antecedent rainfall (mm). (**Middle**) Minimum subsurface storage observed during the rainfall event vs. initial storage prior to the rainfall event in the SCM. (**Right**) Stream response at the CH1 watershed outlet vs. minimum subsurface storage observed during the rainfall event.

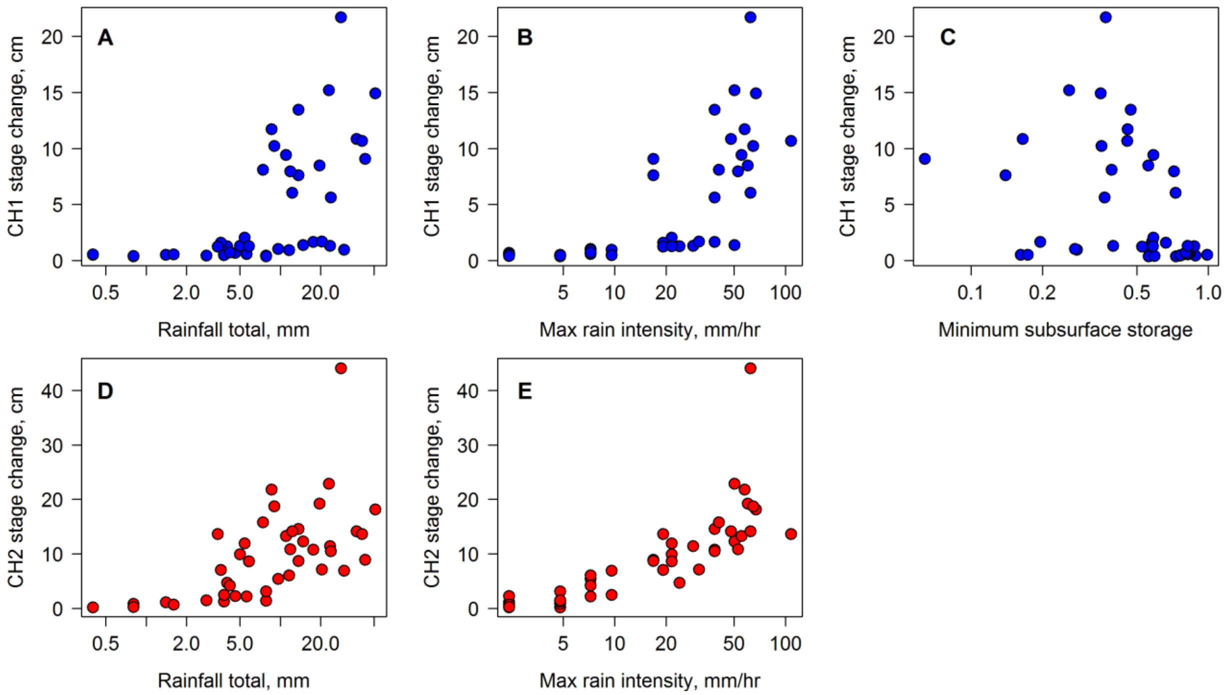

Figure 3.6: (**Top row**) Relationships between (A) rainfall total, (B) rainfall intensity, and (C) minimum storage characteristics and stream response at the CH1 catchment outlet. (**bottom row**) Relationships between rainfall total (D) and rainfall intensity (E) and stream responses at the CH2 catchment outlet (n = 43 rainfall events).

