Quantifying the ecological uplift and effectiveness of differing stream restoration approaches in Maryland

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EXECUTIVE SUMMARY

Urban stream restorations are an increasingly important tool to mitigate degradation due to human activities such as urban development and road construction. Despite the promise and allure of repairing damaged streams, there is little evidence for ecological uplift after a stream's geomorphic attributes have been repaired. Our over-arching goal in this report was to determine whether stream restoration activities produce ecological uplift compared to sections on the same stream that have not been restored. Within this context, we also explored the extent to which watershed land use, time since restoration, restoration activities, and several other co-varying factors might explain restoration success or lack thereof. A secondary objective was to determine realistic restoration goals by identifying the extent to which restoration activities could return a stream to conditions representative of lesser disturbed states.

We sampled 40 urban stream restorations across the Piedmont and Coastal Plain physiographic regions in the grater Baltimore/Washington DC Metropolitan area of Maryland. For each restoration we used a triplet design where samples were collected in restored sections as well as upstream in an unrestored section and downstream of the restoration. Samples for each stream triplet combination occurred on the same day to minimize temporal variation, and the triplet design itself minimized spatial variation that exists due to differing chemistry, topography, and land use history among watersheds. At each site, we collected benthic macroinvertebrate samples to measure the ecological response of restored sections compared to unrestored. We also collected geomorphic, chemical, and riparian information to assess restoration success from a physico-chemical perspective as well as to provide covariate information for explaining the ecological response.

We found substantially divergent results based on two broad success measures applied across the common set of restoration projects. Physical aspects of streams were substantially improved by restoration activities, particularly in the Coastal Plain. For these geomorphic components, we rate the restoration activities a success and believe that these projects largely achieved positive outcomes. Restorations stabilized streambanks and the channel, created more heterogeneous habitats, reduced fine sediment deposition, and improved conditions for the biota.

Unfortunately, the ecological aspects rarely improved despite the improved physical measures. We therefore reject our hypothesis that stream restorations improve overall ecological condition or even its subcomponents. Benthic macroinvertebrate communities in restored sections remained similar to unrestored sections on the same stream and were significantly dissimilar to MBSS Sentinel Sites. Similarly, the numerous metrics used in ecological assessments also showed a lack of response. The non-response even extended to measures examining urban-tolerant taxa, which we thought might show benefits. The few ecological improvements observed seemed idiosyncratic and could not be attributed to any broad classes of restoration technique or other attributes that we measured. There simply were few ecological differences between restored and unrestored sites. In fact, the unrestored sections upstream were often ecologically better than the restored sections or those downstream of restorations.

Our results suggest that restoration activities do not mitigate the reasons causing the ecological declines. Higher levels of Impervious Surface Cover (ISC) in the watershed has an overarching influence on Piedmont streams (but not in the Coastal Plain). Restorations actually decreased in ecological health measures to a greater extent as ISC increased than their unrestored counterparts upstream.

Ecological measures also responded negatively to the degree of disruption caused by the restoration. Longer restorations and those with more installed structures had lower ecological uplift measures in the Piedmont, while those in the Coastal Plain responded negatively to greater amounts of installed root wads and step pools. A key point here is that the amount or intensity of restoration did not improve outcomes in either region. The time since restoration completion partially mitigated these effects when focusing only on responses in restored sections, but it did not produce significant trends when compared against unrestored sections.

Within the Coastal Plain, Natural Channel Design (NCD)-dominant restorations had significantly higher ecological measures than Regenerative Stormwater Conveyance (RSC)-dominant approaches. However, streams chosen for RSC-dominant approaches seemed to already be in lower ecological condition based on their upstream samples. Therefore, we cannot make definitive conclusions. RSC restorations seemed to show a slightly better uplift response than NCD-dominant restorations, but this was complicated by a lack of true upstream controls for many RSC projects as they often extended to the headwaters. Additionally, most improvements were small and within the sampling error range.

We conclude there is little evidence that urban stream restorations can produce meaningful improvements in traditional measures of stream condition as measured with benthic macroinvertebrates. Unfortunately, the possibility of restoring the ecology of urban streams to resemble conditions of streams in lesser disturbed watersheds is limited.

INTRODUCTION

Many human activities negatively affect ecosystem health on a spectrum of spatial and temporal scales (Bernhardt and Palmer 2007; Hautier et al. 2015; Roy et al. 2016). Stream ecosystems are particularly sensitive because they occur in the lowest-lying areas in a landscape and thus are products of the land, collecting sediments and nutrients through run-off (Booth et al. 2016). The long flowpath of water from ridgetops to the channel means streams are influenced by both local and distant activities. Streams therefore receive, concentrate, and integrate all activities in the catchment upstream so that a watershed in poor condition produces a stream in poor condition (Hilderbrand et al. 2010; Hassett et al. 2018). The results of watershed actions may influence streams for decades via direct insults such as erosion (Harding et al. 1998) or by dispersal limitation for recolonization and recovery because streams are functionally and effectively linear landscapes (Hilderbrand and Utz 2015). Thus, few freshwater streams in the United States remain in undisturbed condition (Benke 1990), and nearly half are considered impaired or polluted (United States Environmental Protection Agency 2009).

An increasing number of stream ecosystems are affected by urbanization. Urbanized landscapes tend to replace natural features with impervious surfaces, which dramatically alters hydrological characteristics to make flood pulses higher and more frequent while also reducing base flows and increasing stream temperatures, nutrients, and sediments (Walsh et al. 2005; Sharp 2010; Vietz et al. 2015; Booth et al. 2016; McGrane 2016). The altered hydrology alters geomorphology to erode banks and simplify stream channels (Booth et al. 2016). These general hydrology, geomorphology, and chemistry changes are symptoms of what is commonly referred to as an "Urban Stream Syndrome" (Walsh et al. 2005; Booth et al. 2016) that ultimately affects the in-stream physical, chemical, and hydrologic habitats and thus, the biology within aquatic systems. Urban streams often have low values for species diversity, trophic complexity, and habitat diversity, as well as, altered food webs and community structure (Walsh et al. 2005; Booth et al. 2016; Hassett et al. 2018). Even small amounts of land alteration can limit sensitive taxa (Violin et al. 2011; Stranko et al. 2008; Utz et al. 2010), and areas of high urbanization can eliminate nearly 50% of aquatic biodiversity (Hilderbrand et al. 2010; Johnson et al. 2013).

Benthic macroinvertebrate assemblages occurring in highly degraded streams within urban catchments are often dominated by a few genera of oligochaetes and chironomids and little other biodiversity (Walsh et al. 2005; Brown et al. 2009). Strong negative correlations between urbanization and sensitive Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa are shown consistently in assessments conducted in urban catchments (Brown et al. 2009; Utz et al. 2010; Williams et al. 2010). Streams in lesser disturbed watersheds regularly have predictable assemblages containing many functional feeding groups (Vannote et al. 1980), while urban streams contain fewer feeding groups. Thus, life in urban catchments is difficult for invertebrates: small and homogeneous substrate in urban streams reduces habitat (Vietz et al. 2015), fluctuating temperature regimes stress their physiologies, flashy hydrographs scour and wash away individuals, pollutants poison organisms, and increased nutrient and sediment pollutant concentrations inhibit functioning invertebrate assemblages (Wenger et al. 2009).

Efforts to reduce and repair anthropogenic effects to aquatic systems are becoming commonplace in resource management. The most important regulatory vehicle to protect streams is the Clean Water Act (CWA), which establishes a structure for regulating pollutants entering the waters of the United States and regulating quality standards for surface waters. Recent revisions in 2008 and 2015 have included headwater streams and wetlands for their influence downstream (Section 404, Clean Water Act) and makes it unlawful to discharge pollutants into any body of water protected under the CWA unless a permit is obtained through the EPA and Army Corps of Engineers (Title 33 Clean Water Act). Specific steps noted in section 404 are to avoid, minimize, and mitigate degrading impacts on protected water resources. Compensatory mitigation is the third step in this process if avoiding or minimizing impacts is unavoidable.

Mitigation in the form of stream restoration has become an important tool to protect human infrastructure and repair damage. As human population increases in the United States, so does the need for methods to repair and limit human damages to streams because quality waters are vital for human health, recreation, and ecosystem services that provide ecological and economic value. Stream Restoration, as defined by Kauffman et al. (1997) is the "Reestablishment of processes, functions, and related biological, chemical, and physical linkages between the aquatic and associated riparian ecosystems; it is the repairing of damage caused by human activities". Over the last 40 years, stream restoration projects have increased exponentially on a national scale and in response to the increase of human population. Over 14 billion dollars were allocated toward stream restoration since between 1990 and 2005 (Bernhardt et al. 2005), with the highest density of projects occur in the Chesapeake Bay watershed (Palmer et al. 2010). The State Highway Administration of Maryland, for example, is a common mitigation permittee that implements many restorations in the immediate Chesapeake Bay watershed to balance losses due to the effects of highways.

Restoration goals vary widely per project and region, but most aim to reverse physical degradation through some combination of: reconnecting streams to their floodplain, reducing erosion during high flows, reshaping the channel, inducing substrate and habitat heterogeneity, and replanting riparian vegetation. (Hassett et al. 2007; Palmer et al. 2014). However, in many urban centers, space is frequently limited and land ownership and acquisition can be complicated resulting in river restorations that focus exclusively on bank stabilization to protect infrastructure (Bernhardt & Palmer 2007; Lake et al. 2007; Wohl et al. 2015). Many projects thus rely on a 'Field of Dreams' approach (Palmer et al. 1997) that assumes that altering stream geomorphology and habitat to mimic stable conditions will lead to the recovery of native biodiversity. By addressing the base of the functional pyramid (Harmon 2012), the conditions are set for ecological recovery. However, there is little evidence that restoring physical structure alone is sufficient (Hilderbrand et al. 2005; Violin et al. 2011; Stranko et al. 2012).

Despite the ecological shortcomings of urban stream restorations, there remains a need to repair damaged and degraded streams. Excess sedimentation and nutrients are a main concern in the Chesapeake Bay watershed and contribute to multiple dead zones and habitat degradation. Degraded and eroded urban streams are often in need of bank stabilization and floodplain reconnection to reduce sedimentation from high flows. In compacted urban centers, sewer lines, culverts, and necessary anthropogenic structures become degraded over time that can be mitigated during restoration efforts. The aesthetic value of restored streams can lead to increases in property value and stimulate local economy (Kenney et al. 2012). Total daily maximum load (TMDL) mandates in the Chesapeake Bay watershed, require reductions in nutrient inputs that may be achieved through restoration. For these reasons, stream restoration efforts will continue regardless of ecological outcomes.

Throughout the Chesapeake Bay watershed, natural channel design (NCD) and regenerative stormwater conveyance (RSC) are important approaches toward restoration (Palmer, Filoso, et al. 2014; Sudduth et al. 2011; Walsh et al. 2016). Natural channel design (NCD) is a common (and often criticized) tactic aimed at reforming the profile and dimensions of a degraded stream channel to resemble a pre-disturbed state (Rosgen 1996; Simon et al. 2007). Most projects using NCD are extremely invasive in the stream channel and surrounding riparian zone and employ heavy machinery to reshape, regrade, and sometime re-locate the stream channel while adding hard structures such as logs and cross vanes to maintain gradient. Root wads and coarse substrate are placed to add bank stability and riffles creating a newly constructed stream to resemble natural reference streams (United States Department of Agriculture (USDA) & National Resources Conservation Service (NRCS) 2007). NCD restorations manipulate the landscape a great deal and often result in streams that look artificial. In urban settings, however, the effects of channelization and decades of sedimentation often require intensive alterations to reverse channel degradation (Roy et al. 2016). The critical assumption with NCD is that streams with similar morphology will behave similarly (Simon et al. 2007). NCD approaches can be successful (Roni et al. 2008), which is why the method is so widespread. However, the over-application in a cookbook style (Hilderbrand et al. 2005) has also resulted in disappointments, which is why NCD is also commonly criticized (Lave 2009).

RSC and stream-wetland approaches are rapidly gaining popularity, particularly in the Coastal Plain and in heavily impacted urban areas where mimicking natural landscapes and processes are not feasible options (Palmer, Filoso, et al. 2014). RSC approaches capture stormwater runoff to dampen its downstream pulses, in contrast to historical stormflow management which sought to transport water quickly to the stream to minimize property damage during storm events (Williams et al. 2017). Runoff is dampened by creating a series of in-channel step pools and seepage wetlands filled with native plants that decrease water velocity and allow for settlement of suspended solids and recharge of groundwater (Williams et al. 2017). The process of slowing discharge also leads to nitrogen removal via denitrification by microbes (Wenger et al. 2009).

Design approaches for stormwater conveyance systems vary, but are generally categorized into dry channel and wet channel RSC systems (Berg et al. 2012). Dry channel RSC's are placed at storm drain outfalls in an eroded gully or ephemeral stream. The primary use of this type of RSC is strictly stormwater management as they are built above the water table and only flow during storm events; these ephemeral channels are devoid of fish. Wet Channel RSCs focus on returning stormwater flow to the perennial stream channel through step pools leading to sand or wetland seepage systems that leach into the substrate. RSC restoration projects attempt to reduce runoff degradation and enhance stream functions in a realistic approach, sacrificing immediate biodiversity to improve conditions in surrounding and downstream ecosystems (Williams et al. 2017).

Similar to NCD, both dry and wet-channel RSC approaches are highly intrusive and heavily modify the stream channel by design for maximum effectiveness. RSC approaches use heavy equipment to excavate and construct the depositional pools. Riparian areas may also be affected by root compaction or tree removal in some cases (Berg et al. 2012). Many RSC projects in Maryland occur in heavily eroded stream channels, and substantial filling is required to raise the channel grade to its previous levels. In the process, substantial amounts of sand, wood chips, and other organic materials are layered in the substratum to form a bioreactor matrix to fuel the microbial activities that remove nitrogen from the discharge. The added organic material in RSC systems is designed to increase microbial activity when storm flows add Nitrogen (N), Phosphorus (P), and other organic nutrients into the stream channel (Williams et al. 2017). At base flow conditions, even wet-channel RSC systems can be reduced to stagnant anaerobic pools that become so low in dissolved oxygen, macroinvertebrate and fish assemblages are eliminated, thus reducing biological integrity. Under these conditions, however, microbial denitrification may be enhanced (Wenger et al. 2009).

Whereas evidence for ecological recovery is limited with urban NCD restorations, the relative effectiveness of RSC approaches is mostly unknown and requires further research (Palmer et al. 2014). RSC restorations reduce sediments, nutrient loads, and peak flows for smaller storm events as designed, but reductions for medium and large events are minimal and perhaps worse than pre-restoration states as net exporters (Filoso et al. 2011; Filoso and Palmer 2011; Filoso et al. 2015; Cizek et al. 2018; Koryto et al. 2017). Thus, information on ecological performance is needed in order to weigh potential benefits of RSC approaches against the invasiveness and limited performance with respect to TMDL measures.

Uplift is an important concept in measuring restoration success, and improvement from the unrestored state in any form for an ecosystem is a step in the right direction. Concerns over the possible uplift and recolonization of biological communities in urban stream restorations remain a prevalent research question (Wenger et al. 2009; Walsh et al. 2016; Hassett et al. 2018). Despite years of monitoring and decades of restoration projects, the general effectiveness of restoration approaches remains controversial. Much has been published on stream restorations and ecological recovery (Violin et al. 2011; Stranko et al. 2012; Palmer et al. 2014; Swan and Brown 2017), but few projects have sufficient sample size and breadth to provide statistical rigor to provide broader generalizations.

Benthic macroinvertebrate sampling is perhaps the most widely used mode of biological monitoring in aquatic environments (e.g. Hassett et al., 2018; Selvakumar, O'Connor, & Struck, 2010; Stranko et al., 2012; Swan & Brown, 2017; Walsh, Waller, Gehling, & Mac Nally, 2007) because they exhibit many valuable properties. Most macroinvertebrate taxa have limited migration patterns, and their often sessile life habits make them valuable in restoration impact research (Barbour et al. 1999). A broad spectrum of generalist and specialist taxa allow for pollution tolerances to be established and tested through assessment. Stream benthos are easily sampled with inexpensive materials and minimal need of personnel. Benthic assemblage responses tend to show sensitive taxa disappear in areas within urban watersheds (Hilderbrand et al. 2010; Stranko et al. 2012), while those occurring in highly degraded streams within urban catchments exhibit low species richness (Walsh et al. 2007; Booth et al. 2016). Since macroinvertebrate assemblages have been so widely studied and show consistent and measurable community responses to urbanization, they are a useful group for assessing aquatic ecosystem responses to urban stream restorations. Stream restoration is a relatively new concept, and it will be both ecologically and economically important to understand the effectiveness of techniques to improve ecosystem health and function. Because geomorphology, hydrology, and thus stream habitat is altered during stream restoration construction, physical habitat assessments are necessary in linking biota response to restoration tactics.

The over-arching goal of this research was to determine whether stream restoration activities produce ecological uplift compared to sections on the same stream that have not been restored. Within this objective, we also explored the extent to which watershed land use, time since restoration, and several other co-varying factors might explain restoration success or lack thereof. A secondary objective was to determine realistic restoration goals by identifying the extent to which restoration activities could return a stream to conditions representative of lesser disturbed states.

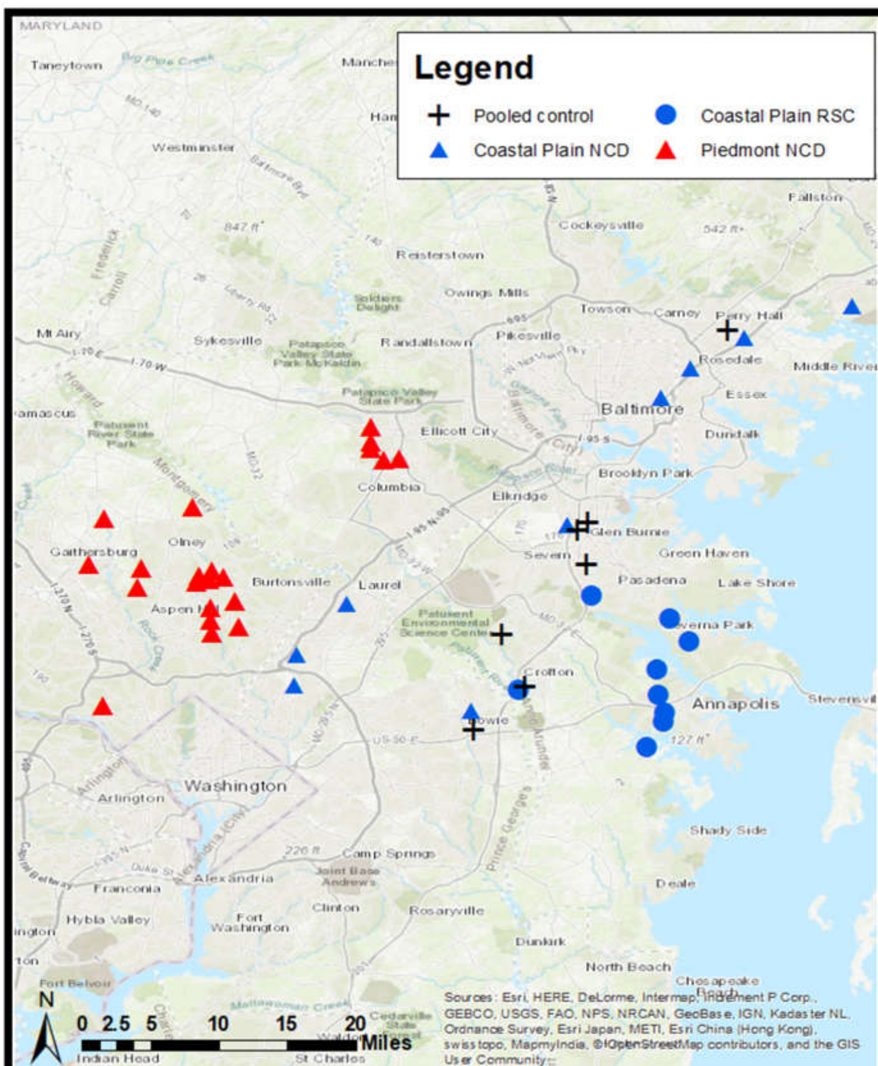


Figure 1. All 40 restoration sites separated by physiographic province and restoration type

METHODS

Study area

Data were collected from 40 stream restorations from urban localities spread over two physiographic regions within Maryland (Figure 1). Twenty-two stream restorations were selected from the Piedmont region and 18 from the Coastal Plain region. Within the Piedmont, all sites occured in Montgomery and Howard counties, while restorations in the Coastal Plain are spread across Anne Arundel, Baltimore, Harford, and Prince George's counties because few sites met the criteria for inclusion. Stream restorations were selected primarily by the availability of unrestored sections upstream and downstream of the restoration. Selected restoration projects in the Piedmont region were all completed using NCD methods, while selected projects in the Coastal Plain were split between NCD and RSC approaches. Watershed characteristics for each segment were calculated through USGS StreamStats v3 using GPS points taken at every sampled site. Attributes for each of the selected sites, including restoration length, age, and landscape setting are shown Appendix A.

Study Design

For each selected restoration site, data were collected in a "triplet" approach (Figure 2). This design approach allowed us to collect and compare data from segments within the same stream, highlighting the differences between unrestored sections to the restored. Conditions in the unrestored upstream reach may be comparable to ecosystem conditions in the absence of restoration activities, while conditions downstream may show if stream restorations affect the ecosystem outside of the actual areas restored. Assessment reaches were 75-meter-long segments within areas upstream of the restored reach, within restored, and downstream of restored. Several sites within the Coastal Plain were restored to the headwaters and thus, do not have an unrestored upstream reach. To mitigate this and keep data consistent, we sampled unrestored streams with similar watershed attributes to act as a comparable upstream reach reference. These sites are referred to as 'pooled sites' in our data analyses as they form a sampling "pool" of sites. Sites within this sampling pool are all compared against each of the restoration sites that do not have an upstream control section.

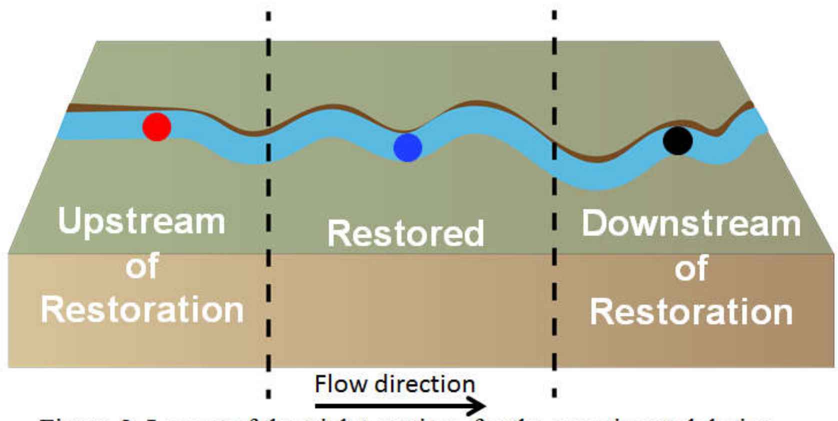


Figure 2. Layout of the triplet sections for the experimental design.

Field Sampling

All data were collected consistent with the Maryland Biological Stream Survey (MBSS) protocol for ease of design structure, consistency, and eventual data collaboration. Field technicians and team members are all MBSS trained, certified, and capable of collecting data to the required standard. The MBSS has developed and used an effective sampling protocol to assess conditions of ecological systems in Maryland streams since the mid 1990's (Stranko et al. 2017).

Benthic Macroinvertebrate samples were collected during the 2016 spring index period at each site following MBSS protocol, comprising 20ft^2 of multi-habitat sample using a D-frame net (Stranko et al. 2017). Samples were sorted and identified to the lowest possible taxon. Due to the sheer number of samples and effort in identification, samples were divided by physiographic province with Piedmont sites being sent to EcoAnalysts Inc. for identification and Coastal Plain sites being processed by staff at UMCES Appalachian Laboratory. Benthic samples were randomly sorted using the EPA standard grid system. All benthic samples were sorted into one 300-organism subsample, which differs slightly from MBSS protocol who sort to two 100-oganism subsamples. The 300 organism samples were compared against 100 count MBSS samples by bootstrapping randomized subsets to determine an average of the response statistic.

Physical habitat and water chemistry (Table 1) were also measured at each sampling site to provide supporting data to the benthic macroinvertebrate findings as well as to determine if there were physico-chemical differences in the restored sections. The MBSS protocol partitions habitat assessments into spring and summer sampling seasons, but we combined them into one sampling season to coincide with the benthic invertebrate sampling since there was no summer sampling planned. In addition to the MBSS habitat measures, pebble counts were included in the assessments to gain an understanding of the proportion of particles sizes within active riffles, as well as, size distribution to indicate stream channel roughness, and the USEPA Rapid Biological Assessment habitat assessment (Barbour et al. 1999) was completed for all sites. We used a Hydrolab to collect basic water chemistry data such as conductivity, dissolved oxygen, and pH at each site.

Table 1. Physical and chemical attributes measured for each sampling section.

Section Attributes		Description
Habitat and Riparian: Scored 1 (low) -20 (high)	Epifaunal Substrate/Available Cover	The relative quality and variety of natural structures such as cobble, logs, and refugia.
	Embeddedness	The extent to which stream substrate and snags are covered and sunkin into silt.
	Velocity Depth	Presence and quality of all 4 pool water velocity patterns (slow/fast-deep/shallow)
	Channel Alteration	A measure of large-scale changes of channel shape and structure.
	Sediment Deposition	A measure of the amount of sediment accumulating in pools in the form of bars or islands.
	Riffle Frequency	The relative occurance of riffles within the stream reach.
	Channel Flow Status	The level in which water fills the channel.
	Total Bank Stability	The extent of erosion on stream banks. Each bank evaluated separetly.
	Total Bank Vegetative Protection	The amount of vegetative protection to the stream bank. Each bank evaluated separetly.
	Total Width Undisturbed Vegetative Zone	The width of natural vegetation from the stream bank through the riparian zone. Each bank
	Pool Substrate Characterization	The type and condition of pool substrate. More heterogeneous substrate yields higher score.
	Pool Variability	Presence and quality of all 4 pool water depth patterns (small/large-deep/shallow)
	Channel Sinuosity	The level of meandering of the stream channel.
	Dominant Surrounding Land Use	Categorization of immediate land use
	Distance to nearest Road (m)	Distance from stream channel to road in meters.
	Total Riparian Veg. Width (m)	Width of riparian region in meters
	Total # LWD	Quantity of large woody debris within stream reach.
	Trash Rating	Evalution of relative trash pollutants in stream reach.
Substrate Composition	% Fines	Percentage of fine particles within pebble counts
	% Sand	Percentage of sand particles within pebble counts
	% Gravel	Percentage of gravel sized substrate within pebble counts
	% Cobble	Percentage of cobble sized substrate within pebble counts
	% Boulder	Percentage of boulder sized substrate within pebble counts
	% Bedrock	Percentage of bedrock within pebble counts
	% Artificial	Percentage of artificial within pebble counts
	D16	The diameter of the substrate within the 16th percentile.
	D35	The diameter of the substrate within the 35th percentile.
	D50	The diameter of the substrate within the 50th percentile.
	D65	The diameter of the substrate within the 65th percentile.
	D84	The diameter of the substrate within the 84th percentile.
	D95	The diameter of the substrate within the 95th percentile.
Water Chemistry	Water Temp (°C)	Various water chemistry measures collected via hydrolab.
	Specific Conductance (µS/L)	
	Dissolved Oxygen (mg/L)	
	pH	
	Flow (cfs)	Water flow taken via flow meter.
	Flow Regime	Categorization of flow status during sample period.
	Water Width (ft)	Width of water in channel in feet.

In addition to the physical and chemical measures, we obtained and extracted information from the restoration plans and 'as-built' surveys to quantify the structures and methods used during construction (Table 2). These measures provide a larger habitat context than our physical surveys could provide and were used in our statistical analyses. Since many restorations are site specific and vary per tactic, region, and engineering organization, the quantity of in-stream structures is a reasonable scale to analyze as-built surveys. The use of as-built surveys was not originally planned, but was incorporated in part because of comments received during the annual regulators meetings. We also included the as-built portion because we realized that our ideas on whole-stream metabolism were not practical to implement in a rigorous fashion due to the scale and extent of the study design. While we were able to obtain as-built surveys for most restorations, we are missing some sites. Obtaining the as-built surveys took well over a year and a lot of persistent follow-up; we did not receive several until January 2019. We knew they were coming, but were delayed for nearly two months before reaching us. We felt they might very important for explaining uplift and decided to risk a late report submission in order to produce the best analyses possible.

Table 2. Attributes of restoration activities within restored sections. The numbers of structures were tabulated from 'as-built' surveys when available or design plans and represent the total upstream of the sampling point within each restored section.

Restoration Attribute	
# Cross Vanes	# Log Cross Vanes
# Rock Vanes	# Root Wads
# Log Vanes	# Step Pools
# Total Vanes	# Total Structures
# Rock J-Hook	
# Log J-Hook	Restoration Age (years until sampling)
# Total J-Hook	Restoration Length (m)

Statistical analyses

We used an ANOVA / ANCOVA analysis structure as our primary method to compare differences among the restored, upstream control, and downstream of restored sections. Each stream served as a blocking factor to constrain comparisons to within each triplet combination in order to minimize variation among watersheds. Our principle focus was to test for differences between the restored section and the upstream control and to identify the direction of any differences found. Differences detected between the upstream control and the section downstream of the restoration were designed to detect the presence of ecological uplift downstream of restored sections as a value-added effect. Piedmont and Coastal Plain restorations were analyzed separately.

Our analyses were conducted at two hierarchical levels. The main hypothesis test examining differences between restored and upstream or downstream sections used an ANCOVA with the covariates ISC, Restoration Age, and Restoration Length to assess differences among sections and whether the covariates interacted significantly with each section of the triplet to produce a differential response. That is, did ISC, Restoration Age, or Restoration Length cause the restoration to respond differently than upstream or downstream sections, and in what way. However, neither the upstream or downstream sections have a restoration length or age, nor do they have many other covariates such as installed structures. Therefore, we conducted a second analysis that focused only on restored sections to test for restoration attributes promoting or inhibiting ecological uplift. This second analysis used a regression model building approach. We fitted all possible models and used the ΔAIC to identify the best models. Models with low ΔAIC relative to the best performing model were considered to be alternative hypotheses to explain measures of ecological uplift.

Because seven of our sites in the Coastal Plain were missing upstream controls due to the restoration running to the headwaters, we used a model averaging approach to test for difference among upstream, restored, and downstream sections. Each stream not containing an upstream control was matched with a pooled site and the analysis conducted. We did this seven times with a different pooled site substituted as the upstream control and the results averaged. This average modeled response was used to assess the statistical significance of the independent (predictor) variables.

We tested a number of response variables (Table 3) to assess the ability of stream restoration activities to improve benthic macroinvertebrate taxon and community attributes above those found in the upstream controls. Response variables were mostly variations of the Benthic Index of Biotic Integrity (BIBI) metrics or the IBI itself. We also included responses designed to assess different components of recovery. Nonchironomid taxon richness (Intolerant Richness) assessed the recovery of pollution-sensitive taxa. Conversely, taxon richness of Urban-Tolerant taxa was analyzed to assess the ability of biodiversity to improve even if the pollution-sensitive taxa were not physiologically capable of inhabiting a stream. In effect, this was a test of whether the restoration actually improved conditions, but not substantively enough to allow for sensitive taxa to recover. We also included physical attributes as response variables to assess differences due to restoration activities. From the individual habitat measures, we calculated a Physical Habitat Index (PHI) score following the EPA method (Barbour et al. 1999). The PHI is similar to other indices in that it provides an overall estimate of the habitat quality at a site and is analogous in nature to the BIBI calculated for benthic macroinvertebrates.

Table 3. List of Response Variables analyzed for differences in uplift among restored, upstream, and downstream sections.

Response Variable	Description
Physical Habitat Index	An index of habitat using all in-channel habitat metrics from EPA rapid bioassessment protocol. Score is calculated as a percentage by dividing all habitat metric values per site divided by highest possible value. Habitat metrics are list in Table 1.
Total Richness	Number of differing taxa present in sample
EPT Abundance	Total number of Ephemeroptera, Plecoptera, and Trichoptera individuals
EPT Richness	Number if differing Ephemeroptera, Plecoptera, and Trichoptera taxa present in sample
Non-Tolerant Taxa Richness	Number of differing taxa of low urbanization tolerance. This corrects for possible total richness infliation from tolerant taxa (e.g. Chironomid, Oligochaeta)
Urban-Tolerant Taxa Richness	Number of differing taxa insensitive to Urbanization
Shannon-Wiener Index	Index that characterizes species diversity in a community. It accounts for both abundance and evenness where p is calculated as the proportion of each species in sample. Index = $-\sum(n_i/n)\ ln(n_i/n)$
Margalef Richness	Relative measure of species richness $(S\text{-}1)/\ln(n)$, where S is the number of taxa, and n is the number of individuals in sample.
Benthic Index of Biological Integrity (BIBI)	Maryland Biological Stream Survey (MBSS) Index of biological integrity. Metrics for BIBI's differ between physiographic region. Appropriate metrics were used for sites in the Coastal Plain and Peidmont.

In addition to the metrics of stream health, community-level analyses of the benthic macroinvertebrate data were performed. These are necessarily multivariate measures, and we used non-metric multidimensional scaling (NMDS) for visualization purposes. ADONIS, which is similar in practice to ANOVA for community-level data, was used to formally test for differences in community structure in pairwise comparisons (restored vs upstream, restored vs downstream, upstream vs downstream). All ADONIS analyses were set up with each stream serving as a blocking factor to minimize watershed differences. We also included benthic macroinvertebrate community data from the MBSS Sentinel Sites for Piedmont and Coastal Plain streams to compare communities at restored sites against what is representative of least-disturbed conditions. The MBSS Sentinel Sites were not on the same streams as the restoration and violates our triplet design. To incorporate the MBSS Sentinel Site data as well as the pooled sites for coastal plain restorations not having upstream controls, we implemented a bootstrapped resampling approach within the ADONIS analyses.

The bootstrapped resampling technique consisted of running the ADONIS analysis 1,000 times. Each iteration randomly drew an MBSS Sentinel Site from the appropriate region (Piedmont or Coastal Plain) and paired this site with one of the streams in our triplet to form a 'quadruplet' combination (upstream, restored, downstream, and MBSS). This quartet of samples was processed in the ADONIS analysis, and an F-value for each of the pairwise comparisons was generated. If the 95% Confidence Interval for the F-value of a specific pairwise comparison (e.g., restored vs upstream) across the 1,000 iterations was greater than 8 for the Piedmont and 8.3 for the Coastal Plain, we considered the comparison to be statistically significant as these values correspond with the Critical F-value adjusted for the six possible pairwise combinations to control the experimentwise error rate. We performed this adjustment because the same data are used for every pairwise comparison of the upstream, downstream and restored sections. Thus, each result within the bootstrap is identical when sections other than Sentinel Sites are compared. For MBSS vs sections comparisons, MBSS Sentinel Sites were sampled with replacement, so the same site could be applied to multiple triplets within any given iteration. These results yield a mean and a confidence interval. Comparisons of sections against MBSS Sentinel Sites were considered statistically significant if the confidence interval for the result did not encompass 1, which is the expected F-value under the null hypothesis of no difference.

Our final set of analyses placed the restorations into the context of what is realistic to expect. Here, we calculated BIBI scores for each restoration, bootstrapped via resampling for 1,000 iterations to the 100 bug count that the MBSS program uses. We calculated the mean resampled BIBI score to serve as the basis for calculating ecological uplift. Realized uplift was defined as the difference between a stream's upstream unrestored control and the restored section. Potential uplift was defined as the difference between an unrestored stream and the highest level of attainment achieved by a restored stream within a region. Maximal uplift was determined by estimating the difference between each stream and the average, minimally disturbed stream as represented by MBSS Sentinel Sites for that region.

RESULTS

Piedmont Benthic Macroinvertebrate Community Structure

Within the Piedmont region, the benthic community structure was not different among restored, upstream, or downstream reaches in our ADONIS analyses. However, MBSS Sentinel Sites are very different from all treatment reaches (ADONIS, F>10, P<0.01 for all comparisons; Table 4). The NMDS representation of benthic community structure in the Piedmont (Figure 3) shows clear separation of Sentinel Sites from everything else, but it is impossible to graphically visualize the within-stream effects that the statistical analyses incorporate when comparing among sections while controlling for the effects of each stream. When community structure is limited to only those taxa that are tolerant of urban conditions, the same patterns hold. Community structure in MBSS Sentinel Sites remain dissimilar to all sections within restored urban streams. Once the stream-to-stream variation is included, the differences are very pronounced as shown in the ADONIS results.

Table 4. ADONIS analysis F-value results comparing the community similarity of Restored, Upstream, Downstream, and MBSS Sentinel Sites. The MBSS Sentinel Site comparisons were generated using the bootstrapping approach described in the Methods. **Bolded values** with an asterisk indicate statistically significant differences between the pairwise comparisons.

	Restored vs Downstream	Upstream vs Downstream	Restored vs Upstream	Sentinel vs Upstream	Sentinel vs Downstream	Sentinel vs Restored

Piedmont	0.64	1.17	0.62	**12.6***	**13.99***	**17.38***

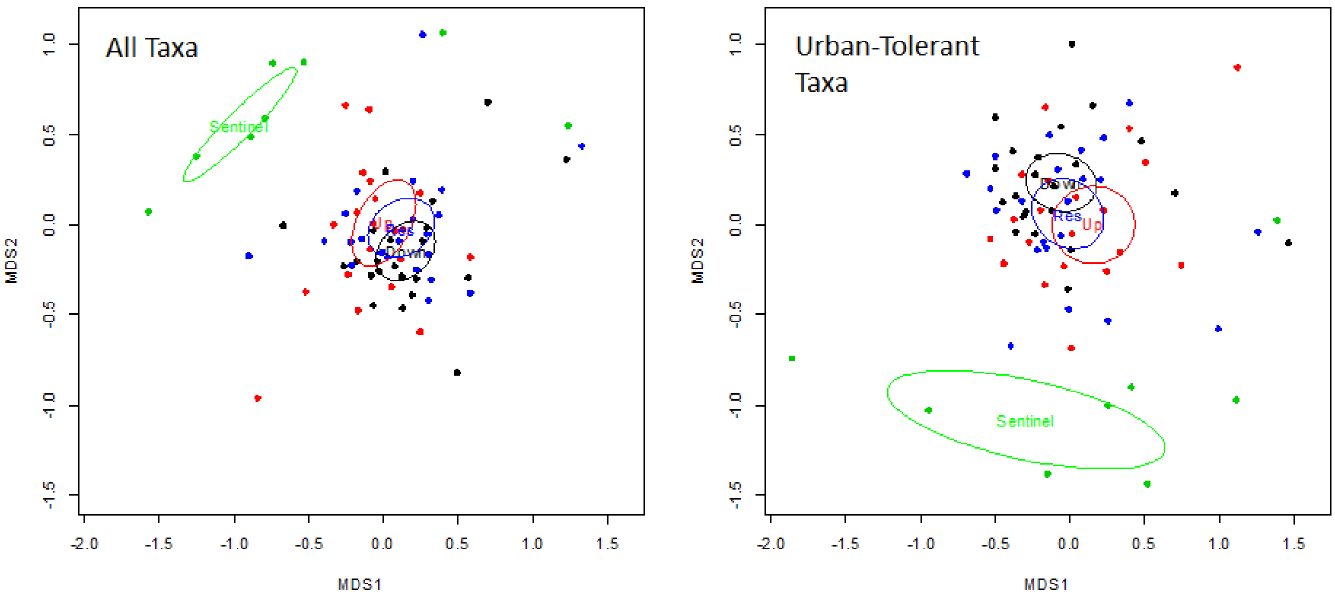


Figure 3. NMS ordination plot of benthic mcaronivertebrate community structure in Restored (blue), Upstream (red), and Downstream (black) sections compared with MBSS Sentinel Sites (green). Ellipses represent 95% CI around the centroid for each section.

Piedmont BIBI scores

BIBI scores decreased at a significantly greater rate in restored sections compared to upstream sections as ISC increased (significant Section X ISC interaction, ANCOVA, P=0.04; Table 5). Downstream sections were not different from either upstream or restored. Because of the nature of the ISC X Section interaction, the significant main effect for section cannot be interpreted cleanly; BIBI scores are higher in restored sections when ISC is low, but lower in restored sections when ISC is high (Figure 4). No other covariates such as Restoration Age or Restoration Length had significant interactions with Section. Similarly, we found no significant differences among upstream, restored, and downstream sections when analyzed without covariates (P>0.05).   While Figure 4 is graphically demonstrative of the BIBI-ISC relationship, it is somewhat misleading because it is not possible to display the within-stream specific effects (triplet design) and how that fits within the ANCOVA calculations. A complete set of ecological measures plotted against ISC for both Piedmont and Coastal Plain streams can be found in Appendix B.

Table 5. Results of ANCOVA analyses testing for differences among Upstream, Restored, and Downstream sections while accounting for ISC, Restoration Age, and Restoration Age as covariates and their interactions. Results in **BOLD** are statistically significant.