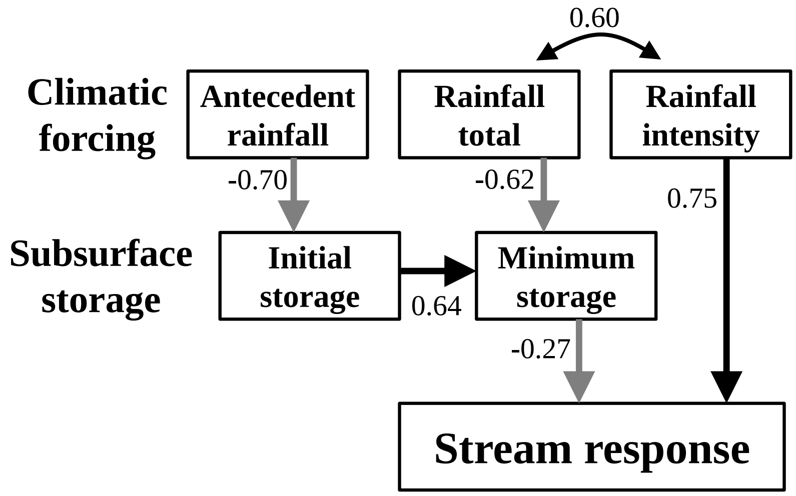

Figure 3.7: Results from the confirmatory factor analysis (CFA) relating climatic factors, subsurface storage characteristics, and stream responses observed at the CH1 watershed outlet. Results were generated from data during 43 rainfall events during the monitoring period. Black arrows indicate a significant positive relationship and gray arrows indicate a significant negative relationship ($p < 0.05$). The strength of the relationship is denoted with standardized regression weights.

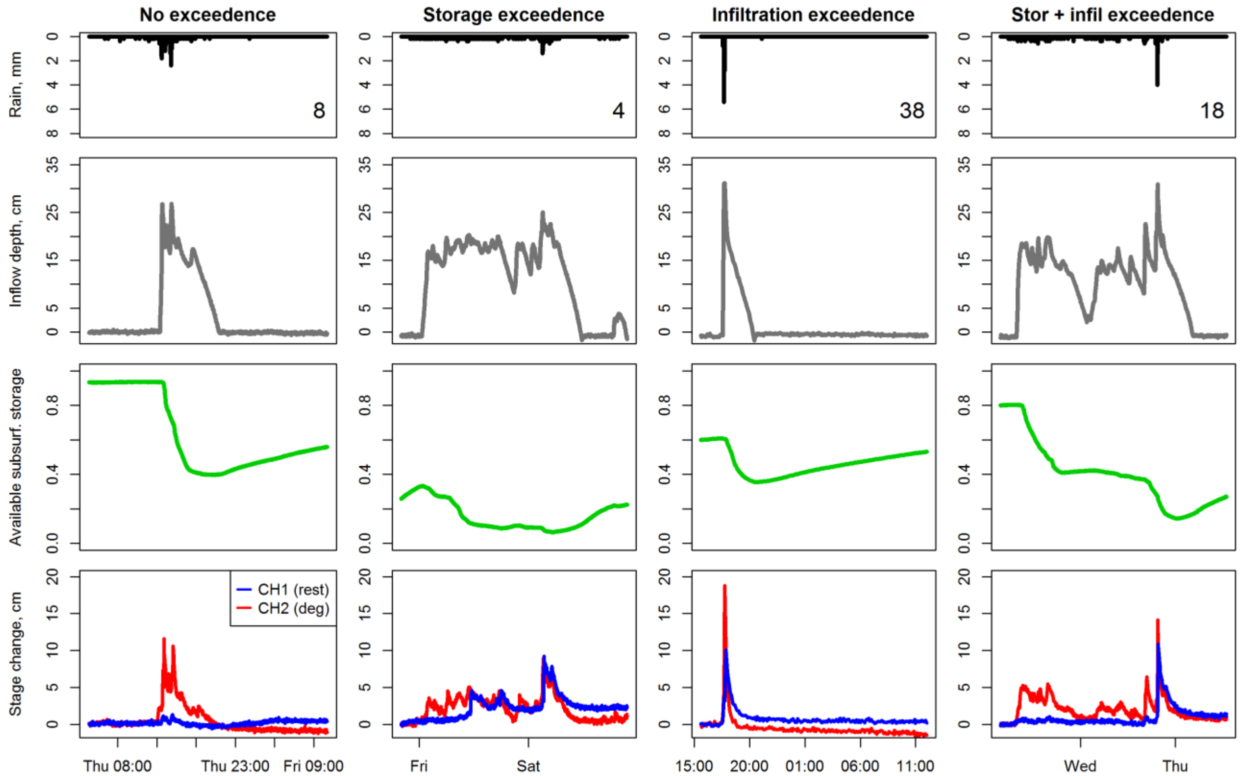

Figure 3.8: Surface and subsurface hydrologic responses during examples of the four types of runoff responses at CH1 (rainfall event IDs 8,4,38, and 18, respectively; see Table 4 for descriptions and Table 5 for event details). (**Top row**) Rainfall in 5-minute totals. Number on plot denotes the event ID. (**Second row**) Stage hydrograph of the inlet into the SCM structure (station location denoted with green arrow on Figure 1b). (**Third row**) Total observed subsurface storage in the seepage bed of the SCM at CH1over the course of the rainfall event. (**Bottom row**) Stream stage responses to the rainfall events for both CH1 and CH2 catchment outlets.