	BIBI	Total Richness	EPT Richness	Intolerant Richness	Urban-Tolerant Richness	Shannon-Wiener Richness	Margalef Richness
Section	**Rest>Up at low ISC; Up > Rest at high ISC; Down not different**	NS	NS	NS	NS	NS	NS
ISC	**Negative**	NS	**Negative**	NS	NS	**Negative**	**Negative**
Restoration Age	NS	NS	NS	NS	NS	NS	NS
Restoration Length	NS	NS	NS	NS	NS	NS	NS
Section * ISC	**P=0.04**	NS	NS	NS	NS	NS	NS
Section * Restoration Age	NS	NS	NS	NS	NS	NS	NS
Section * Restoration Length	NS	NS	NS	NS	NS	NS	NS

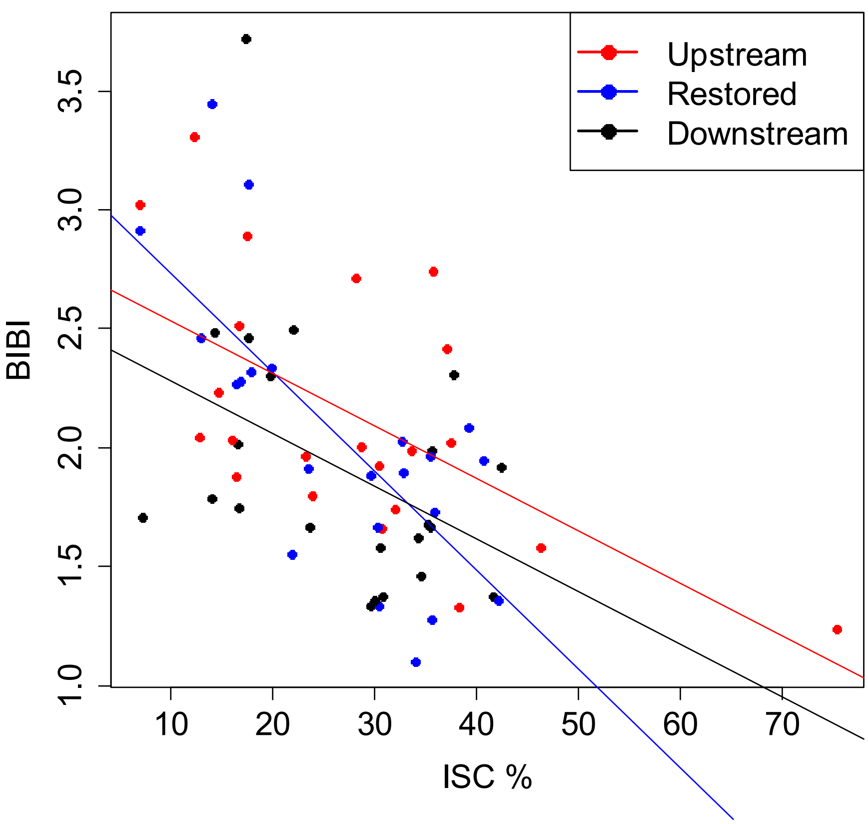


Figure 4. Relationship of BIBI scores in Restored (blue), Downstream (black), and Upstream (red) sections of Piedmont streams to %ISC in the watershed.

A more detailed examination of BIBI scores within and among streams shows that fewer than half of the sites showed improvements in restored versus upstream sections, and only two streams had higher BIBI scores in both the restored and downstream sections than in the upstream unrestored (Table 6). When plotted along the impervious surface gradient, there is a clear pattern of decreasing BIBI with increasing watershed ISC across all sections (Figure 5) as well as graphical evidence showing about half of the streams had higher BIBI scores in the upstream unrestored section compared to both restored and downstream sections. A complete set of section-specific comparisons of ecological measures for Piedmont and Coastal Plain streams can be found in Appendix E.

Table 6. BIBI scores for each section of all streams sampled and realized Ecological Uplift in the Piedmont. Realized Ecological Uplift was calculated as the difference between Restored BIBI – Upstream BIBI and are coded as red (no uplift) or blue (positive uplift). Positive Streams with restored section BIBI scores greater than their upstream unrestored counterpart are marked with a ✓, while those having lower scores are marked with X. The same applies for comparing downstream sections to the upstream unrestored. Streams in **BOLD** showed uplift for both restored and downstream sections. Sections in Red are rated as Very Poor, whereas those in Orange are Poor and Yellow are Fair. No sections ranked as Good.

Project	Upstream	Restoration	Downstream	Realized uplift	Restoration Uplift?	Downstream Uplift?
Batchellors East	2.04	2.46	1.79	0.42	✓	X
Bryants Nursery	3.31	3.45	2.49	0.14	✓	X
Turkey Branch	1.58	1.36	1.38	-0.22	X	X
Upper N.B.	2.03	2.27	2.02	0.24	✓	X
Batchellors Run	1.88	2.28	1.75	0.4	✓	X
NW Branch	1.8	1.91	1.67	0.11	✓	X
Sherwood	2.71	1.89	1.58	-0.82	X	X
Hawlings River	3.02	2.92	1.71	-0.1	X	X
Booze Creek	1.24	1.28	1.67	0.04	X	✓
Stream Valley	**2.89**	**3.11**	**3.72**	**0.22**	**✓**	**✓**
Hollywood Br.	2	2.03	1.99	0.03	✓	X
NB 1	1.66	1.34	1.33	-0.32	X	X
NB 3	1.93	1.67	1.38	-0.26	X	X
RC 2	1.33	1.1	1.62	-0.23	X	✓
PB 109	2.23	2.34	2.3	0.11	✓	X
NW 4	1.99	1.9	1.36	-0.09	X	X
Elmmede	1.74	1.96	1.68	0.22	✓	X
Little Patuxent	2.74	1.95	1.92	-0.79	X	X
Meadowbrook	**2.02**	**2.08**	**2.31**	**0.06**	**✓**	**✓**
U.L. Patuxent	1.97	1.55	2.5	-0.42	X	✓
Windflower	2.42	1.73	1.46	-0.69	X	X
Woodlawn	2.51	2.32	2.46	-0.19	X	X

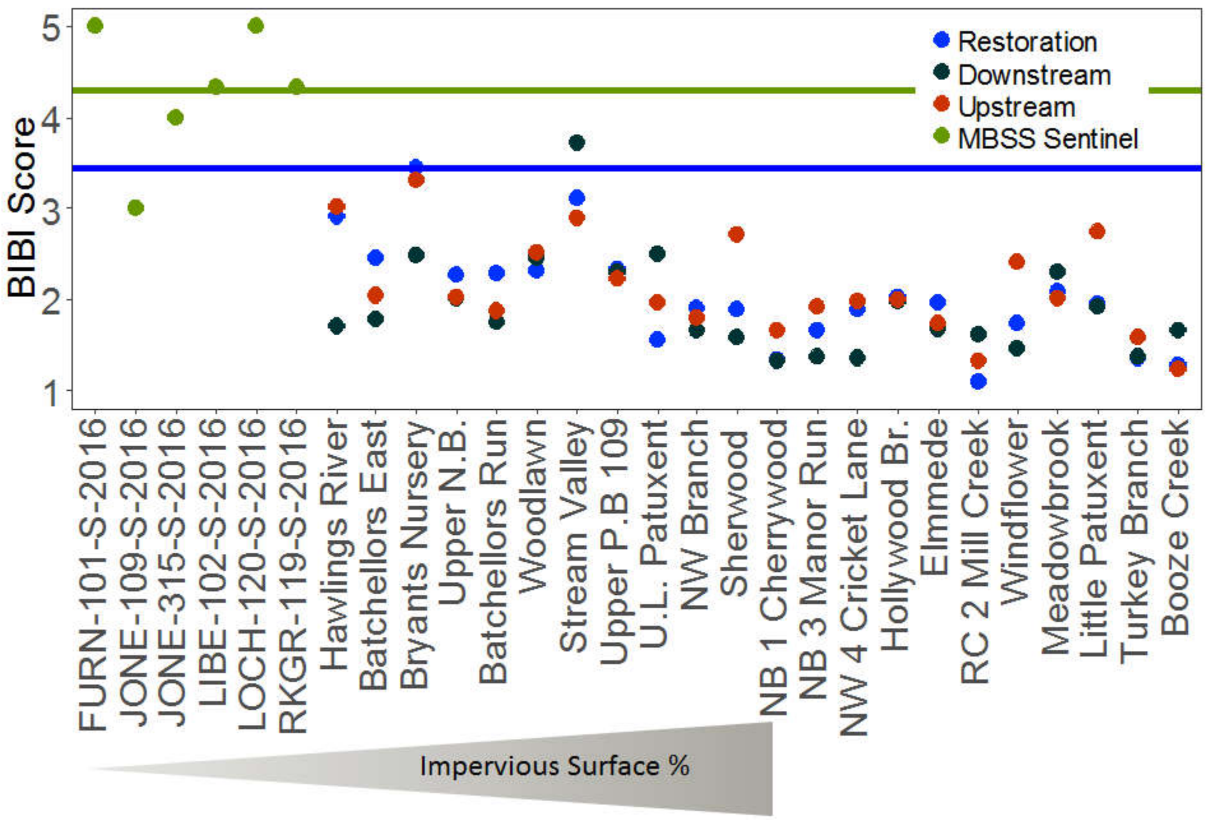


Figure 5. Plot of BIBI scores for Restored (blue), Downstream (black), and Upstream (red) sections for each Piedmont stream and ordered across the ISC gradient. MBSS Sentinel Site scores (green) are provided for context and to define the maximal expected ecological uplift (green horizontal line). The blue horizontal line denotes the potential uplift for restored streams.

Our more focused analyses restricted to the restored sections identified two alternative models that best explained BIBI scores (Table 7). Both models identified ISC (negative relationship to BIBI), the number of step pools (positive), and the total number of installed structures (negative, Figure 6) as highly important variables, while pH (negative, Model 1) and percent gravel (positive, Model 2) were also influential. The best model explained nearly 80% of the variation in BIBI. We were surprised that neither Restoration Length nor Restoration Age (Figure 7) was important and equally surprised that the total number of structures seemed to reduce ecological uplift.

Table 7. Top performing models to predict BIBI scores within restored sections of Piedmont streams.

BIBI	ISC	Number of Step Pools	Total Number of Structures	pH	Gravel	Prob	Adjusted R^2	ΔAIC
Model 1	β= -0.045 t= -6.68 P=0.00001	β= 0.10 t= 5.04 P=0.0002	β= -0.04 t= -5.22 P=0.0001	β= -0.45 t= -2.41 P=0.03		0.00002	79.3	0
Model 2	β= -0.04 t= -5.42 P=0.00009	β= 0.89 t= -3.92 P=0.118	β= -0.031 t= -3.58 P=0.003		β= 0.01 t= 2.00 P=0.065	0.00004	77.1	1.82
Overall Importance	**0.99**	**0.94**	**0.93**	**0.58**	**0.27**			

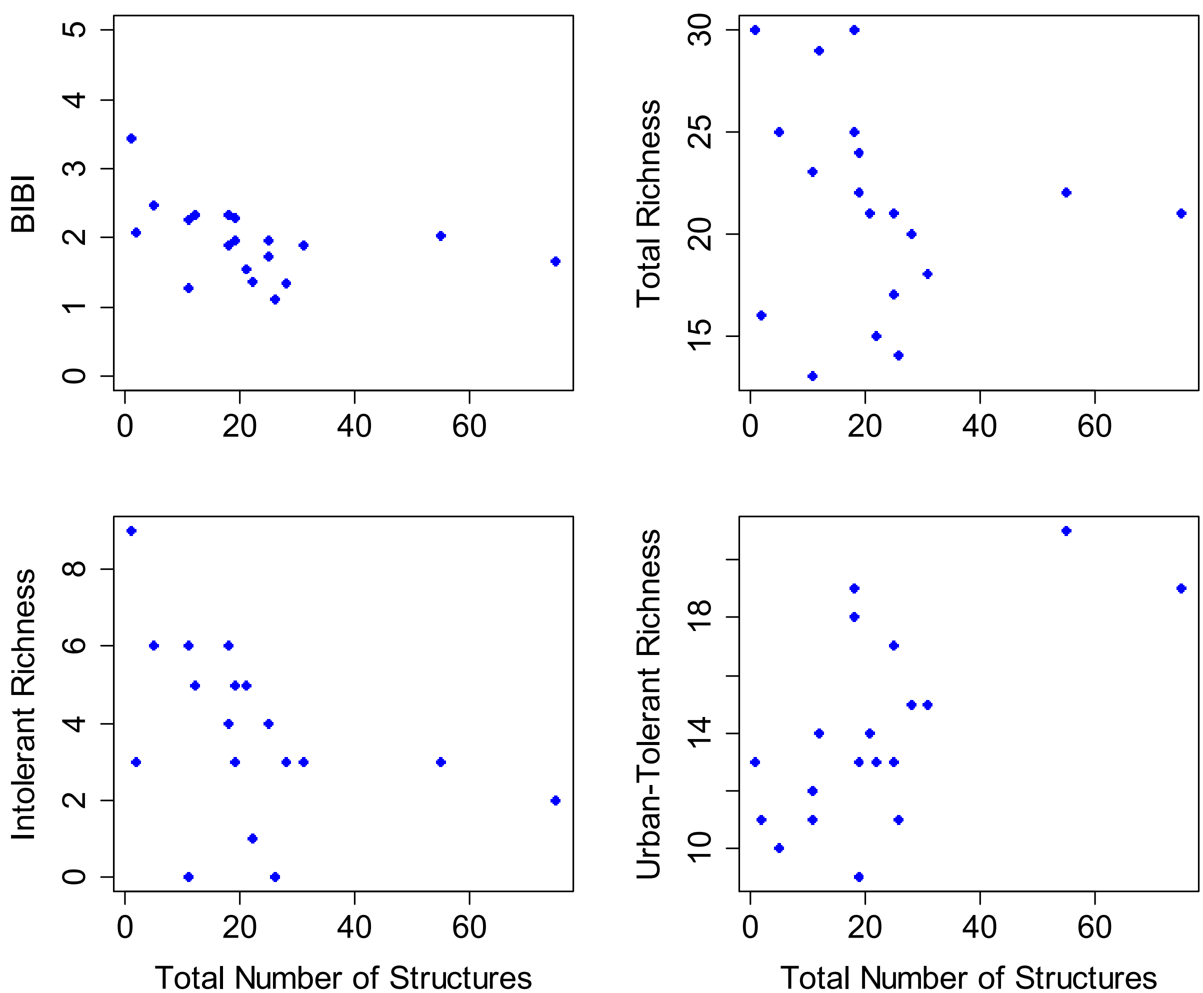


Figure 6. Plots of BIBI, Total Richness, Richness of Intolerant Taxa, and Richness of Urban Tolerant Taxa against the total number of structures in the restored section upstream of the sample.

Complete sets of ecological measures plotted against Restoration Age, Restoration Length, and the different types of installed structures for both Piedmont and Coastal Plain streams can be found in Appendices C, D, and F, respectively.

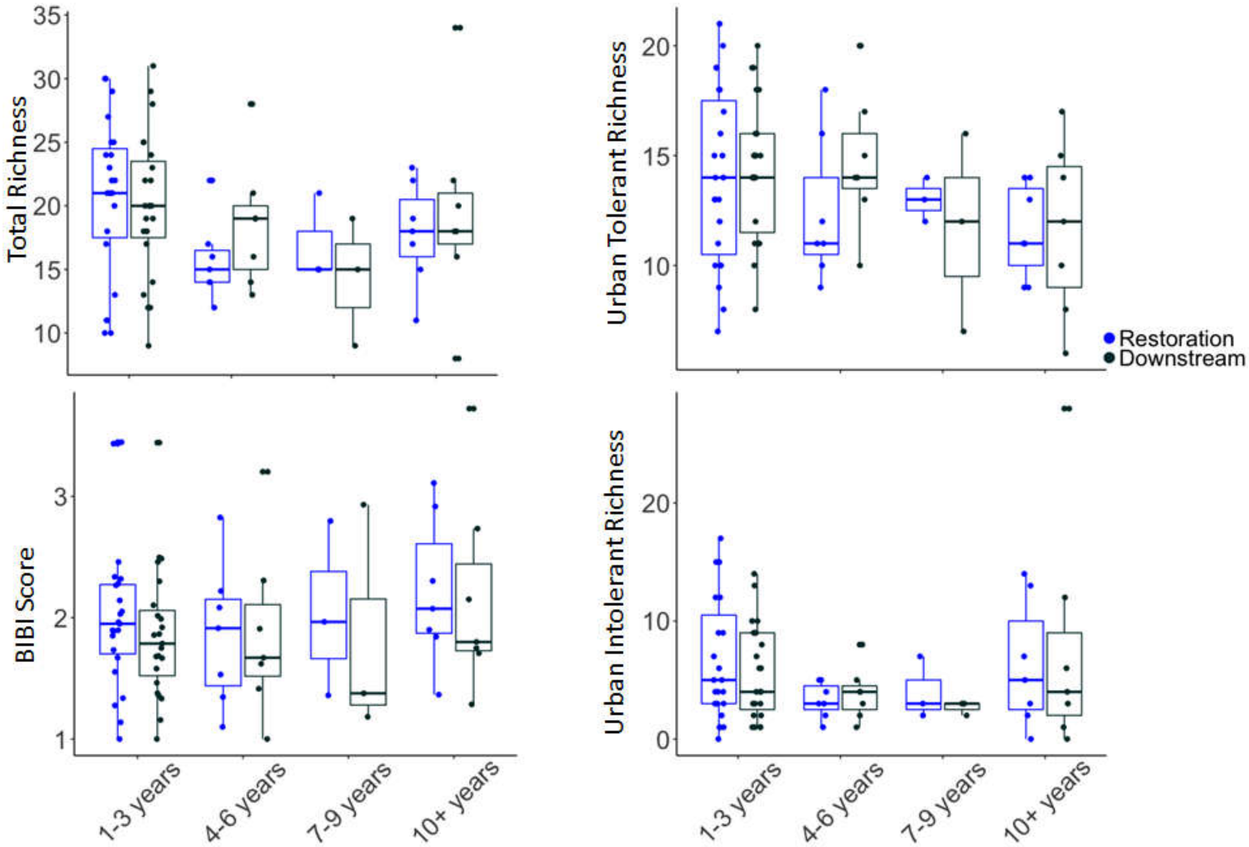


Figure 7. Boxplots of ecological uplift measures in Restored (blue) and Downstream (black) sections and grouped by Restoration Age.

We conclude that there is little overall positive effect of stream restoration activities on increasing BIBI scores compared to unrestored sections in Piedmont streams. Watershed-level drivers such as ISC and pH appear to be the main influences on this aspect of ecological uplift. Within restored sections, increasing the total numbers of step pools and the amount of gravel has a positive influence on the BIBI, but increasing the total number of structures and longer Restoration Length seem to depress BIBI. We speculate that the disturbance involved in installing structures and the overall invasiveness of larger restorations might be more detrimental than beneficial, and that the system does not strongly recover with time since restoration activities have ceased.

Piedmont Taxonomic Richness Measures

Overall Taxonomic Richness in the Piedmont was not different among the triplet sections, nor were there any significant interactions with the covariates: ISC, Restoration Length, or Restoration Age (ANCOVA, $P>0.05$; Table 5). As with the BIBI, there was a trend for overall taxonomic richness to decline more rapidly in restored sections as ISC increased (Figure 8), but it was not statistically significant. The remaining richness metrics had similar relationships, except for Urban-Tolerant Richness, which was positively related to ISC (Appendix Figure B1).

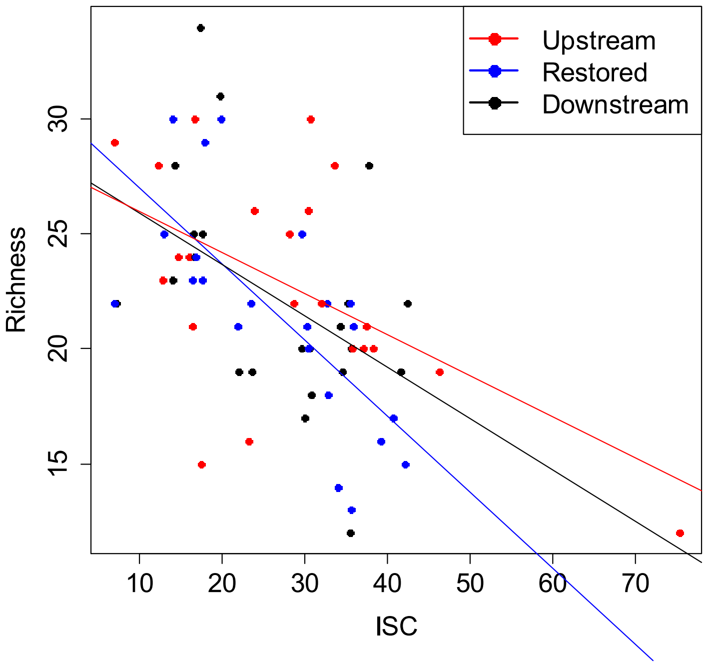


Figure 8. Relationship of Taxonomic Richness in Restored (blue), Downstream (black), and Upstream (red) sections of Piedmont streams to %ISC in the watershed.

A more detailed look at Total Richness displaying within-stream relationships shows that some restored sites outperformed MBSS Sentinel Sites (Figure 9). While Richness in most streams was higher in upstream sections compared to restored, several streams had higher overall Richness than Sentinel Sites. Thus, the Potential Ecological Uplift is high for Richness where the best restorations may equal or exceed values found in lesser disturbed watersheds. We found similar a pattern for Urban-Tolerant Richness, but not for measures based on sensitive taxa (Appendix B).

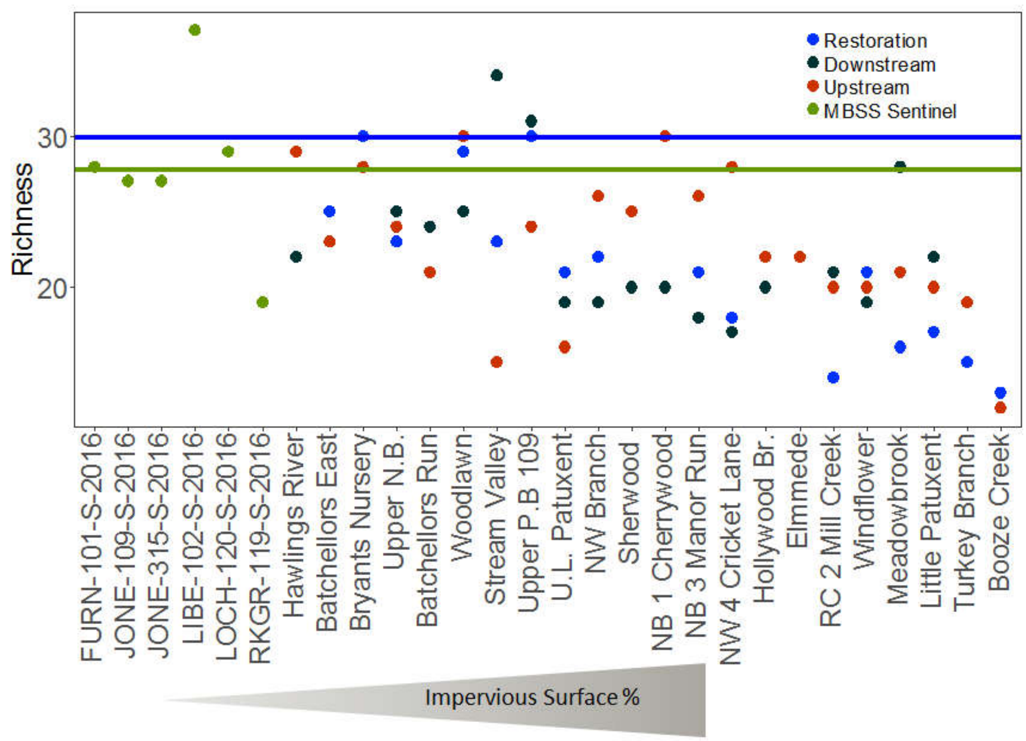


Figure 9. Plot of Total Richness for Restored (blue), Downstream (black), and Upstream (red) sections for each Piedmont stream and ordered across the ISC gradient. MBSS Sentinel Site scores (green) are provided for context and to define the maximal expected ecological uplift (green horizontal line). The blue horizontal line denotes the potential uplift for restored streams.

Focusing only on the restored sections, we found taxonomic richness to be best predicted by two competing models. The best performing model (P<0.00001) identified a positive relationship between Richness and the total number of Rock Vane structures and negative relationships with the Restoration Length and increasing Specific Conductance, Percent Fine sediments, and higher Riffle Frequency/Channel Sinuosity scores (Table 8). This best performing model explained over 88% of the variation. The competing model (P=0.00004) explained nearly 85% of the variation and identified all variables in the best model except for the total number of Rock Vane structures. Every variable in the competing model exerted a negative effect on overall taxonomic Richness.

Table 8. Top performing models to predict Total Richness within restored sections of Piedmont streams.

Richness	Specific Conductance	Riffle Freqquency/ Channel Sinuosity	Restoration Length	Percent Fines	Number of Rock Vanes	Prob	Adjusted R^2	ΔAIC
Model 1	β= -0.01 t= -6.78 P=0.00001	β= -0.495 t= -6.56 P=0.00002	β= -0.0023 t= -4.62 P=0.0005	β= -0.361 t= - 2.35 P=0.033	β= 0.137 t= 2.25 P=0.042	<0.00001	88.2	0
Model 2	β= -0.01 t= -6.01 P=0.00003	β= 0.443 t= -5.43 P=0.00008	β= -0.001 t= -3.56 P=0.003	β= -0.408 t= - 2.36 P=0.033		0.00004	84.9	1.06
Overall Importance	**1**	**1**	**0.99**	**0.7**	**0.64**			

We also examined Intolerant Richness and Urban-Tolerant Richness separately to assess if one component (e.g., Urban-Tolerant richness) was responding to restorations while the other might not. Our logic here was to test the assertions from discussion with restoration practitioners that measures such as the BIBI are not realistic measures because water quality might limit the sensitive taxa regardless of the quality of the restoration, while Urban-Tolerant taxa might actually respond positively.

Unfortunately, neither component responded positively to restoration activities compared to the upstream unrestored sections (Table 5). We found no differences in the richness of either Urban-Tolerant or Intolerant taxa among the different sections after accounting for stream-specific effects in the Piedmont (ANOVA, P>0.05), nor did we detect any significant interactions with the covariates, ISC, Restoration Length, or Restoration Age (P>0.05). Thus, restored sections had similar numbers of Urban-Tolerant and Intolerant genera compared to the upstream unrestored and downstream sections across the gradients of the covariates.

Constraining analyses to focus only on restored sections, we found that the Richness of Intolerant taxa was best predicted by two competing models (Table 9). In the top performing model, Intolerant Richness increased with increasing the % Gravel in restored sites and decreased with increasing ISC, increasing % Sand/Silt, higher Total Number of Structures, and Restoration Length (nonsignificant variable). The competing model identified ISC, %Sand/Silt, %Cobble, and Restoration Length to be associated with Intolerant Richness and a nonsignificant trend for increasing richness in older restorations.

Table 9. Top performing models to predict Richness of Intolerant taxa within restored sections of Piedmont streams.

Intolerant Richness	ISC	Total Number of Structures	% Gravel	% Sand/Silt	Length	Restoration Age	% Cobble	P	Adjusted R^2	ΔAIC
Model 1	β= -0.368 t= -7.98 P<0.00001	β= -0.095 t= -3.51 P=0.004	β= 0.081 t= 2.69 P=0.02	β= -0.146 t= -2.76 P=0.016	β= -0.00035 t= -0.778 P=0.45			<0.00001	91	0
Model 2	β= -0.306 t= -4.23 P=0.0006			β= -0.223 t= -2.33 P=0.034	β= -0.0004 t= -1.85 P=0.0.083	β= 0.187 t= 0.828 P=0.42	β= -0.096 t= -1.91 P=0.07	0.00002	76.5	1.4
Overall Importance	**1**	**0.71**	**0.52**	**0.43**	**0.31**	**0.3**	**0.27**			

As mentioned previously, we found no benefits of restoration activities in the taxonomic richness of Urban-Tolerant taxa in restored sections compared to upstream and downstream sections (Table 5). This result falsifies our hypothesis that restorations might increase the Urban-Tolerant Richness even if sensitive taxa did not respond.

Within restored sections, three competing models best described the richness of Urban-Tolerant taxa. The best performing model as ranked by ΔAIC explained 52% of the variation and identified ISC, Specific Conductance, and Restoration Age as important. In contrast to BIBI and Intolerant Richness, increasing ISC promoted Urban-Tolerant Richness, while Specific Conductance and Restoration Age were negatively related (Table 10). The two other competing models identified the total number of Rock Vane structures (Models 2 and 3) as promoting Urban-Tolerant Richness and increasing amounts of sediment deposition (Model 3) as reducing richness.

Table 10. Top performing models to predict Richness of Urban-Tolerant taxa within restored sections of Piedmont streams.

Urban-Tolerant	ISC	Specific Conductance	Restoration Age	Number of Rock Vane Structures	Sediment Deposition	Prob	Adjusted R^2	ΔAIC
Model 1	β= 0.336 t= 3.87 P<0.001	β= -0.01 t= -3.57 P=0.003	β= -0.82 t= -2.66 P=0.017			0.0027	52	0
Model 2	β= 0.06 t= 3.63 P=0.0027	β= -0.009 t= -3.61 P=0.0028	β= -0.89 t= -3.064 P=0.008	β= 0.119 t= 1.84 P=0.09		0.0021	58.6	0.27
Model 3	β= 0.29 t= 3.74 P=0.0024	β= -0.012 t= -4.31 P=0.0008	β= -1.09 t= -3.78 P=0.002	β= 0.14 t= 2.29 P=0.04	β= -0.29 t= -1.85 P=0.09	0.002	64.7	1.03
Overall Importance	**0.85**	**0.85**	**0.76**	**0.33**	**0.19**			

Piedmont EPT Richness

Overall EPT richness was not different among sections within each triplet, nor were there significant interactions with ISC, Restoration Age, or Restoration Length (ANCOVA, P>0.05; Table 5). Across all sites and sections, EPT Richness decreased significantly with increasing ISC (P<0.0001).

Focusing specifically on restored reaches, two competing models best explained EPT Richness (Table 11). Across both models, increases in ISC, Restoration Length, and Riffle Frequency/Channel Sinuosity were all negatively related to EPT Richness and suggests that restoration activities may be detrimental. Additionally, the Total Number of J-Hook Structures was negatively related to EPT Richness, but was non-significant despite being included in Model 2.

Table 11. Top performing models to predict Richness of EPT taxa within restored sections of Piedmont streams.

EPT Richness	ISC	Number of J-Hook Structures	Restoration Length	Riffle Freqquency/ Channel Sinuosity	Prob	Adjusted R^2	ΔAIC
Model 1	β= -0.10 t= -3.21 P<0.006		β= -0.0001 t= -2.79 P=0.01	β= -0.16 t= -3.25 P=0.005	<0.0001	75.6	0
Model 2	β= -0.12 t= -3.77 P<0.002	β= -0.33 t= -1.75 P=0.10	β= -0.0008 t= -2.93 P=0.01	β= -0.16 t= -3.44 P=0.004	0.0021	78.5	0.64
Overall Importance	**0.97**	**0.32**	**0.7**	**0.8**			

Piedmont Taxonomic Diversity Measures

Shannon-Wiener and Margalef richness indices complement the purely taxonomic richness measures by also considering the relative abundances of taxa within the sampled community. Neither Shannon-Wiener nor Margalef Richness responded differently in restored sections when compared to unrestored sections, upstream or down, after accounting for stream specific effects (ANCOVA, P > 0.05). We also found no significant interactions with ISC, Restoration Age, or Restoration Length (ANCOVA, P>0.05). However, increasing ISC significantly decreased both Shannon-Wiener (P=0.001) and Margalef Richness (P<0.0001). Within restored sections, Shannon-Weiner diversity was positively related to the Total Number of Step Pools installed, but negatively related to the Total Number of Structures as well as increasing Specific Conductance, % Fine Sediments, and Sediment Deposition (Table 12)

Table 12. Top performing models to predict Shannon-Wiener Richness within restored sections of Piedmont streams.

Shannon-Wiener Diversity	Specific Conductance	% Fine Sediment	Total Number of Step Pools	Total Number of Structures	Sediment Deposition	Prob	Adjusted R^2	ΔAIC
Model 1	β= -0.002 t= -7.64 P<0.0001	β= -0.07 t= -3.67 P=0.003	β= 0.11 t= 5.29 P=0.0001	β= -0.04 t= -5.12 P=0.0002	β= -0.04 t= -2.38 P<0.03	<0.0001	78.3	0
Model 2	β= -0.001 t= -6.51 P<0.0001	β= -0.07 t= -3.02 P=0.009	β= 0.11 t= 4.48 P=0.0005	β= -0.04 t= **-4.41** P=0.0006		0.0002	71	1.71
Overall Importance	**0.97**	**0.78**	**0.78**	**0.77**	**0.32**			

Within restored sections, Margalef Richness was negatively related to %Fine Sediment, Restoration Age, Restoration Length, Riffle Frequency/Channel Sinuosity, and Specific Conductance (Table 13).

Table 13. Top performing models to predict Margalef Richness within restored sections of Piedmont streams.

Margalef Richness	Specific Conductance	Riffle Freqquency/ Channel Sinuosity	% Fine Sediment	Restoration Age	Restoration Length	Prob	Adjusted R^2	ΔAIC
Model 1	β= -0.001 t= -5.64 P<0.0001	β= -0.06 t= -4.28 P=0.0001	β= -0.08 t= -2.57 P<0.02	β= -0.11 t= -2.23 P<0.04		<0.0001	80.3	0
Model 2	β= -0.001 t= -5.72 P<0.0001	β= -0.10 t= -3.21 P=0.06	β= -0.06 t= -2.00 P=0.06		β= -0.0002 t= -1.97 P=0.07	<0.0001	79	1.15
Overall Importance	**0.92**	**0.86**	**0.35**	**0.24**	**0.27**			

To summarize, we found limited evidence that restorations in Piedmont urban streams promoted any aspects of taxon richness as compared to upstream unrestored sections or sections downstream of restorations. Thus, we did not detect any signatures of ecological uplift in taxon richness due to restoration activities. Within restorations, ISC was a dominant controlling factor that decreased richness in sensitive taxa, while increasing richness in urban-tolerant taxa. The effect of ISC was not mediated by restoration activities. Restoration Age was nonsignificant, but tended to improve richness of sensitive taxa. Restoration specific activities of increasing %Gravel and the number of Rock Vanes improved richness, while increasing the Total Number of Structures, %fines, and the Riffle Frequency reduced richness. In addition, longer restorations were negatively related to richness and might indicate that the larger or more intensive the restoration, the more limited the ecological recovery potential.

Piedmont Physical Habitat Measures

Although PHI scores were on-average higher within restored sections, they were not significantly different from either upstream or downstream sections (P>0.05). Many restoration tactics focus on improving measures that are included in physical habitat indices such as the PHI, but we found no habitat response to restorations at this level of aggregation (Figure 10).

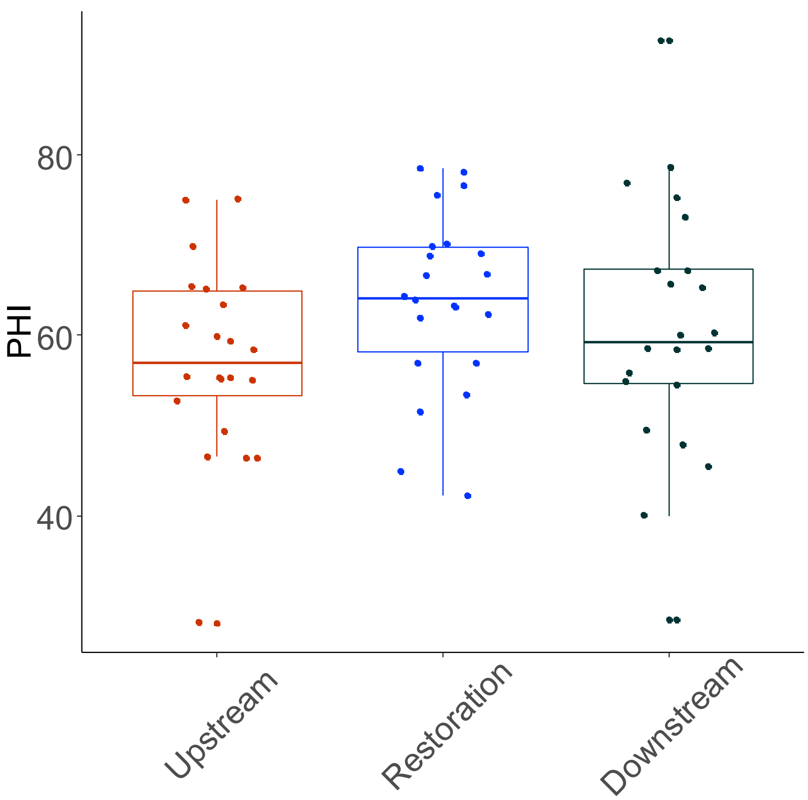


Figure 10. Physical Habitat Index scores for Upstream, Restored, and Downstream sections in Piedmont streams. Note that the figure does not incorporate the stream-specific effects that were modeled in the statistical analysis.

While the overall PHI in Piedmont sites showed no response, several individual habitat measures were significantly different among sections. Restored sections had significantly lower Channel Alteration scores (low is worse), higher Total Bank Stability, higher Sediment Deposition scores (high is good), lower % Sand and %Sand and Silt, and higher D50 substrate values (Table 14). Lower Channel Alteration scores are indicative of restoration activities reversing noticeable changes to the channel in the form straightening, dredging, or other similar modifications. The other significant habitat measures indicate that Piedmont restorations have larger particle sizes and less sedimentation than either upstream or downstream sections. Shifts towards coarser sediments generally improve conditions for benthic macroinvertebrates and other biota. Thus, we view these results as a restoration success.

To summarize, while the overall PHI did not differ among sections, several individual habitat metrics were significantly improved in restored sections compared to upstream unrestored. We view habitat responses in Piedmont stream restorations as a success.

Table 14. Analysis results comparing specific habitat measures among Upstream, Restored, and Downstream sections in Piedmont streams. Comparisons with a "+" indicate that the first section mentioned is significantly greater than the second, while a "-" indicates significantly lower, and NS = Not Statistically Significant.

Habitat Attribute	Restored vs Upstream	Restored vs Downstream	Downstream vs Upstream
Epifaunal Substrate	NS	NS	NS
Embeddedness	NS	NS	NS
Velocity Depth Pool Variability	NS	NS	NS
Channel Alteration	-	-	NS
Sediment Deposition	+	+	NS
Riffle Frequency Channel Sinuosity	+	NS	+
Total Bank Stability	+	+	+
Total Bank Vegetative Protection	+	+	NS
Total Width Undisturbed Vegetative Zone	NS	NS	NS
Pool Substrate Characterization	NS	NS	NS
Pool Variability	NS	NS	NS
Sinuosity	NS	NS	NS
% Fines	NS	NS	NS
% Sand	-	-	NS
% Sand and Silt	-	-	NS
% Gravel	NS	NS	NS
% Cobble	NS	NS	NS
% Boulder	NS	NS	NS
D50	+	+	NS
D84	NS	NS	NS

Coastal Plain Results

Coastal Plain Benthic Macroinvertebrate Community Structure

Similar to the Piedmont, Coastal Plain benthic macroinvertebrate community structure in restored streams was significantly different between Sentinel Sites and the study streams for both RSC and NCD restorations. However, no differences existed among upstream, restored, and downstream sections (Table 15). NMS ordinations for Coastal Plain sites show strong separation of Sentinel Sites from the other reaches for both RSC and NCD restorations, but little separation between NCD and RSC restorations or among sections (Figure 11). Surprisingly, the same pattern holds when examining only urban-tolerant taxa. Thus, urban streams that have been restored do not resemble the minimally disturbed Sentinel Sites. Restored reaches do not differ from unrestored reaches, and show no signs of ecological uplift downstream with respect to benthic macroinvertebrate community structure.

Table 15. ADONIS analysis F-value results comparing the community similarity of Restored, Upstream, Downstream, and MBSS Sentinel Sites. The MBSS Sentinel Site comparisons were generated using the bootstrapping approach described in the Methods. **Bolded values** with an asterisk indicate statistically significant differences between the pairwise comparisons.

	Restored vs Downstream	Upstream vs Downstream	Restored vs Upstream	Sentinel vs Upstream	Sentinel vs Downstream	Sentinel vs Restored
Coastal Plain NCD	0.37	0.42	0.89	**4.56***	**6.75***	**5.92***
Coastal Plain RSC	0.56	0.47	0.81	**4.62***	**5.93***	**5.11***

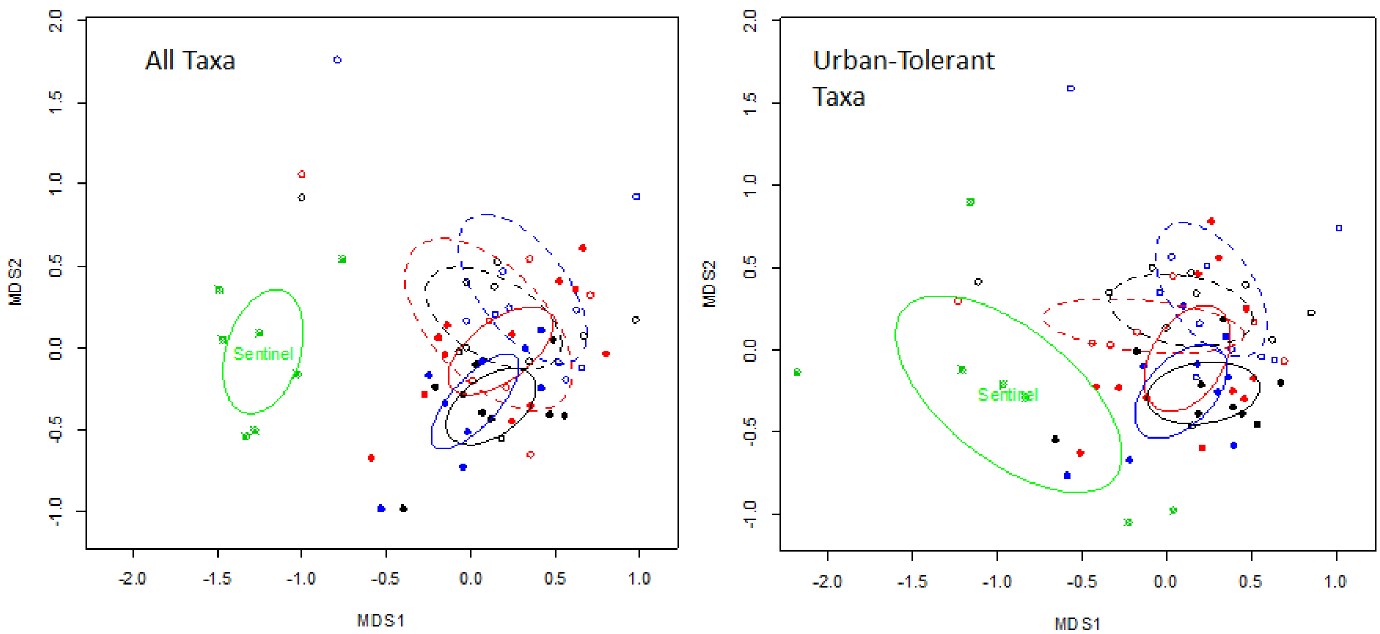


Figure 11. NMS ordination plot of benthic mcaronivertebrate community structure in Restored (blue), Upstream (red), and Downstream (black) sections compared with MBSS Sentinel Sites (green). RSC-dominant restorations are shown as open circles and dashed lines, whereas NCD-dominant restorations are shown as solid dots and lines. Ellipses represent 95% CI around the centroid for each group.