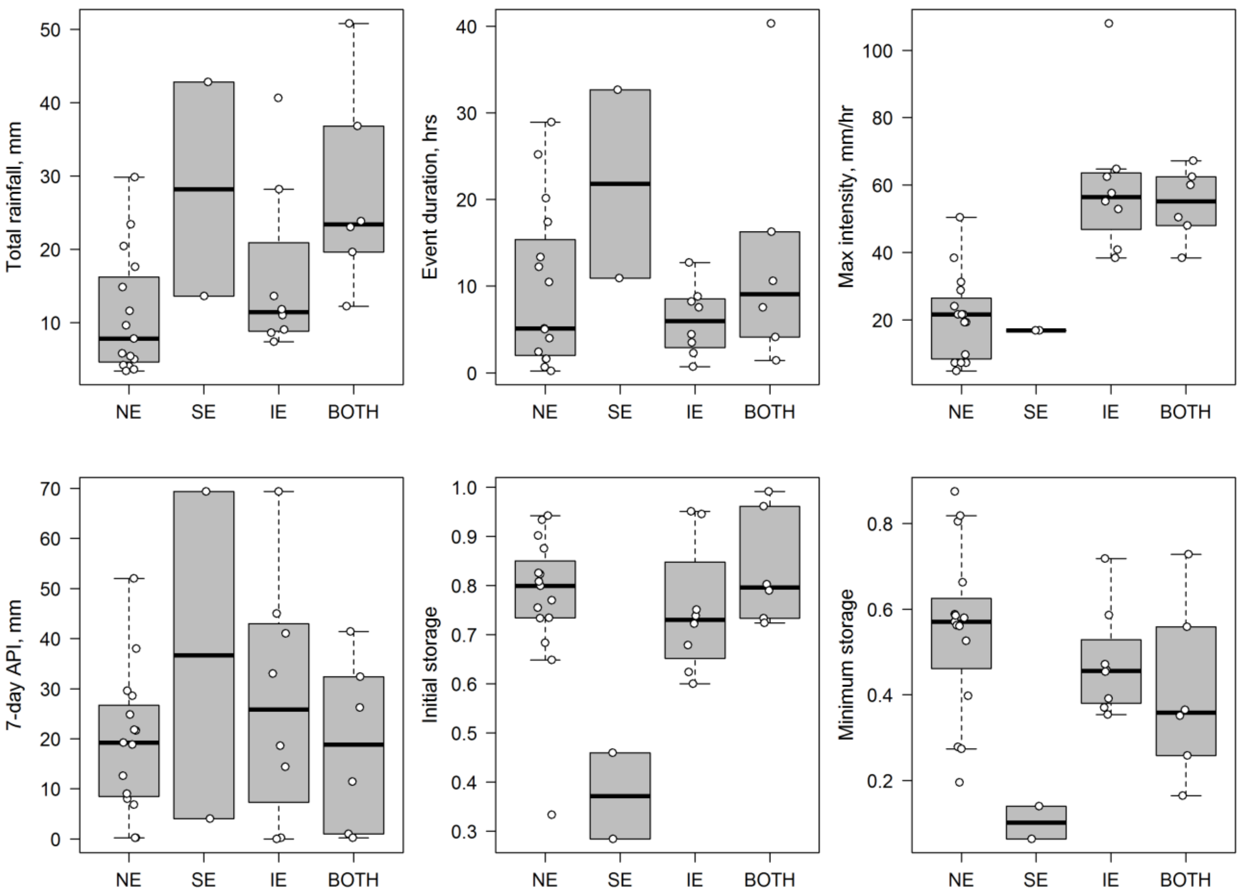

Figure 3.9: Rainfall characteristics of the four types of hydrologic responses. NE = no exceedance (CH1 response < 3 cm); SE = storage exceedance runoff response; IE = infiltration exceedance runoff response; BOTH = both storage and infiltration exceedance runoff response.