Coastal Plain BIBI Scores

Our main hypothesis of stream restorations improving BIBI scores in the Coastal Plain was rejected. We found no BIBI score differences among the upstream, restored, or downstream sections or any significant interactions with Restoration Type, ISC, Restoration Length, or Restoration Age after accounting for the individual effects of each stream (Model averaged ANCOVA, P>0.05; Table 16). BIBI scores for NCD-dominant restorations were significantly higher than RSC-dominant restorations overall (Model Averaged ANOVA, P=0.02). However, this difference was not due to restoration-specific activities, but due to the selection of streams to which the restorations were applied; unrestored upstream sections followed the same patterns with higher BIBI scores in NCD-dominant restorations than in RSC projects, although not statistically significant (ANOVA, P=0.06). Nonetheless, the same trends strongly indicate the differences are due to the selection of streams and not due to the Restoration Type.

Table 16. Results of ANCOVA analyses testing for differences among Upstream, Restored, and Downstream sections and Restoration types while accounting for ISC, Restoration Age, and Restoration Age as covariates. Results in **BOLD** are statistically significant. Note that differences marked with an asterisk (*) indicate a stream-wide effect and not a true difference due to restoration type. Please see results text for a full explanation.

	BIBI	Total Richness	EPT Richness	Intolerant Richness	Urban-Tolerant Richness	Shannon-Wiener Richness	Margalef Richness
Section	NS	NS	NS	NS	NS	NS	NS
Restoration Type	**NCD > RSC***	NS	**NCD > RSC***	NS	NS	**NCD > RSC***	NS
ISC	NS	NS	NS	NS	NS	NS	NS
Restoration Age	NS	NS	NS	NS	NS	NS	NS
Restoration Length	NS	NS	NS	NS	NS	NS	NS
Section * ISC	NS	NS	NS	NS	NS	NS	NS
Section * Restoration Age	NS	NS	NS	NS	NS	NS	NS
Section * Restoration Length	NS	NS	NS	NS	NS	NS	NS
Section * Type	NS	NS	**0.01**	NS	NS	NS	NS
Type * ISC	NS	NS	NS	NS	NS	NS	NS
Type * Restoration Age	NS	NS	NS	NS	NS	NS	NS
Type * Restoration Length	NS	NS	NS	NS	NS	NS	NS

The major influence on BIBI scores across all triplets was a positive relationship with pH (model averaged ANCOVA, P=0.005), which is the opposite of the relationship in the Piedmont. No other covariates (e.g., Restoration Age, ISC, Restoration Type or Total Number of Structures used, etc.) were related to BIBI scores.

Ten of the 18 Coastal Plain restorations had BIBI scores higher in restored sections than their upstream control, and 8 of 18 were higher in downstream compared to upstream sections (Table 17). Two NCD restorations (Biddison Run and Red House) had ecological uplift in both the restored and downstream sections. Similarly, three RSC restorations (Cabin Branch, Howard Branch, and N.B. Cypress Branch) exhibited uplift in both restored and downstream. The results must be tempered somewhat for two reasons. Three of the five restorations had no upstream control, and the average BIBI of the pooled sites was very low. In addition, the actual improvements in BIBI scores were small for Biddison Run and Cabin Branch. Nonetheless, the improvements in the Red House restoration are impressive, and the downstream section scored in the "Fair" category.

The site with the highest recorded BIBI score in our study (Paint Branch) actually had a higher score in the upstream unrestored section than either the restoration or downstream sections. Unfortunately, the reduction in BIBI scores in the restored and downstream sections were substantial and beyond what one might expect due to chance. BIBI scores in Paint Branch suggest that the restoration had a negative effect.

Table 17. BIBI scores for each section of all streams sampled and realized Ecological Uplift in the Coastal Plain. Realized Ecological Uplift was calculated as the difference between Restored BIBI – Upstream BIBI and are coded as red (no uplift) or blue (positive uplift). Positive Streams with restored section BIBI scores greater than their upstream unrestored counterpart are marked with a ✓, while those having lower scores are marked with X. The same applies for comparing downstream sections to the upstream unrestored. Streams in **BOLD** showed uplift for both restored and downstream sections. Streams with an * for upstream unrestored section have no BIBI score due to the restoration going to the headwaters and instead have the averaged BIBI score for all of the pooled sites. Sections in Red are rated as Very Poor, whereas those in Orange are Poor, Yellow are Fair, and Green is Good.

Approach	Project	Upstream	Restoration	Downstream	Realized Uplift	Restoration Uplift?	Downstream Uplift?
NCD	Bear Branch	2.01	2.22	1.91	0.21	✓	X
	Biddson Run	**2.74**	**2.8**	**2.93**	0.06	✓	✓
	Foster Branch	2.27	2.05	1.86	-0.22	X	X
	L.Paint Branch	2.05	2.3	1.75	0.25	✓	X
	Muddy Bridge	1.88	1.37	1.29	-0.51	X	X
	Paint Branch	4.04	3.43	3.44	-0.61	X	X
	Patuxent Run	1.44	1.35	1	-0.09	X	X
	Red House	1.58*	2.83	3.2	1.25	✓	✓
	White Marsh	2.06	1.84	2.73	-0.22	X	✓
RSC	Cabin Branch	1.58*	1.85	1.87	0.27	✓	✓
	Church Creek	1.58*	1.14	1.69	-0.44	X	✓
	Central San.	1.58*	1.97	1.18	0.39	✓	X
	Croften Trib.	1.58*	1.53	1.41	-0.05	X	X
	Dividing Creek	1	1	1.16	0	X	✓
	Howard Branch	**1.86**	**2.07**	**2.15**	0.21	✓	✓
	N.B. Cypress	1.58*	1.9	2.1	0.32	✓	✓
	Warehouse Creek	1.58*	2.14	1	0.56	✓	X
	Wilelinor	1.82	1.9	1.8	0.08	✓	X

Any Realized Ecological Uplift was generally modest for Coastal Plain restorations. While the scores for Paint Branch show a high Potential Ecological Uplift in NCD restoration approaches, the lack of responses and negative responses suggest that uplift is unlikely (Figure 11). Similarly, Coastal Plain NCD restorations are far from the Maximal Ecological Uplift that could be achieved if watersheds were returned to lesser disturbed conditions.

Coastal Plain RSC restorations fared even worse with respect to their recovery potential. The large difference between Maximal and Potential Ecological Uplift indicate that the most successful RSC restoration was far from what is average in lesser disturbed watersheds (Figure 12).

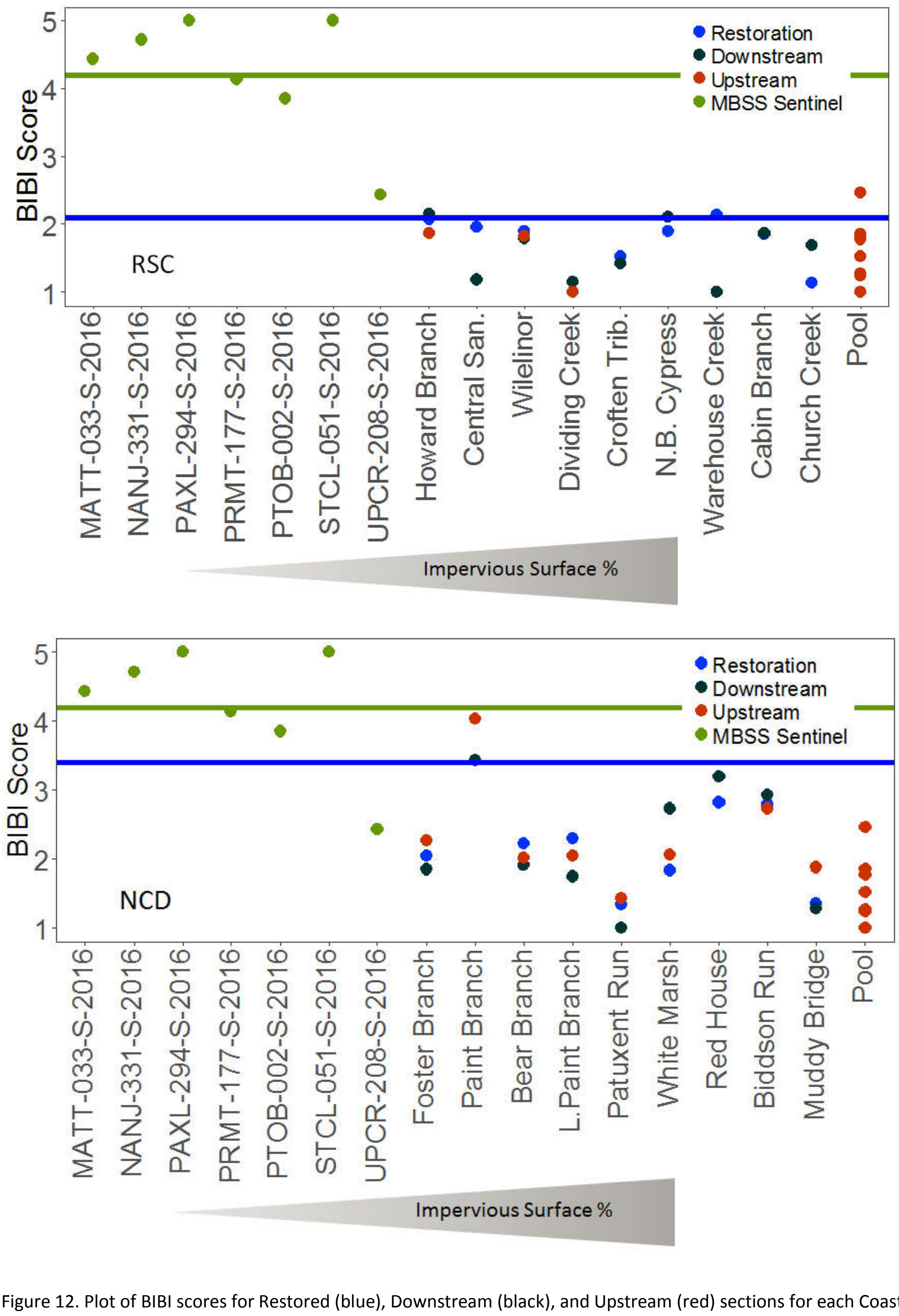


Figure 12. Plot of BIBI scores for Restored (blue), Downstream (black), and Upstream (red) sections for each Coastal Plain stream and ordered across the ISC gradient. The top panel shows RSC-dominant restorations, while the bottom panel shows responses for NCD-dominant restorations. MBSS Sentinel Site scores (green) are provided for context and to define the maximal expected ecological uplift (green horizontal line). The blue horizontal line denotes the potential uplift for restored streams. Sites labeled as “Pool” are those samples serving as matched upstream controls for restorations that extended to the headwaters.

Analyses incorporating installed restoration structures identified negative relationships between BIBI scores and the Number of Step Pools and BIBI and Epifaunal Substrate Available Fish Cover (Table 18). We found no interactions between Restoration Type and installed structures, indicating similar BIBI responses across both NCD and RSC-dominant approaches. We also found positive relationships with BIBI and Total Bank Stability, substrate Embeddedness, and pH. As mentioned previously, analyses examining restoration structures contained a reduced sample size because we could not obtain as-built surveys for every restoration. For the Coastal Plain, we obtained as-built surveys from 7 of 9 NCD-dominant and 8 of 9 RSC-dominant restorations. Thus 3 were missing from this focused analysis.

Table 18. Top performing models to predict BIBI scores within restored sections of Coastal Plain streams.

BIBI	Total Bank Stability	Embeddedness	pH	Toal Number of Step Pools	Epifaunal Substrate Available Fish Cover	Prob	Adjusted R^2	ΔAIC
Model 1	β= 0.11 t= 6.77 P=0.00007	β= 0.08 t= 6042 P=0.0001	β= 0.275 t= 4.02 P=0.003	β= -0.03 t= -5.01 P=0.0007	β= -0.03 t= -2.89 P=0.02	0.00001	93	0
Overall Importance	**0.91**	**0.9**	**0.72**	**0.69**	**0.43**			

In summary, we reject our main hypothesis that restoration activities improved the BIBI scores in restored sections as compared with unrestored sections upstream or with sections downstream of restorations. We similarly found no differences in outcomes between NCD- and RSC-dominant restoration approaches. Rather, differences in BIBI scores between the two restoration approaches were due to inherently higher BIBI scores in NCD-restored streams even in unrestored sections. Restoration-specific activities such as increasing the number of step-pools actually reduced BIBI scores compared to restorations with fewer step pools. However, increasing bank stability and lowering substrate sediment embeddedness was related to higher BIBI scores.

Coastal Plain Taxonomic Richness Measures

Benthic macroinvertebrate total taxonomic Richness was not different among sections or restoration approaches, nor were there any significant interactions with the Restoration Type, ISC, Restoration Length, or Restoration Age (Model averaged ANCOVA, $P>0.05$). We similarly found a lack of any differences for Intolerant Richness or Urban-Tolerant Richness (Model averaged ANCOVA, $P>0.05$). We thus reject our main hypothesis that taxonomic richness and its constituent components will be higher in restored sections, and we reject the notion that restoration activities will improve richness downstream.

Two competing models best described Richness (Table 19). The Total Number of Vane Structures and the Number of J-Hook Rock Structures were both positively related to richness, but no other specific structure classes or aggregated classes were important. We found no interactions between structures and Restoration Type, which indicates that Richness did not differentially change between restoration approaches for any types of structures. Additionally, Richness was positively related to Total Bank Stability, greater Velocity Depth Pool Variability, and reduced Sediment Deposition. Both models explained around 86% of the variation in Richness.

Table 19. Top performing models to predict Total Richness within restored sections of Coastal Plain streams.

Total Richness	Total Bank Stability	Sediment Deposition	Velocity Depth Pool Variability	Total Number of Vane Structures	Number of J-Hook Rock Structures	Prob	Adjusted R^2	ΔAIC
Model 1	β= 0.83 t= 4.24 P=0.002	β= 0.71 t= 4.97 P=0.0006	β= 0.40 t= 2.53 P=0.03	β= 0.43 t= 3.02 P=0.01		0.00004	86.6	0
Model 2	β= 0.64 t= 3.34 P=0.008	β= 0.83 t= 6.56 P<0.0001		β= 0.52 t= 3.70 P=0.004	β= 1.03 t= 2.35 P=0.04	0.00006	85.9	0.79
Overall Importance	**0.83**	**1**	**0.38**	**0.49**	**0.32**			

Richness of Intolerant Taxa within restored sections was positively related to increased Total Bank Stability, Pool Variability, and the Number of Rock Vane Structures, and negatively related to increasing substrate Embeddedness, Pool Substrate, and the Number of Root Wad Structures (Table 20). Interestingly, Intolerant Richness was similar between RSC and NCD streams.

Table 20. Top performing models to predict Richness of Intolerant taxa within restored sections of Coastal Plain streams.

Intolerant Richness	Total Bank Stability	Pool Variability	Embeddedness	Pool Substrate	Number of Rock Vane Structures	Number of Root Wad Structures	Prob	Adjusted R^2	ΔAIC
Model 1	β= 0.78 t= 5.27 P=0.0003				β= 2.45 t= 4.09 P=0.002	β= -0.42 t= -4.09 P=0.002	0.0009	69.7	0
Model 2	β= 0.82 t= 5.95 P=0.0001	β= 0.51 t= 3.73 P=0.003	β= 0.17 t= 2.23 P=0.05	β= -0.37 t= -3.51 P=0.006		β= -0.27 t= -2.67 P=0.02	0.0006	77.2	0.15
Model 3	β= 0.82 t= 5.11 P=0.0003	β= 0.59 t= 3.85 P=0.002		β= -0.41 t= -3.35 P=0.006			0.001	68.9	0.38
Overall Importance	**0.98**	**0.36**	**0.23**	**0.31**	**0.51**	**0.59**			

Taxonomic richness of Urban-Tolerant taxa was positively related to increasing the Riffle Frequency/Channel Sinuosity and negatively related to the %Gravel in restored sections (Table 21). It was additionally influenced by NCD streams already having higher taxonomic richness, including higher Urban-Tolerant richness.

Table 21. Top performing models to predict Richness of Urban-Tolerant taxa within restored sections of Coastal Plain streams.

Urban-Tolerant Richness	Riffle Frequency Channel Sinuosity	Restoration Type	% Gravel	Prob	Adjusted R^2	ΔAIC
	β= 0.52	β= -4.86	β= -0.06			
	t= 4.46	t= -3.53	t= -2.04	0.0001	79.1	0
Model 1	P=0.001	P=0.005	P=0.07			
	β= 0.55	β= -2.80				
	t= 4.20	t= -2.68		0.0001	73.7	0.13
Model 2	P=0.001	P=0.02				
Overall Importance	**0.98**	**0.9**	**0.49**			

Coastal Plain EPT Richness

EPT Richness in Coastal Plain streams was complicated by a significant section X restoration type interaction (Model Averaged ANOVA, P=0.01; Table 16). However, the nature of the interaction still allows for interpreting the main effects. Restored sections had significantly higher EPT Richness than the upstream unrestored sections (P<0.05), but were not different from sections downstream of the restorations. In addition, NCD-dominant restorations had significantly higher EPT Richness than RSC restorations (P<0.05), but that appears to be an artefact of the stream rather than the restoration. As mentioned previously, streams in which RSC restorations were performed naturally had lower richness measures, including EPT Richness.

Within restored sections, several restoration activities influenced EPT Richness. We found EPT Richness was positively related to increased numbers of Rock Vane Structures and J-Hook Structures and decreased Sediment Deposition (higher score is better). In contrast, the Number of Root Wads was associated with decreases in EPT Richness as well as being in an RSC-dominated restoration (Table 22).

Table 22. Top performing models to predict Richness of EPT taxa within restored sections of Coastal Plain streams.

EPT Richness	Number of Root Wads	Number of Rock Vane Structures	RSC Restoration Type	Total Number of J-Hook Structures	Sediment Deposition	Prob	Adjusted R^2	ΔAIC
	β= 0.25	β= 1.18	β= -2.00					
	t= -4.25	t= 3.38	t= -3.91			<0.0001	80.7	0
Model 1	P=0.001	P=0.006	P=0.002					
						<0.0001		
	β= -0.21	β= 1.18		β= 0.53	β= 0.11			
	t= -3.55	t= 3.75		t= 3.64	t= 2.24	<0.0001	83.6	1.93
Model 2	P=0.005	P=0.004		P=0.005	P=0.05			
Overall Importance	**0.83**	**0.81**	**0.6**	**0.4**	**0.38**			

Coastal Plain Taxonomic Diversity Measures

Shannon-Wiener Diversity was significantly lower in RSC-dominant streams (Model Averaged ANCOVA, P=0.047). However, this difference applies to the entire stream and not specifically to the restored sections. The result indicates that streams receiving RSC approaches naturally have lower Shannon-Wiener diversity.

We found no other significant effects or interactions that would be expected if the restoration approach exerted an appreciable effect.

Within restored sections, Shannon-Wiener Diversity was best predicted by four competing models (Table 23). All models explained more than 85% of the variation. Shannon Diversity was positively related to lower amounts (higher scores) of Embeddedness and Sediment Deposition, higher Pool Substrate scores, higher %Boulders, lower D50 substrate particle size, and lower amounts of Cobbles.

Table 23. Top performing models to predict Shannon-Wiener Richness within restored sections of Coastal Plain streams.