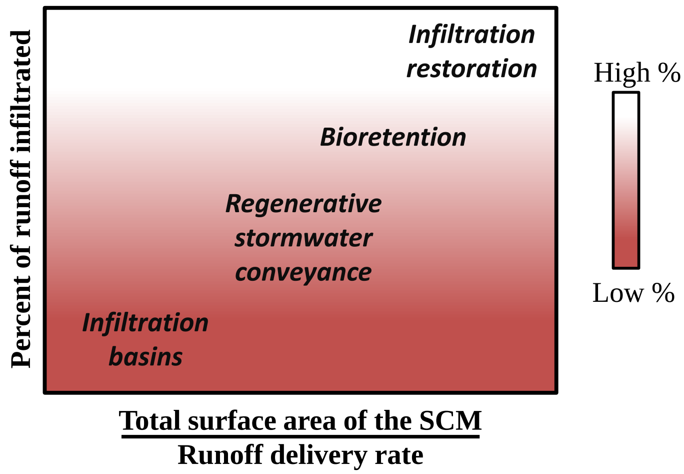

Figure 3.10: Conceptual diagram showing the relationship between SCM design (e.g., surface area) and the characteristics of its contributing area (runoff delivery rate) both control the overall hydrologic functioning of various infiltration-based SCMs.