Shannon-Wiener Richness	Embeddedness	Sediment Deposition	D50	% Cobble	Pool Substrate	% Boulder	Prob	Adjusted R^2	ΔAIC
Model 1	β= 0.03 t= 4.51 P=0.0009	β= 0.03 t= 4.52 P=0.0009		β= -0.005 t= 3.80 P=0.003			<0.0001	86	0
Model 2	β= 0.04 t= 5.56 P=0.0002	β= 0.03 t= 4.57 P=0.0008	β= -0.002 t= -3.77 P=0.003				<0.0001	85	0.16
Model 3	β= 0.03 t= 4.92 P=0.0006	β= 0.04 t= 5.62 P=0.0002	β= -0.003 t= -4.77 P=0.001			β= 0.01 t= 2.08 P=0.06	<0.0001	89	0.59
Model 4	β= 0.04 t= 6.53 P<0.0001	β= 0.03 t= 4.99 P=0.0005	β= -0.003 t= -4.73 P=0.0008		β= 0.01 t= 2.05 P=0.07		<0.0001	89	0.71
Overall Importance	**0.99**	**0.94**	**0.57**	**0.41**	**0.25**	**0.18**			

Margalef Richness was not different among sections, Restoration Types, nor were there any significant interactions with Restoration Age, ISC, or Restoration Length (Model Averaged ANCOVA, P>0.05). Within restored sections, a number of restoration activities influenced Margalef Richness (Table 24). We found positive relationships with lower Sediment Deposition (higher scores are good), greater Total Bank Stability, and more Vane Structures and J-Hook Rock Structures. RSC-restorations tended to have lower Margalef Richness overall.

Table 24. Top performing models to predict Margalef Richness within restored sections of Coastal Plain streams.

Margalef Richness	Sediment Deposition	Total Bank Stability	Number of Vane Structures	RSC Restoration Type	Number of J-Hook Rock Structures	Prob	Adjusted R^2	ΔAIC
Model 1	β= 0.11 t= 3.90 P=0.002	β= 0.11 t= 2.79 P=0.02		β= -0.76 t= -3.29 P=0.007		0.0002	78	0
Model 2	β= 0.14 t= 5.80 P=0.0002	β= 0.10 t= 2.69 P=0.02	β= 0.09 t= 3.34 P=0.007		β= 0.19 t= 2.35 P=0.04	<0.0001	86	0.97
Overall Importance	**0.99**	**0.65**	**0.39**	**0.38**	**0.28**			

In summary, we found very limited evidence of restoration activities improving the taxonomic richness of benthic macroinvertebrates in Coastal Plain streams. EPT Richness was higher in NCD restorations than upstream, but that was the only significant improvement. Neither NCD-, nor RSC-dominant approaches noticeably increased total richness or the constituent parts: richness of intolerant taxa or richness of urban-tolerant taxa. Across all streams and sections, pH was the dominant driver and was related to higher richness as pH increased. Within restored sections, restoration activities that promote greater Velocity Depth Pool Variability and Total Bank Stability, while reducing sediment deposition may increase taxonomic richness and richness of intolerant taxa. Additionally, Vane Structures, especially Rock Vanes, and J-Hook Structures are associated with increasing richness measures, while Root Wads may limit richness. Finally, streams chosen for RSC-dominant restorations tend to have lower richness, likely even before restoration activities begin.

Coastal Plain Physical Habitat Measures

PHI scores in restored sections were significantly greater than those in the unrestored upstream sections (Model Averaged ANOVA, P=0.0002), but did not differ from downstream sections. In addition, PHI scores in NCD-dominant restorations were significantly greater than in RSC-dominant restorations (Model Averaged ANOVA, P=0.04). We found no significant interactions among section, Restoration Type, ISC, Restoration Age, or Restoration Length. However, there was a weak, non-significant trend for a Restoration Type X section interaction because RSC-dominant restorations had low PHI scores (Figure 13).

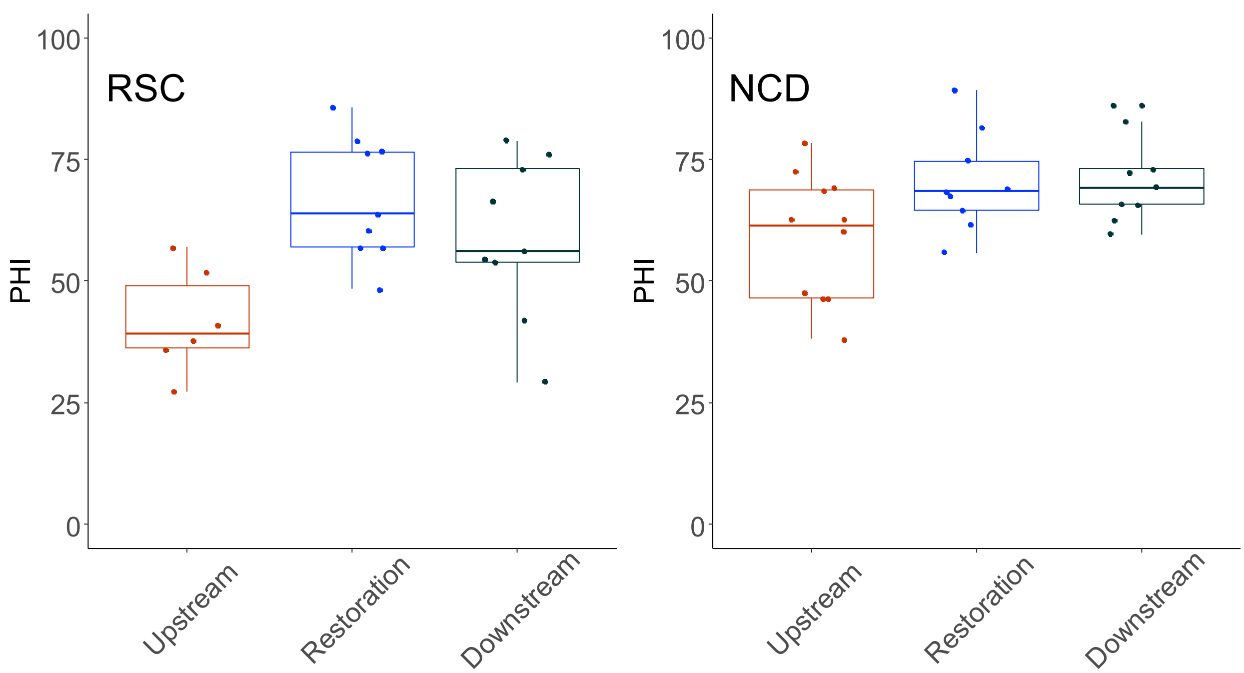


Figure 13. Physical Habitat Index scores for Upstream, Restored, and Downstream sections in Coastal Plain streams. Note that the figure does not incorporate the stream-specific effects that were modeled in the statistical analysis.

Many habitat measures were significantly different between restored and upstream or downstream sections, and almost all of these are generally regarded as improvements over unrestored conditions (Table 25). Restoration-induced changes improved biotic habitat conditions (Epifaunal Substrate, Velocity Depth Pool Variability, Riffle Frequency Channel Sinuosity, Pool Variability), decreased sedimentation (Embeddedness, Sediment Deposition, % Fines, % Sand, % Sand and Silt), protected streambanks (Total Bank Stability, Total Bank Vegetative Protection), and provided larger substrates (% Cobble, % Boulder, D50, D84) compared to upstream unrestored sections. Many of these differences appeared to also carry into the sections downstream of restorations. We are not sure the extent to which the downstream effect is due to restoration or due to increasing stream size since many of these streams are small.

In addition to the overall restoration effect, we also found several habitat differences between NCD and RSC dominant approaches (Table 25). In all but one case of significant differences, NCD-dominant approaches had habitat responses generally considered to be better than RSC. NCD-dominant restorations had significantly higher (better) scores for Epifaunal Substrate, Embeddedness, Velocity Depth Pool Variability, Riffle Frequency Channel Sinuosity, Total Bank Protection, and Pool Variability. Similarly, NCD-dominant restorations had significantly lower (better) % Fines and % Sand and Silt. Only in the case of % Gravel did RSC-dominant restorations have better scores than NCD.

Table 25. Analysis results comparing specific habitat measures among Upstream, Restored, and Downstream sections in Piedmont streams. Comparisons with a "+" indicate that the first section mentioned is significantly greater than the second, while a "-" indicates significantly lower, and NS = Not Statistically Significant.

Habitat Attribute	Restored vs Upstream	Restored vs Downstream	Downstream vs Upstream	NCD vs RSC
Epifaunal Substrate	+	+	NS	+
Embeddedness	+	NS	+	+
Velocity Depth Pool Variability	+	NS	+	+
Channel Alteration	NS	NS	NS	NS
Sediment Deposition	+	NS	+	NS
Riffle Frequency Channel Sinuosity	+	NS	+	+
Total Bank Stability	+	+	+	NS
Total Bank Vegetative Protection	+	NS	NS	+
Total Width Undisturbed Vegetative Zone	+	NS	+	NS
Pool Substrate Characterization	NS	NS	NS	NS
Pool Variability	+	+	+	+
Sinuosity	NS	NS	NS	NS
% Fines	-	NS	-	-
% Sand	-	NS	-	NS
% Sand and Silt	-	NS	-	-
% Gravel	NS	NS	NS	+
% Cobble	+	NS	+	NS
% Boulder	+	NS	+	NS
D50	+	NS	+	NS
D84	+	NS	+	NS

From a habitat perspective in the Coastal Plain, we view NCD-dominant restorations as a success and RSC-dominant streams as improved.

DISCUSSION, INTERPRETATIONS, AND POTENTIAL APPLICATIONS

The success of stream restorations is often in the eye of the beholder (Jahnig et al. 2011). What we choose to examine and the response measures used can have substantial effects on how the outcomes are evaluated.

For example holding urban streams to standards that are not attainable (e.g., Stranko et al. 2012) is guaranteed to result in a negative outcome. In contrast, focusing only on the human dimensions aspects may produce wildly 'successful' outcomes (e.g., Chou 2016) that remain ecologically derelict. Sometimes such 'loaded' comparisons are necessary to highlight a critical issue or unmask false expectations that otherwise perpetuate unrealistic goals. However, optimism or pessimism must be balanced with realism so that results are taken seriously. In our current research, we tried to achieve a balance between using high ecological standards, which are unlikely to be met yet promoted as reasons for restorations, with more nuanced measures that recognize urban stream restorations might attain some increase in ecological or hydro-geomorphic function compared to no intervention. Our goals in this section are to highlight the main points and to provide recommendations to aid future urban stream restoration efforts.

We found substantially divergent results based on two broad success measures applied across a common set of restoration projects in the Coastal Plain and Piedmont regions of Maryland's Western Shore. Physical aspects of streams were substantially improved by restoration activities, particularly in the Coastal Plain. For these geomorphic components, we rate the restoration activities a success and believe that these projects largely achieved positive outcomes. Restorations stabilized streambanks and the channel, created more heterogeneous habitats, and improved conditions for the biota.

Unfortunately, the ecological aspects rarely improved despite the improved physical measures. We therefore reject our hypothesis that stream restorations improve overall ecological condition or even its subcomponents. Benthic macroinvertebrate communities in restored sections remained similar to unrestored sections on the same stream and were significantly dissimilar to MBSS Sentinel Sites. Similarly, the numerous metrics used in ecological assessments also showed a lack of response. The non-response even extended to measures examining urban-tolerant taxa, which we thought might show benefits. The few ecological improvements observed seemed idiosyncratic and could not be attributed to any broad classes of restoration technique or other attributes that we measured. There simply were few ecological differences between restored and unrestored sites. In fact, the unrestored sections upstream were often ecologically better than the restored sections or those downstream. While it is entirely possible that we did not detect some specific positive outcomes in specific restorations, it is impossible to determine if such rare improvements are real, by chance, or due to sampling bias because we cannot sample the entire biodiversity in streams. We unfortunately conclude that urban stream restorations on balance do not improve ecological condition of benthic macroinvertebrates, a key measure of stream health. As discussed in detail below, we speculate that the disruptions caused by restoration activities may do more harm than good, particularly in Piedmont streams as watershed ISC increases.

Physical Aspects of Urban Stream Restorations

Attributes of stream habitat and channel geomorphology were substantially improved in restored sections compared to upstream unrestored sections and to those downstream of the restorations. We fully expected to see these differences because modern urban stream restoration is mostly about channel stabilization and revision or reconfiguration while striving for a more natural appearance. Nonetheless, statistically detecting improvements was a validation for both our study design and for urban stream restoration as a practice in Maryland. Stream restoration projects here and in other regions have detected few if any geomorphic improvements (Violin et al. 2011; Laub et al. 2012), so our results are a positive development. The lack of improvements in Maryland stream restorations in Laub et al. (2012) may also be due to the heterogeneity metrics used, whereas we focused on EPA assessment metrics targeting stabilization and general habitat for biota. Thus, our measures are less about true heterogeneity or ecological niche diversity and more about geomorphic indicators of stable or less degraded streams that ultimately influence the biota.

Despite mixed results in the published literature, stream restorations can be effective at stabilizing eroded banks and returning geomorphic conditions to better resemble those in less disturbed watersheds (Jähnig et al. 2010). In restored sections in both Piedmont and Coastal Plain streams, physical attributes associated with sediment reduction, bank stability, and larger substrate particle sizes improved significantly above ambient conditions upstream. Larger substrate particle sizes generally reflect a more stable stream channel with lower rates of erosion and sedimentation, likely with less sediment carried downstream into receiving waters and ultimately the Bay. Bank and channel stabilization are also important for protecting human infrastructure both instream (e.g., bridges, buried lines and pipes) and streamside (houses and roads).

The observed physical improvements were usually larger and more widespread for restorations in the Coastal Plain. The PHI scores in Coastal Plain restorations were significantly higher than in the upstream unrestored sections, and the higher PHI scores extended to downstream sections as well. In fact most of the restoration-associated physical improvements shown in Table 25 also extended into sections downstream of restorations and might indicate value-added uplift benefits. Compared to Piedmont streams, Coastal Plain streams had substantially more habitat measures that responded positively to restoration activities and included more measures descriptive of biotic habitats. In contrast, we found fewer positive responses in the Piedmont, no differences in PHI scores between restored and unrestored sections, and few examples of improvements extending into downstream sections. The PHI is an aggregate habitat index reflecting the contributions of many different measures of fine sediment, bank and channel stability, and overall habitat quality for the biota. The lack of PHI differences among Piedmont sections likely reflects fewer overall differences in individual habitat measures as well as potentially fewer improvements to make in Piedmont streams.

Stream Restoration and Ecological Uplift

While the restorations substantially improved habitat measures and presumably improved conditions for benthic macroinvertebrates, we saw few actual improvements in the biota. Across both the Piedmont and Coastal Plain fewer than half of the stream restorations had ecological attributes that were better than the unrestored sections upstream. Many of these 'improvements' were only fractionally better than the upstream scores (Tables 6 and 17) and fall within the margin of error of approximately +/- 0.3 for BIBI calculations (Versar 2013). Realized ecological uplift was therefore minimal for most of the sites, and even the stream with the highest BIBI score had a substantially lower score in the restored section than in the unrestored section upstream. Similarly, restored sections did not have higher richness or diversity measures than unrestored sites and follow similar trends elsewhere (Violin et al. 2011; Stranko et al. 2012). The disappointing amounts of realized ecological uplift limit expectations for the potential uplift. The limitations become even more pronounced when comparing restored sites against MBSS Sentinel Sites, which represent streams in lesser disturbed watersheds. However, many sites did have taxa richness equal to or above values for MBSS Sentinel Sites and show that there is substantial potential uplift in this area. Nonetheless, the lack of differences between restored and upstream sections suggest there is little evidence that urban stream restorations can produce meaningful improvements in traditional measures of stream condition as measured with benthic macroinvertebrates. Unfortunately, the possibility of restoring the ecology of urban streams to resemble conditions of streams in lesser disturbed watersheds is very limited.

The lack of differences in IBI, richness, and diversity metrics also translated into no differences in overall community structure among our treatment sections. ADONIS analyses showed very similar benthic macroinvertebrate communities in upstream, downstream, and restored sections in both regions. In contrast, all of these sections were significantly dissimilar from MBSS Sentinel Sites and indicate substantial differences between restored sites and our expectations for restored sites. Surprisingly, differences in community structure between urban and MBSS sites extended to the subgroup of urban-tolerant taxa and suggests that urban streams are different in more ways than just the absence of sensitive taxa. Expecting similar community structure between restored urban streams and minimally disturbed MBSS Sentinel Sites may be unrealistic given results elsewhere (Violin et al. 2011; Stranko et al. 2012; Haase et al. 2013, but it provides important context for setting realistic expectations and for determining if restoration-associated shifts in community structure are in a desired direction. In this context, we saw no shifts and again conclude there is no evidence for ecological uplift due to restoration activities.

Our results suggest that restoration activities do not mitigate the reasons causing the ecological declines. Stream restoration activities may improve structural heterogeneity and habitats, but there is limited evidence for positive ecological responses when streams are already degraded (Palmer et al. 2009; Lepori et al. 2005; Haase et al. 2013). Unfortunately, we noticed the same pattern of limited ecological response despite geomorphic improvements. In essence, the Field of Dreams (Palmer et al. 1997) was built, but nothing came.

Many possible factors might explain why the biota do not respond to improved physical conditions. Urban restorations usually reside within a landscape surrounded by altered land uses and high ISC. Watershed-scale ISC is an important determinant of where benthic taxa can occur (Utz et al. 2009) and was the dominant predictor of BIBI scores and richness in Piedmont streams. The numerous negative influences associated with ISC likely create chemical, thermal, or hydrologic conditions outside of the physiological limits of many benthic taxa. Urban streams are warmer (Somers et al. 2013) and stormflow heat pulses can extend >1km downstream into forested areas (Somers et al. 2016) as can chemical pulses such as road salt. Even the best physical reconstruction of stream channels may not mitigate against such acute pulses associated with stormflows. Similarly, flashy hydrographs and associated bed mobility in urbanized streams may limit many taxa. While most aquatic taxa have evolved to withstand occasional flooding, the increased flood frequencies encountered in urban streams may be too much. Restored sections may also suffer from a recolonization deficit with no organism dispersal from donor streams (Sundermann et al. 2011; Tonkin et al. 2014) because urbanization and sprawl tend to form concentrated areas of higher ISC and may result in numerous degraded streams in proximity to a focal stream.

The limited spatial scales of stream restoration activities may also influence the potential for ecological uplift. Because stream restorations are expensive and often involve private lands, only small amounts of the total stream lengths are usually restored. These reach-scale efforts may not adequately address larger-scale influences such upstream land use or upstream habitat quality (Lorenz and Feld 2013) and may not provide sufficient high-quality area to harbor enough individuals to maintain viable populations even if the restoration itself is sufficient. Surprisingly however, ecological uplift measures were neutral to negatively associated with Restoration Length. Piedmont streams had lower Total Richness, Intolerant Richness, EPT Richness, and Margalef Richness as Restoration Length increased, while Coastal Plain streams were non-responsive. We believe that the invasiveness of restorations may actually hinder recovery and the larger the restoration, the greater the disruption of substrates and depletion of trapped organic matter that serves as infrastructure for the ecosystem.

Just as humans require recovery periods from invasive surgery, restored streams probably require a recovery period, and the greater the invasiveness, the longer the recovery period. Even though benthic invertebrates can rapidly colonize restored areas, community succession may require many years (Winking et al. 2016).

Thus, ecological uplift measures should trend higher in older restorations. Unfortunately, we found little support for time related recovery. Evidence of a recovery phase would have produced a significant Section X Restoration Age interaction, but we found none across the 14 different analyses comparing restored against unrestored sections. Focusing only on restored sections and ignoring upstream responses, Restoration Age was associated with increased richness of intolerant taxa and decreased richness of urban-tolerant taxa in Piedmont streams. Thus, a slight recovery signal may be present, but even the oldest restorations were not noticeably improved compared to their upstream counterparts. The signal is further complicated because we had very few old restorations, and these also tended to be in watersheds with lower ISC.

Inadequate monitoring might also explain divergent responses in restoration measures. Until the Clean Water Act revision of 2008, failure to monitor post-restoration was a valid criticism and limited our analysis abilities (Hassett et al. 2005; Bernhardt et al. 2007). Currently, there may be too much monitoring and not enough alignment with restoration objectives. Inadequate monitoring density or measuring the wrong attributes may prevent change detection or learning from differences in restoration approaches. Our sampling design acknowledges, and is limited by, the lack of pre-restoration data and no long-term post-restoration monitoring. However, such strong designs require many years of sampling and makes implementation difficult. Our design matches restored sections against unrestored sections upstream and downstream within a single year. Constraining comparisons to within-stream differences allows us to factor out much of the stream-to-stream variation that limits other designs as does single year sampling for annual variability. Any restoration-associated changes should be detected between restored and upstream of restored sections. Similarly, carry-over effects of the restoration should be detected as differences between upstream and downstream sections. Aside from only a single year of sampling, the major limitation with our design is the species-area relationship, which implies that some taxa may not be collected in our sampling. Nonetheless, most taxa will be detected because we employed a 300 individual subsample, which is triple that of the MBSS program. We are therefore reasonably confident that our results truly reflect conditions and differences among restorations aside from the very rare taxa.