Supplemental Information

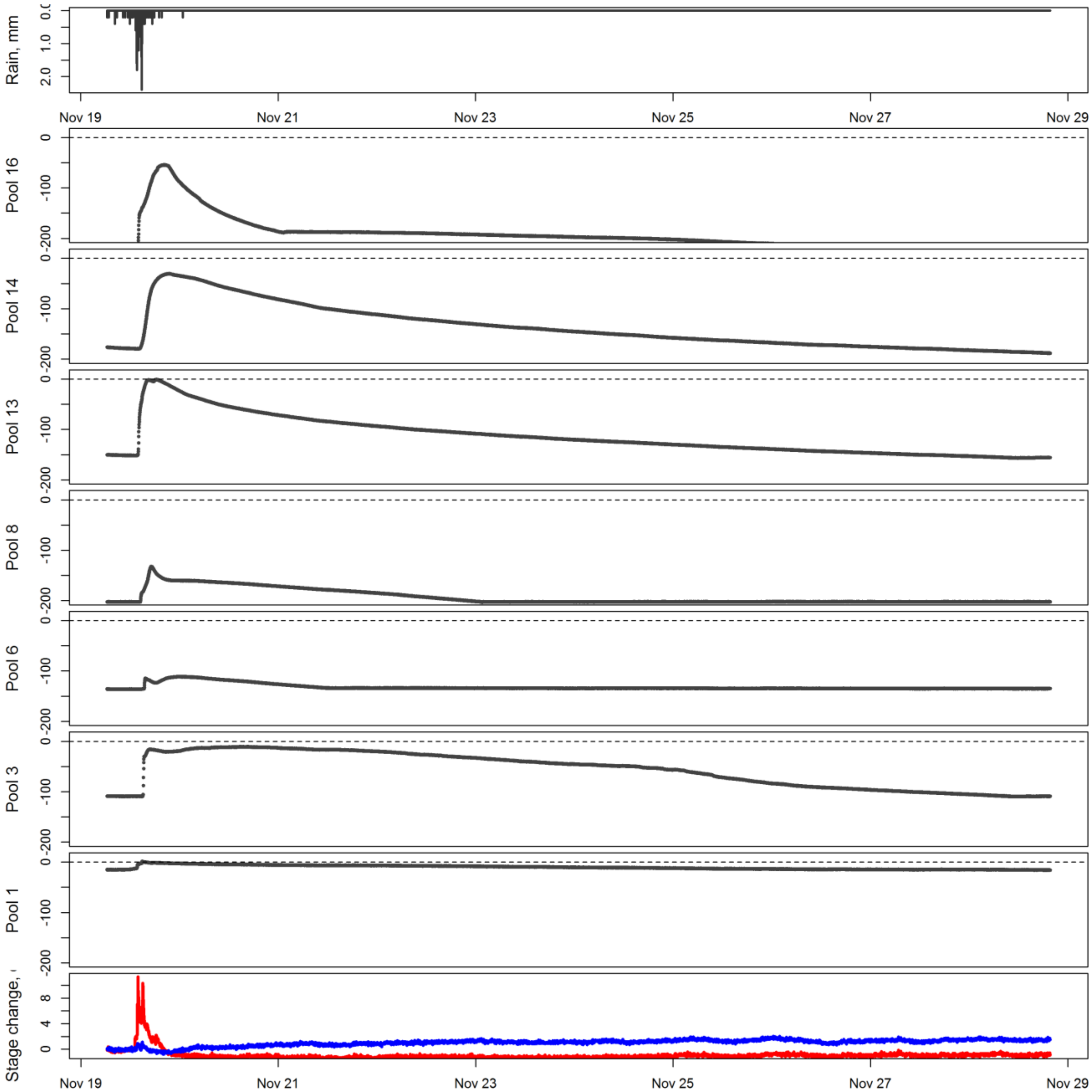

Figure SI-3.1. Water table fluctuations (depth below GSE, cm) for rainfall event # 13, November 19, 2015. Rainfall total = 23.4 mm; duration approximately 13 hours; maximum 5-min rainfall intensity = 28.8 mm/hr.

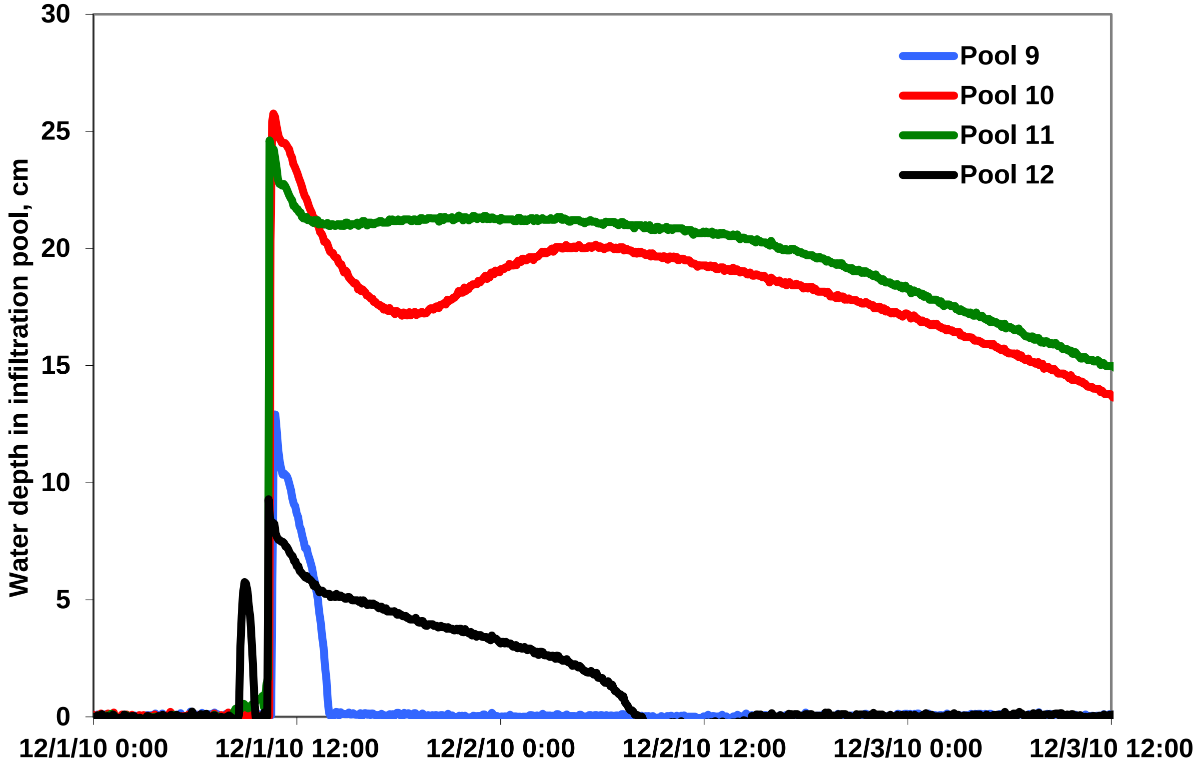

Figure SI-3.2: Variability in drainage rates for infiltration pools 9, 10, 11, and 12 after a rainfall event in 2010 (15 mm rainfall on December 1, 2010 from 6:45 to 11:45 am EST).