Recommendations and Potential Applications

Although we found few examples of ecological uplift in restored sections, several restoration-specific activities influenced conditions and may guide future restoration practice. Most importantly, expectations for ecological uplift must be limited when restorations are conducted in watersheds with high ISC. Unfavorable conditions associated with ISC are likely to constrain ecological benefits provided by restorations. In these situations, stream restoration should be motivated more by infrastructure protection, channel stabilization, or aesthetics and human dimensions than by the desire to improve the ecosystem. Justifying degrading activities by claiming that restoration will solve the problems the activity caused is untrue and will lead to misdirected human and financial resources. The steep declines in IBI and richness in restored sections as ISC increases are particularly troubling and suggest that restorations in high ISC watersheds may do more ecological harm than good. Similarly, the amount of human intervention in restorations may also be counter-productive. Ecological measures tended to decrease with Restoration Length and with the Total Number of Structures installed. We view both as surrogate measures of invasiveness and speculate that higher invasiveness and the long recovery times limit the potential for ecological uplift. While restored stream ecosystems may eventually recover given enough time, a more likely scenario is the channel will destabilize and the stream will need more intervention before acceptable ecological recovery can occur.

Despite the negative tone of this section, not all restoration actions are negative. We found several relationships that may help to guide practice. Unfortunately, the different regions responded differently and so a single recommendation is not possible. Within Coastal Plain restorations, Vanes and J-Hook structures were beneficial, while Root Wads and Step Pools were detrimental. Activities that reduce deposition of fine sediments, increase bank stability, or increase particle substrate size tended to improve ecological measures. Most benthic invertebrates associated with improved stream 'health' occupy riffle habitats with more clean gravels. In comparison, ecological measures in Piedmont restorations also responded positively to decreased fine sediments and more gravel, but larger particle sizes were detrimental. Piedmont streams also responded favorably to increased Step Pools and Vanes. However, the Total Number of Structures and longer Restoration Lengths, which may be surrogates for the degree of intervention and disruption, decreased ecological measures. Piedmont restorations also tended to improve with Restoration Age indicating that slow recovery may be occurring.

With respect to restoration types, comparisons must be tempered by the nature of the streams in which they occur. RSC-dominant restorations clearly had lower ecological condition, and our observations during sampling predicted such outcomes. Dissolved oxygen levels in some RSC sites were already at or below 5ppm when we sampled in April. Such stressful anoxic conditions will limit ecological uplift and preclude the sensitive benthic taxa. However as a group, streams receiving RSC approaches already had substantially lower ecological measures in unrestored sections and represent a set of highly degraded streams. Thus, the RSC streams that we sampled were already at low ecological condition compared to streams receiving NCD approaches, and comparing the effectiveness of approaches is not appropriate. In relative terms, RSC-dominant restorations performed similarly to NCD-dominated; both showed limited to no ecological uplift due to restoration activities. While other aspects of RSC designs may be valuable (Filoso 2015), we believe RSC-dominant approaches will not produce ecological uplift in benthic invertebrates because the conditions are not physiologically favorable. Our impression is that amphibians or other air breathing groups may benefit from RSC approaches, but those requiring higher dissolved oxygen levels will not.

To conclude, urban stream restorations improved the physical conditions, but we found few ecological benefits in the 40 projects examined. Urban river restorations in France, and Europe in general, are more focused on riparian and societal objectives and acknowledge that ecological recovery will be limited (Hamed et al. 2017). We think adopting a similar approach is warranted for urban stream restorations in Maryland. While the practitioners may realize the ecological limits, the public and regulators do not. The consequences of degradation and the limitations of restoration need to be more effectively communicated to non-practitioners in order to develop more realistic expectations.

Literature Cited

Barbour, M. T. M. T., Gerritsen, J., Snyder, B. D., & Stribling, J. B. (1999). *Rapid bioassessment protocols for use in streams and wadeable rivers*. *USEPA, Washington*. doi:10.1093/gerona/60.10.1351

Becker, J. F., Endreny, T. A., & Robinson, J. D. (2013). Natural channel design impacts on reach-scale transient storage. *Ecological Engineering*, *57*, 380–392. doi:10.1016/j.ecoleng.2013.04.051

Bendor, T., Lester, T. W., Livengood, A., Davis, A., & Yonavjak, L. (2015). Estimating the size and impact of the ecological restoration economy. *PLoS ONE*, *10*(6), 1–15. doi:10.1371/journal.pone.0128339

Benke, A. C. (1990). A Perspective on America's Vanishing Streams. *Journal of the North American Benthological Society*. doi:10.2307/1467936

Berg, J., Burch, J., Cappuccitti, D., Filoso, S., Fraley-McNeal, L., Goerman, D., … Winters, J. (2012). Recommendations of the Expert Panel to Define Removal Rates for Individual Stream Restoration Projects, 35–39.Bernhardt ES et al. (2005) Synthesizing U.S. river restoration efforts. Science

Bernhardt ES, Sudduth EB, Palmer MA, Allan JD, Meyer JL, Alexander G, Follastad-Shah J, Hassett B, Jenkinson R, Lave R, Rumps J, Pagano L. 2007. Restoring rivers one reach at a time, results from a survey of U.S. river restoration practitioners. Restoration Ecology 15:482–493

Blakely, T. J., Harding, J. S., Mcintosh, A. R., & Winterbourn, M. J. (2006). Barriers to the recovery of aquatic insect communities in urban streams. *Freshwater Biology*, *51*(9), 1634–1645. doi:10.1111/j.1365-2427.2006.01601.x

Booth, D. B., Roy, A. H., Smith, B., & Capps, K. A. (2016). Global perspectives on the urban stream syndrome. *Freshwater Science*, *35*(1), 412–420. doi:10.1086/684940

Brown, L. R., Cuffney, T. F., Coles, J. F., Fitzpatrick, F., McMahon, G., Steuer, J., … May, J. T. (2009). Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. *Journal of the North American Benthological Society*, *28*(4), 1051–1069. doi:10.1899/08-153.1

Cambell, K., Bennett, K., Adams, C., Collier, D., & Messerly, V. (2017). *The State of Compensatory Mitigation*.

Cizek, A. R., Hunt, W. F., Winston, R. J., Waickowski, S. E., Narayanaswamy, K., & Lauffer, M. S. (2018). Water Quality and Hydrologic Performance of a Regenerative Stormwater Conveyance in the Piedmont of North Carolina. *Journal of Environmental Engineering*, *144*(8). doi:10.1061/(ASCE)EE.1943-7870.0001344

Coles, J. F., Cuffney, T. F., McMahon, G., & Rosiu, C. J. (2010). Judging a Brook by Its Cover: The Relation between Ecological Condition of a Stream and Urban Land Cover in New England. *Northeastern Naturalist*. doi:10.1656/045.017.0103

Donovan, M., Miller, A., Baker, M., & Gellis, A. (2015). Sediment contributions from floodplains and legacy sediments to Piedmont streams of Baltimore County, Maryland. *Geomorphology*, *235*(1), 88–105. doi:10.1016/j.geomorph.2015.01.025

Doyle, M. W., & Douglas Shields, F. (2012). Compensatory Mitigation for Streams Under the Clean Water Act: Reassessing Science and Redirecting Policy. *Journal of the American Water Resources Association*. doi:10.1111/j.1752-1688.2011.00631.x

Filoso S, Palmer MA (2011) Assessing stream restoration effectiveness at reducing nitrogen export to downstream waters. Ecological Applications 21:1989–2006

Filoso S et al. (2015) The Efficacy of Constructed Stream-Wetland Complexes at Reducing the Flux of Suspended Solids to Chesapeake Bay. Environmental Science and Technology

Haase, P, Hering, D., Jähnig, S.C., Lorenz, A.W., Sundermann, A. 2013. The impact of hydromorphological restoration on river ecological status: a comparison of fish, benthic invertebrates, and macrophytes. Hydrobiologia 704:475-488.

Harman W (2012) A Function-Based Framework for stream assessment and restoration projects.

Hamed, A.Z., Greulick, S., Pauleit, S., Wantzen, K.M. 2017. Urban and rural river restoration in France: a typology. Restoration Ecology 25:994-1004.

Harding JS, Benfield EF, Bolstad PV, Helfman GS, Jones EBD III. 1998. Stream biodiversity: The ghost of land use past. Proceedings of the National Academy of Sciences 95:14843-14847.

Hassett B, Palmer M, Bernhardt E, Smith S, Carr J, Hart D. 2005 Restoring watersheds project by project, trends in Chesapeake Bay tributary restoration. Frontiers in Ecology and the Environment 3:259–267.

Hassett BA et al. (2018) Pulling apart the urbanization axis: patterns of physiochemical degradation and biological response across stream ecosystems. Freshwater Science 37:000–000 doi:10.1086/699387

Hassett BA, Palmer MA, Bernhardt ES (2007) Evaluating stream restoration in the Chesapeake Bay watershed through practitioner interviews. Restoration Ecology 15:563–572

Hautier Y et al. (2015) Anthropogenic environmental changes affect ecosystem stability via biodiversity. Science 348:336–340

Hilderbrand, R.H. and R.M. Utz. 2015. Ecological thresholds and resilience in streams. Pages 461-478 in P.M. Rowinski and A. Radecki-Pawlik, eds. Rivers – physical, fluvial, and environmental processes. Springer.

Hilderbrand RH et al. (2010) Applying thresholds to forecast potential biodiversity loss from human development. Journal of the North American Benthological Society

Hilderbrand RH, Watts AC, Randle AM (2005) The Myths of Restoration Ecology. Ecology and Society

Jähnig, S.C., Brabed, K. Buffagni, A., Erba, S., Lorenz, A.W., Ofenböck, T, Verdonschot, P.F.M., Hering, D. 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology 47:671-680.

Jähnig, S.C., Lorenz, A.W., Hering, D., Antons, C., Sundermann, A., Jedicke, E., Haase, P. 2011. River restoration success: a question of perception. Ecological Applications 21:2007-2015.

ohnson, P. T. J., Hoverman, J. T., McKenzie, V. J., Blaustein, A. R., & Richgels, K. L. D. (2013). Urbanization and wetland communities: Applying metacommunity theory to understand the local and landscape effects. *Journal of Applied Ecology*, *50*(1), 34–42. doi:10.1111/1365-2664.12022

Johnson, R. C., Jin, H. S., Carreiro, M. M., & Jack, J. D. (2013). Macroinvertebrate community structure, secondary production and trophic-level dynamics in urban streams affected by non-point-source pollution. *Freshwater Biology*. doi:10.1111/fwb.12090

Kenney, M. A., Wilcock, P. R., Hobbs, B. F., Flores, N. E., & Martínez, D. C. (2012). Is Urban Stream Restoration Worth It? *Journal of the American Water Resources Association*, *48*(3), 603–615. doi:10.1111/j.1752-1688.2011.00635.x

Koryto KM, Hunt WF, Page JL (2017) Hydrologic and water quality performance of regenerative stormwater conveyance installed to stabilize an eroded outfall. Ecological Engineering 108:263–276

Lake PS, Bond N, Reich P (2007) Linking ecological theory with stream restoration. Freshwater Biology 52:597–615.

Lave, R. 2009. The controversy over natural channel design: substantive explanations and potential avenues for resolution. Journal of the American Water Resources Association 45:1519-1532.

Lepori, F., Palm, D., Brännäs, E. Malmqvist, B. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? Ecological Applications 15:2060-2071.

Lorenz, A.W., Feld, C.K. 2013. Upstream river morphology and riparian land use overrule local restoration effects on ecological status assessment. Hydrobiologia 704:489-501.

McGrane, S. J. (2016). Impacts of urbanisation on hydrological and water quality dynamics, and urban water management: a review. *Hydrological Sciences Journal*, *61*(13), 2295–2311. doi:10.1080/02626667.2015.1128084

Merrit, R. W., & Cummins, K. W. (1984). An Introduction to the Aquatic Insects of North America. *Kendall Hunt Publising Co.* doi:http://dx.doi.org/10.1016/B978-0-443-07033-4.50007-0

Morgan, R. P., & Cushman, S. F. (2005). Urbanization effects on stream fish assemblages in Maryland, USA. *Journal of the North American Benthological Society*, *24*(3), 643–655. doi:10.1899/04-019.1

Palmer, M. A., Ambrose, R. F., & Poff, N. L. R. (1997). Ecological theory and community restoration ecology. *Restoration Ecology*, *5*(4), 291–300. doi:10.1046/j.1526-100X.1997.00543.x

Palmer, M.A., Menninger, H.L., Bernhardt, E. 2009. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biology 55(Supplement 1):1-18.

Palmer MA, Filoso S, Fanelli RM (2014) From ecosystems to ecosystem services: Stream restoration as ecological engineering. Ecological Engineering 65:62–70

Palmer MA, Hondula KL, Koch BJ (2014) Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals. Annual Review of Ecology, Evolution, and Systematics 45:247–269

Palmer, M. A., Menninger, H. L., & Bernhardt, E. (2010). River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice? *Freshwater Biology*. doi:10.1111/j.1365-2427.2009.02372.x

Rathore, S. S., Chandravanshi, P., Chandravanshi, A., & Jaiswal, K. (2016). Eutrophication: Impacts of Excess Nutrient Inputs on Aquatic Ecosystem. *IOSR Journal of Agriculture and Veterinary Science*. doi:10.9790/2380-0910018996

Roni, P., Hanson, K., & Beechie, T. (2008). Global Review of the Physical and Biological Effectiveness of Stream Habitat Rehabilitation Techniques. *North American Journal of Fisheries Management*, *28*(3), 856–890. doi:10.1577/M06-169.1

Rosgen, D. (1996). Applied river morphology. *Wildland Hydrology*. doi:10.1109/ICMA.2009.5246620

Roy AH et al. (2016) Urbanization and stream ecology: diverse mechanisms of change. Freshwater Science 35:272–277

Roy, A. H., Rosemond, A. D., Paul, M. J., Leigh, D. S., & Wallace, J. B. (2003). Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). *Freshwater Biology*. doi:10.1046/j.1365-2427.2003.00979.x

Ruhl, J. B., Salzman, J., & Goodman, I. (2009). *Implementing the New Ecosystem Services Mandate of the Section 404 Compensatory Mitigation Program*. *Society*. doi:/10.1039/c3dt52760c

Selvakumar A, O'Connor TP, Struck SD (2010) Role of Stream Restoration on Improving Benthic Macroinvertebrates and In-Stream Water Quality in an Urban Watershed: Case Study. Journal of Environmental Engineering 136:127–139

harp, J. M. (2010). The impacts of urbanization on groundwater systems and recharge. *Aquamundi*. doi:10.4409/Am-004-10-0008

Simon, A., Doyle, M., Kondolf, M., Shields, F. D., Rhoads, B., & McPhillips, M. (2007). Critical evaluation of how the Rosgen classification and associated "natural channel design" methods fail to integrate and quantify fluvial processes and channel response. *Journal of the American Water Resources Association*. doi:10.1111/j.1752-1688.2007.00091.x

Somers, K.A., Bernhardt, E.S., Grace, J.B., Hassett, B.A., Suddoth, E.B., Wang, S., Urban, D.L. 2013. Streams in the urban heat island: spatial and temporal variability and temperature. Freshwater Science 32:309-326.

Somers, K.A., Bernhardt, E.S., McGlyn, B.L., and Urban, D.L. 2016. Downstream dissipation of storm flow heat pulses: a case study and its landscape-level implications. Journal of the American Water Resources Association 52:281-297. https://doi.org/10.1111/1752-1688.12382

Stranko, S.A., R.H. Hilderbrand, and M.A. Palmer. 2012. Comparing the Fish and Benthic Macroinvertebrate Diversity of Restored Urban Streams to Reference Streams. Restoration Ecology 20:747-755.

Sudduth, E. B., Hassett, B. A., Cada, P., & Bernhardt, E. S. (2011). Testing the field of dreams hypothesis: Functional responses to urbanization and restoration in stream ecosystems. *Ecological Applications*. doi:10.1890/10-0653.1

Swan CM, Brown BL (2017) Metacommunity theory meets restoration: Isolation may mediate how ecological communities respond to stream restoration: Isolation. Ecological Applications 27:2209–2219

U.S. Army Corps of Engineers. (2015). *Baltimore District Mitigation Regulatory Sourcebook*. *Clean Water Act section 404*. doi:/10.1017/CBO9781107415324.004

United States Department of Agriculture (USDA), & National Resources Conservation Service (NRCS). (2007). *Rosgen Geomorphic Channel Design*. *Stream Restoration Design National Engineering Handbook*.

United States Environmental Protection Agency. (2009). *National Water Quality Inventory: Report to Congress*. *National Water Quality Inventory: Report to Congress*. doi:/http://www.epa.gov/owow/305b/2004report/

Utz, R.M., R.H. Hilderbrand, and D.M. Boward. 2009. Identifying regional differences in threshold responses of aquatic invertebrates to land cover gradients. Ecological Indicators 9:556-567.

Utz, R.M. and R.H. Hilderbrand. 2011. Interregional variation in urbanization induced geomorphic change and macroinvertebrate habitat colonization in headwater streams. Journal of the North American Benthological Society 31:25-37.

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences*. doi:10.1139/f80-017

Vietz GJ, Walsh CJ, Fletcher TD (2015) Urban hydrogeomorphology and the urban stream syndrome: Treating the symptoms and causes of geomorphic change. Progress in Physical Geography

Versar, Incorporated. 2013. Maryland Biological Stream Survey 2007-2009 quality assurance report. TR-13-38.

Vietz, G. J., Walsh, C. J., & Fletcher, T. D. (2015). Urban hydrogeomorphology and the urban stream syndrome:

Treating the symptoms and causes of geomorphic change. *Progress in Physical Geography*, *40*(3), 480–492. doi:10.1177/0309133315605048

Violin, C. R., Cada, P., Sudduth, E. B., Hassett, B. A., Penrose, D. L., & Bernhardt, E. S. (2011). Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. *Ecological Applications*, *21*(6), 1932–1949. doi:10.1890/10-1551.1

Walsh, C. J., Booth, D. B., Burns, M. J., Fletcher, T. D., Hale, R. L., Hoang, L. N., … Wallace, A. (2016). Principles for urban stormwater management to protect stream ecosystems. *Freshwater Science*, *35*(1), 398–411. doi:10.1086/685284

Walsh, C. J., Fletcher, T. D., & Burns, M. J. (2012). Urban Stormwater Runoff: A New Class of Environmental Flow Problem. *PLoS ONE*, *7*(9). doi:10.1371/journal.pone.0045814

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society*. doi:10.1899/04-028.1

Walsh CJ et al. (2007) Riverine invertebrate assemblages are degraded more by catchment urbanisation than by riparian deforestation. Freshwater Biology 52:574–587

Wenger, S. J., Roy, A. H., Jackson, C. R., Bernhardt, E. S., Carter, T. L., Filoso, S., … Walsh, C. J. (2009). Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. *Journal of the North American Benthological Society*, *28*(4), 1080–1098. doi:10.1899/08-186.1

Williams, M. R., Bhatt, G., Filoso, S., & Yactayo, G. (2017a). Stream Restoration Performance and Its Contribution to the Chesapeake Bay TMDL: Challenges Posed by Climate Change in Urban Areas. *Estuaries and Coasts*. doi:10.1007/s12237-017-0226-1

Williams, M. R., Bhatt, G., Filoso, S., & Yactayo, G. (2017b). Stream Restoration Performance and Its Contribution to the Chesapeake Bay TMDL: Challenges Posed by Climate Change in Urban Areas. *Estuaries and Coasts*, *40*(5), 1227–1246. doi:10.1007/s12237-017-0226-1

Williams, M. R., Filoso, S., Longstaff, B. J., & Dennison, W. C. (2010). Long-Term Trends of Water Quality and Biotic Metrics in Chesapeake Bay: 1986 to 2008. *Estuaries and Coasts*, *33*(6), 1279–1299. doi:10.1007/s12237-010-9333-y

Winking, C., Lorenz, A.W., Sures, B., Hering, D. 2016. Start at zero: succession of benthic invertebrate assemblages in restored former sewage channels. Aquatic Science 78:683-694.