Table SI-3.1: Sediment grain size for select surficial (0-20 cm) samples from pools 14, 8, and 6

Pool ID	D10, mm	D50, mm	% sample finer than 0.5 mm
6	0.12	0.44	59
8	0.21	0.64	43
14	0.12	0.49	61
Mean	0.15	0.52	54

Conclusions

The objective of this dissertation was to investigate the effects of urbanization and infiltration-based watershed restoration on the hydro-ecology of Coastal Plain headwater streams. To accomplish this, I first examined the differences in hydrological responses in baseflow and stormflow across 11 headwater catchments with varying levels of urbanization and restoration status [Chapter 1]. Secondly, using the same suite of catchments, I quantified differences in water quality and aquatic insect composition across the sites, and explored the linkages between hydrology, water quality, and insect community composition [Chapter 2]. Finally, I used a paired-catchment approach, coupled with detailed groundwater monitoring, to identify potential mechanisms controlling the hydrological functioning of one of the infiltration-based restorations [Chapter 3].

In Chapter 1, I clearly detected the hydrological effects of urbanization across all hydrologic metrics used. I found reduced watershed storage, increased flowpath connectivity, and flashier flow regimes. Only one of the three urban-restored catchments exhibited partially restored hydrologic function, including a significantly larger minimum runoff threshold in comparison with other urban catchments. In this catchment (CH1), enhanced infiltration of stormwater runoff within the restoration structure increased storage of small rainfall events. However, baseflow in the stream draining this restored catchment remained low compared to the forested reference streams. This suggests that localized enhanced infiltration of stormwater runoff in the structure upstream of the channel head was insufficient to replace natural subsurface storage zones that contribute to stream baseflow. I hypothesized that the highly variable responses among the three urban-restored catchments were likely due to the spatial heterogeneity of urban development, including the level of impervious cover and extent of the storm sewer network, as well as catchment area.

In Chapter 2, I observed poor biodiversity in the aquatic insect community in all urban streams, regardless of whether their watershed had been restored with an infiltration-based SCM. Sensitive aquatic insect taxa, including Ephemeroptera, Plecoptera, and Trichoptera, were found predominantly in the forested reference streams. Although infiltration-based SCMs seemed to mitigate stream temperature surges during summer rainfall events, I observed no effect of SCM implementation on conductivity and chloride concentrations; these were driven by changes in watershed impervious cover alone. Dissolved oxygen concentrations were lowest and maximum summer stream temperatures were highest in the urban-restored streams, suggesting there may be negative, unintended water quality consequences from implementing these types of SCMs. Furthermore, I found conductivity, daily maximum summer temperatures, and runoff frequency explained the majority of variability in insect composition across the sites, suggesting that watershed-scale stressors from urbanization may be contributing to poor biodiversity, which are not adequately addressed by this SCM as currently designed.

Finally, in Chapter 3, I examined in detail the SCM project that indicated the most restoration of hydrological function (CH1). I identified several processes that could constrain the storage capacity of this type of SCM, including groundwater intrusion, lateral subsurface storm flow, and low hydraulic conductivity sediments in the subsurface seepage bed. Despite this, overall subsurface storage availability within the SCM remained fairly high most of the time, and the restored catchment completely intercepted runoff for small rainfall events (< 5mm) in comparison to the un-restored control catchment. However, I documented infiltration exceedance as the dominant mechanism controlling SCM hydrologic function, which occurs during high-intensity rainfall events (max 5-minute intensity = 20 mm/h or greater) with associated high runoff delivery rates into the structure. Runoff during these high-intensity events cannot be intercepted quickly enough within the structure to prevent the conveyance of untreated runoff into the stream, even though storage in the seepage bed remained available. I hypothesize that the unique placement of RSCs in topographic convergence zones may limit their hydrologic functionality when compared to other infiltration-based SCMs (e.g, bioretention basins). Given their location between the storm sewer network and the perennial stream channel, concentrated runoff from the contributing area may bypass the system under certain circumstances because their relatively small footprint may limit runoff interception during high-intensity rainfall events.

Evaluating the role of RSCs in the path forward to restore urban stream ecosystems

One of the many motivations of this dissertation was to better inform watershed managers investing in regenerative stormwater conveyances (RSCs), the type of watershed restoration strategy at the focus of this study. Aquatic biodiversity is a common biological metric for assessing overall stream ecosystem health (Karr 1999), and served as the ecological endpoint I used for this dissertation [Chapter 2]. In all three chapters, I documented the shortcomings of RSCs for restoring hydrology and reducing watershed-scale water quality stressors that are likely contributing to persistently poor aquatic biodiversity in urban Coastal Plain headwater streams. Although the concept of centrally-located, infiltration-based SCMs is promising, in reality there are limitations for this design to achieve its intended goals of serving an infiltration hotspot and returning runoff to subsurface storage [Chapter 3].

A central theme echoed throughout this dissertation was the need for restoration strategies to mimic, as closely as possible, the spatial scale and distribution of the initial ecological disturbance—in this case, the replacement of permeable soils and deep-rooted vegetation with impervious surfaces and turf grass (Schwartz and Smith 2016). This principle is embedded in the concept of low-impact development, as well as the application of green infrastructure (Dietz 2007). Low impact development, or ecologically sensitive planning, has the potential of minimizing hydrological disturbance from urbanization (Dietz 2007; Dietz and Clausen 2008; Yang and Li 2010). Likewise, the implementation of fully-distributed, infiltration-based SCMs has been successful in mitigating some hydrological changes from traditional urban development (Jarden and others 2016; Loperfido and others 2014; Shuster and Rhea 2013). However, the difficulty in widespread landowner adoption of these practices may still limit the effectiveness of these strategies in restoring hydrological patterns, water quality, and ultimately improving aquatic biodiversity (Roy and others 2014; Walsh and others 2015).

The results from this dissertation suggest that RSCs may be more effective at restoring hydrologic function to urbanized watersheds if their placement is limited to certain conditions. For example, the RSC that demonstrated the largest mitigating effect was located in a small, moderately urbanized catchment (5.3 ha; 22% impervious cover; [Chapter 1]). Another study on an RSC had shown promise in a lightly developed catchment (Cizek 2014). Given the infiltration exceedance limitations observed in this dissertation [Chapter 3], RSCs may perform best in watersheds with small contributing areas and low to moderate levels of imperviousness. However, it is unlikely that these structures themselves will recover the full hydrological cycle to the watershed, given that they are still sites of focused, rather than distributed, infiltration.

If RSCs were to be part of a watershed-scale strategy to improve stream biodiversity, they should also be coupled with in-stream geomorphic restoration to improve stream habitat only *after* hydrology and water quality improvements have been documented (Stranko and others

2012). Of course, I observed poor water quality in all three urban-restored streams [Chapter 2]. Some of these water quality issues were tied to initial urbanization (e.g., road salt application to impervious surfaces), and could be addressed by creating protection zones with reduced winter-time chloride loading in watersheds undergoing ecosystem restoration (Wallace and Biastoch 2016). However, additional water quality impairments may have actually been exacerbated by the restoration activity itself (low dissolved oxygen; high daily maximum temperatures in the urban-restored streams). If these factors are indeed limiting sensitive aquatic organisms from recolonizing, RSCs may not be the optimal method for addressing stormwater management. Finally, there remain dispersal limitations of aquatic organisms to urban streams if there is broad-scale impairment of streams (e.g., there are no source populations within dispersal distance). If the restoration goal is to increase biodiversity in a stream channel, there must be opportunities for dispersal from the regional meta-community (Smith and others 2015), which is why the preservation of undisturbed stream ecosystems should remain a high priority in all disturbed landscapes (Tonkin and others 2014).

It is important to note that the same restoration strategy may be used to address multiple ecological endpoints: for example, RSCs are approved best management practices for reducing sediment, nitrogen, and phosphorus loads to meet water quality attainment goals for the Chesapeake Bay restoration effort (www.epa.gov/chesapeake-bay-tmdl). Previous studies have shown their limitations in achieving these water quality goals (Filoso and others 2015; Palmer and others 2014a). Although I did not examine RSC effectiveness for reducing sediments and nutrients in this dissertation, the limited changes in hydrology I observed from RSC implementation may hamper any substantial reduction in pollutant loads normally associated with stormflow (e.g., sediments or phosphorus). Reducing nitrate loads in urban landscapes is even more difficult, given the complexity in sources and pathways to the streams (Filoso and Palmer 2011; Pennino and others 2016). Careful documentation of pollutant sources and pathways to stream channels are needed to optimize management strategies to address them (Shields and others 2008).

In summary, this dissertation improved our understanding of how RSCs, a type of infiltration-based SCM, modulated, or failed to modulate, the water quantity and quality impacts of urbanization, which likely hampers any recovery of sensitive taxa to their downstream ecosystems. Their limited success in doing so was largely due to their centralized design. Ideally, restoration practices should mimic, as closely as possible, the distribution of the processes lost through urbanization. Careful design of restoration strategies, which includes prioritizing the goals of the restoration and customizing the design to match that goal, is the best way to address the effects of urbanization.

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