Wohl, E., Lane, S. N., & Wilcox, A. C. (2015). The science and practice of river restoration. *Water Resources Research*, *51*(8), 5974–5997. doi:10.1002/2014WR016874

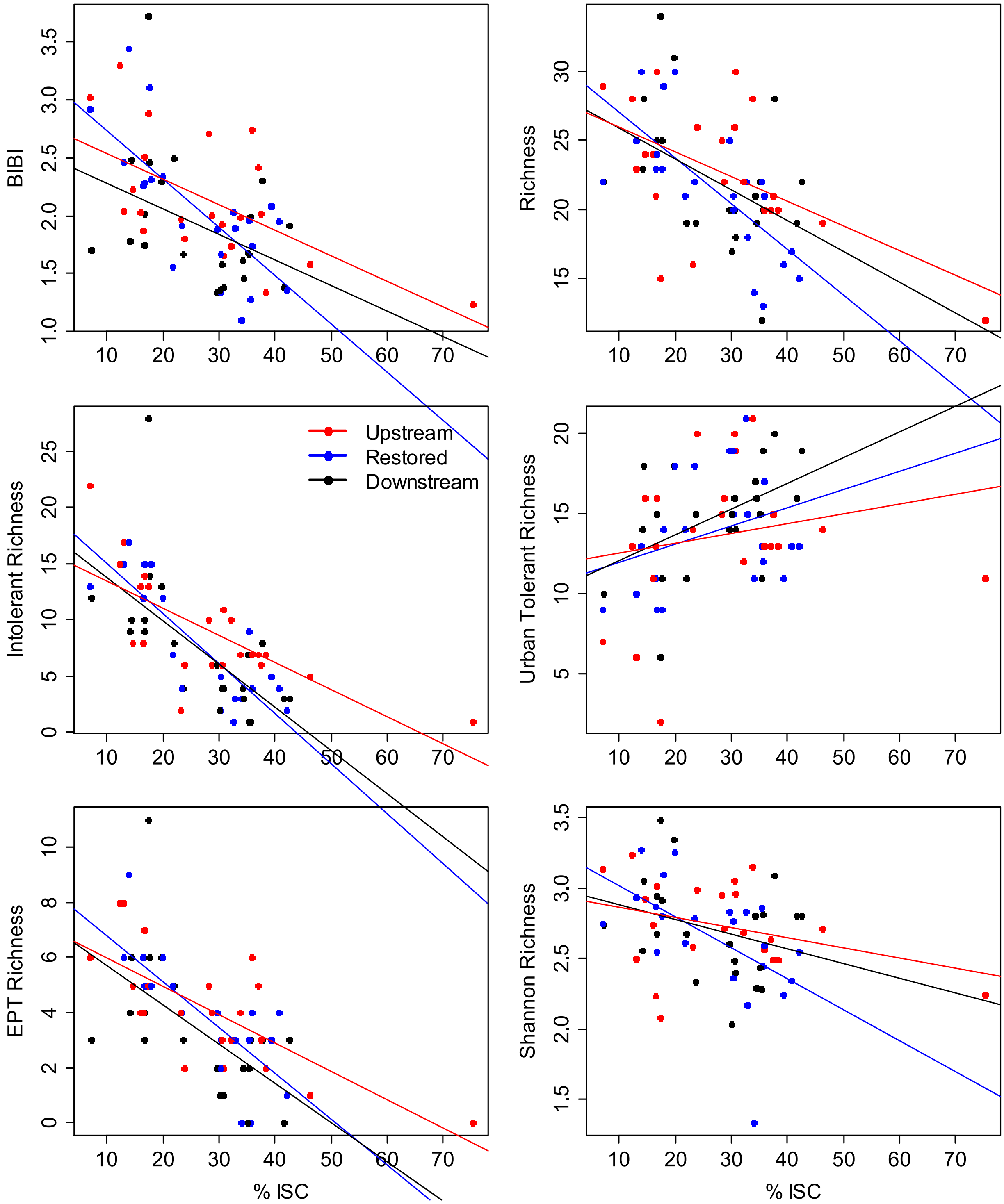
APPENDIX Table A1. Site characteristics for stream restorations sampled in the Coastal Plain region.

Project Name	Latitude	Longitude	Province	County	Restoration Type	Watershed area (sq.mi)	% Forest	% Impervious	Urban	Restoration Age (years)	Restoration Length (ft.)
Bear Branch Stream Stabilization	39.09167	-76.8776	CP	Prince George's	NCD	1.31	35.9	34.16	68.9	4	3500
Biddson Run	39.31186	-76.5458	CP	Baltimore City	NCD	17.8	6.8	14.7	93.4	9	1430
Foster Branch at Trimble Road Restoration	39.40911	-76.3427	CP	Harford	NCD	1.83	39.9	24.5	40.7	1	650
Little Paint Branch Stream Restoration	39.03725	-76.9313	CP	Prince George's	NCD	8.81	20.9	36.3	75.2	12	2400
Muddy Bridge Branch Stream Restoration	39.17557	-76.6441	CP	Anne Arundel	NCD	1	5.5	77.8	97.1	18	3600
Paint Branch Stream Restoration	38.99757	-76.9332	CP	Prince George's	NCD	29.3	21	31.7	18.5	2	3790
Patuxent Run	38.97843	-76.7464	CP	Prince George's	NCD	0.24	0	40	95.5	4	2500
Red House Run Stream Restoration	39.34342	-76.5145	CP	Baltimore	NCD	1.34	2.96	27.9	95.2	5	2500
White Marsh Run A	39.37608	-76.4592	CP	Baltimore	NCD	7.13	14.2	43.8	87	12	1500
Cabin Branch Stream Restoration Project	38.99418	-76.5483	CP	Anne Arundel	RSC	0.18	5.19	60	91.1	3	1700
Church Creek Headwaters/Stream Restoration	38.97517	-76.5415	CP	Anne Arundel	RSC	0.62	1.92	71.4	95.7	0	1500
Central Sanitation Facility Stream Restoration	39.10063	-76.6192	CP	Anne Arundel	RSC	0.19	25.2	32.05	76.2	7	2300
Crofton Tributary Restoration	38.99963	-76.6961	CP	Anne Arundel	RSC	0.33	0.99	39.5	82.8	4	3500
Dividing Creek Outfall retrofit & Stream Restoration	39.05092	-76.5161	CP	Anne Arundel	RSC	0.36	14.2	35.2	59.7	1	1700
Howard's Branch Stream Restoration	39.02098	-76.5490	CP	Anne Arundel	RSC	0.36	55.5	12.3	10.1	13	900
North Branch Cypress Creek Ecological Restoration	39.07587	-76.5355	CP	Anne Arundel	RSC	0.72	1.92	50.2	86	3	3630
Warehouse Creek Stream Restoration	38.93932	-76.5609	CP	Anne Arundel	RSC	0.12	0	57.5	100	0	970
Wilelinor Stream Restoration	38.96625	-76.5425	CP	Anne Arundel	RSC	0.24	19.3	34.5	88.8	14	1311

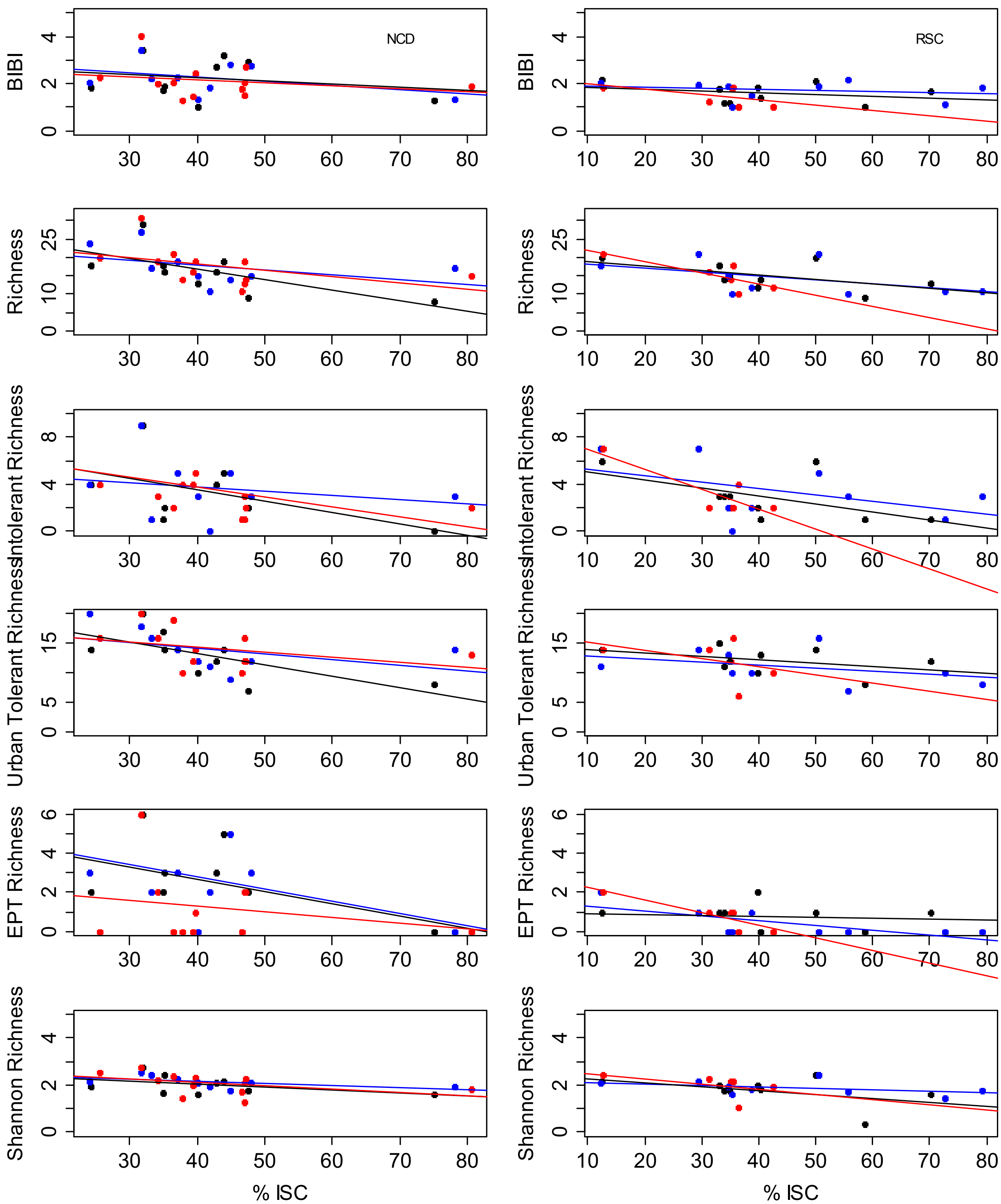
APPENDIX Table A2. Site characteristics for stream restorations sampled in the Piedmont region.

Project Name	Latitude	Longitude	Province	County	Restoration Type	Watershed area (sq.mi)	% Forest	% Impervious	Urban	Restoration Age (years)	Restoration Length (ft.)
Batchellors Run East Stream Restoration Project	39.12213	-77.0338	PD	Montgomery	NCD	0.97	28	10.67	23.4	3	1901
Bryants Nursery Run Stream Restoration Project	39.11994	-77.0079	PD	Montgomery	NCD	1.3	37.8	7.15	38.4	3	1742
Turkey Branch Stream Restoration Project	39.06601	-77.0865	PD	Montgomery	NCD	3.26	7.68	25.8	90.7	9	15019
Upper Northwest Branch Stream Restoration Project	39.11739	-77.0225	PD	Montgomery	NCD	5.6	32	17.1	38.7	3	2165
Batchellors Run Stream Restoration Project	39.11543	-77.0370	PD	Montgomery	NCD	3.39	27.8	3.85	33.5	3	2587
Northwest Branch Stream Restoration (South of Randolph Rd)	39.06070	-77.0203	PD	Montgomery	NCD	22.6	24.6	25.13	54.7	4	5438
Sherwood Forest Stream Restoration Project (Anacostia Package II)	39.07453	-77.0217	PD	Montgomery	NCD	0.9	4.81	10.56	69.1	2	2500
Hawlings River Stream Restoration	39.19531	-77.0408	PD	Montgomery	NCD	15.4	31.6	4.75	11.6	11	2746
Booze Creek Stream Restoration	38.98323	-77.1362	PD	Montgomery	NCD	3.86	0.75	13.7	74.9	3	4646
Stream Valley Drive Stream Restoration	39.18250	-77.1348	PD	Montgomery	NCD	0.35	10.5	3.33	59.9	12	2904
Hollywood Branch Stream Restoration	39.06796	-76.9913	PD	Montgomery	NCD	0.7	5.88	16.4	79	1	4470
NB-1 Cherrywood	39.13070	-77.0819	PD	Montgomery	NCD	1.13	19.1	11.9	70.2	2	3600
NB-3 Manor Run	39.11017	-77.0994	PD	Montgomery	NCD	1	13.9	13.7	69.7	1	6700
RC-2 Mill Creek	39.13370	-77.1505	PD	Montgomery	NCD	1.77	18.3	17.7	73.3	4	3500
PB-109 Upper Paint Branch	39.09435	-76.9958	PD	Montgomery	NCD	0.46	44.6	6.9	48.4	2	2500
NW-4 Cricket Lane	39.08800	-77.0211	PD	Montgomery	NCD	1.06	9.87	13.2	82.9	1	4400
Elmmede Road Stream Restoration	39.28067	-76.8517	PD	Howard	NCD	0.12	6.61	13	86.8	3	1150
Little Patuxent Restoration Projects(Dorsey Hall Outfall and Stream Restoration)	39.24305	-76.8361	PD	Howard	NCD	0.24	32	21.4	71.7	1	1500
Meadowbrook Park Stream Restoration	39.24642	-76.8221	PD	Howard	NCD	1.95	9.58	22.3	81.8	4	1390
Upper Little Patuxant	39.26430	-76.8516	PD	Montgomery	NCD	9.95	24.2	7.8	48.5	1	2500
Windflower Drive Stream Restoration	39.25825	-76.8522	PD	Howard	NCD	0.54	0.34	33.69	80.5	3	1650
Woodlawn (Anacostia Package II)	39.12773	-77.0209	PD	Montgomery	NCD	2.48	32.9	5.8	36.5	3	2500

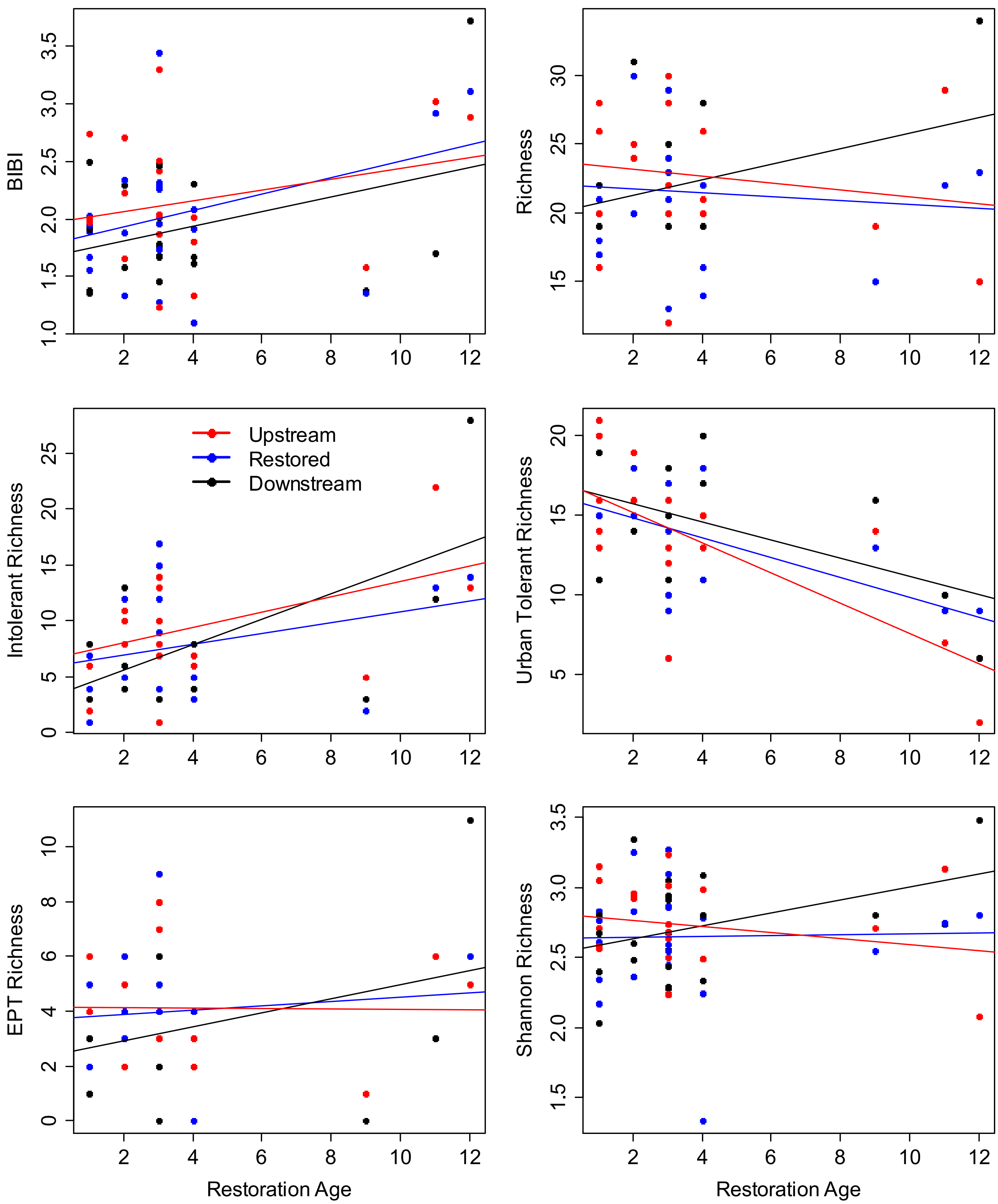
Appendix Figure B1. Relationships between ecological measures and ISC for restored, upstream, and downstream sections in Piedmont streams.



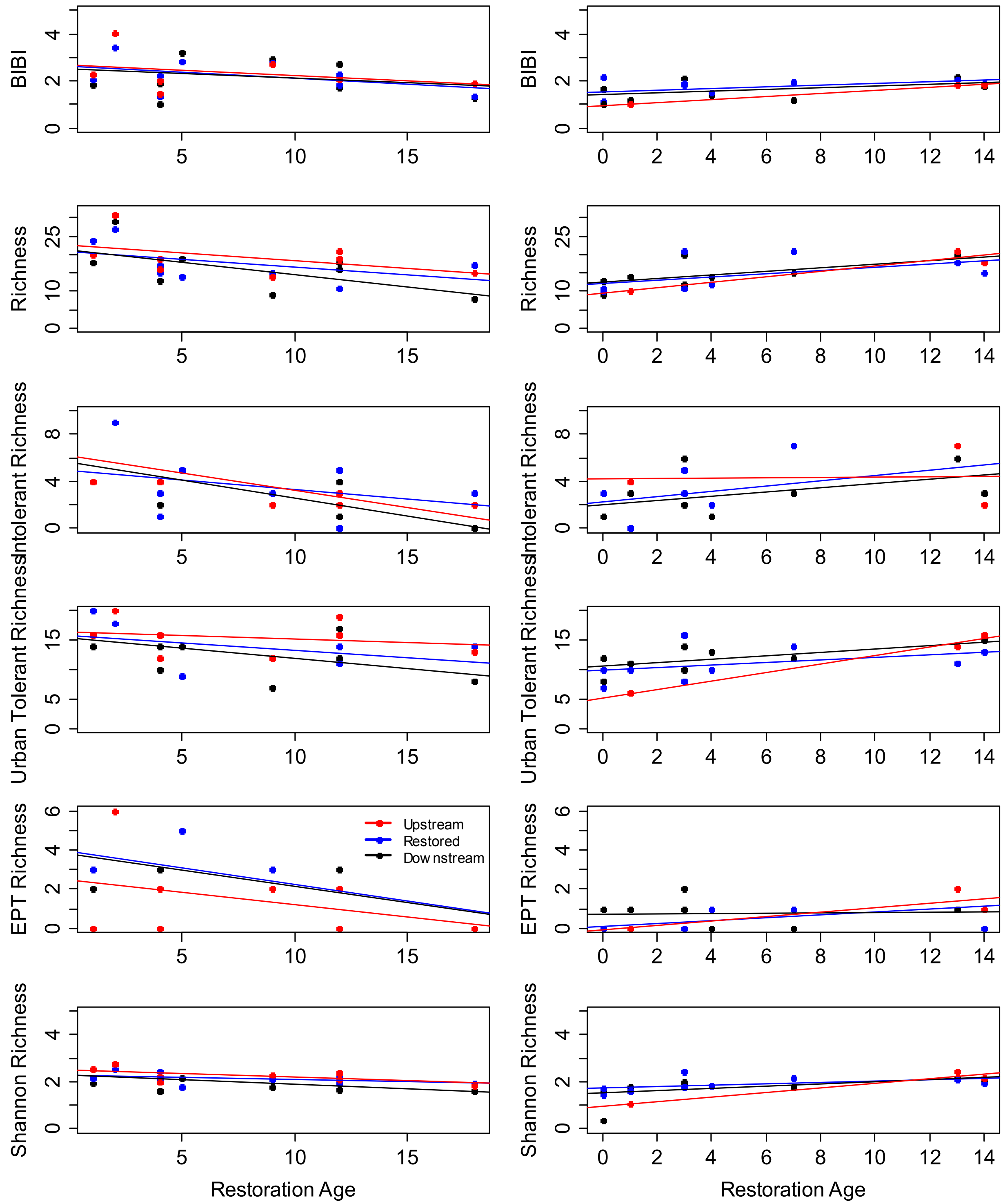
Appendix Figure B2. Relationships between ecological measures and ISC for restored, upstream, and downstream sections in NCD (left column) and RSC (right column) restorations in Coastal Plain streams.



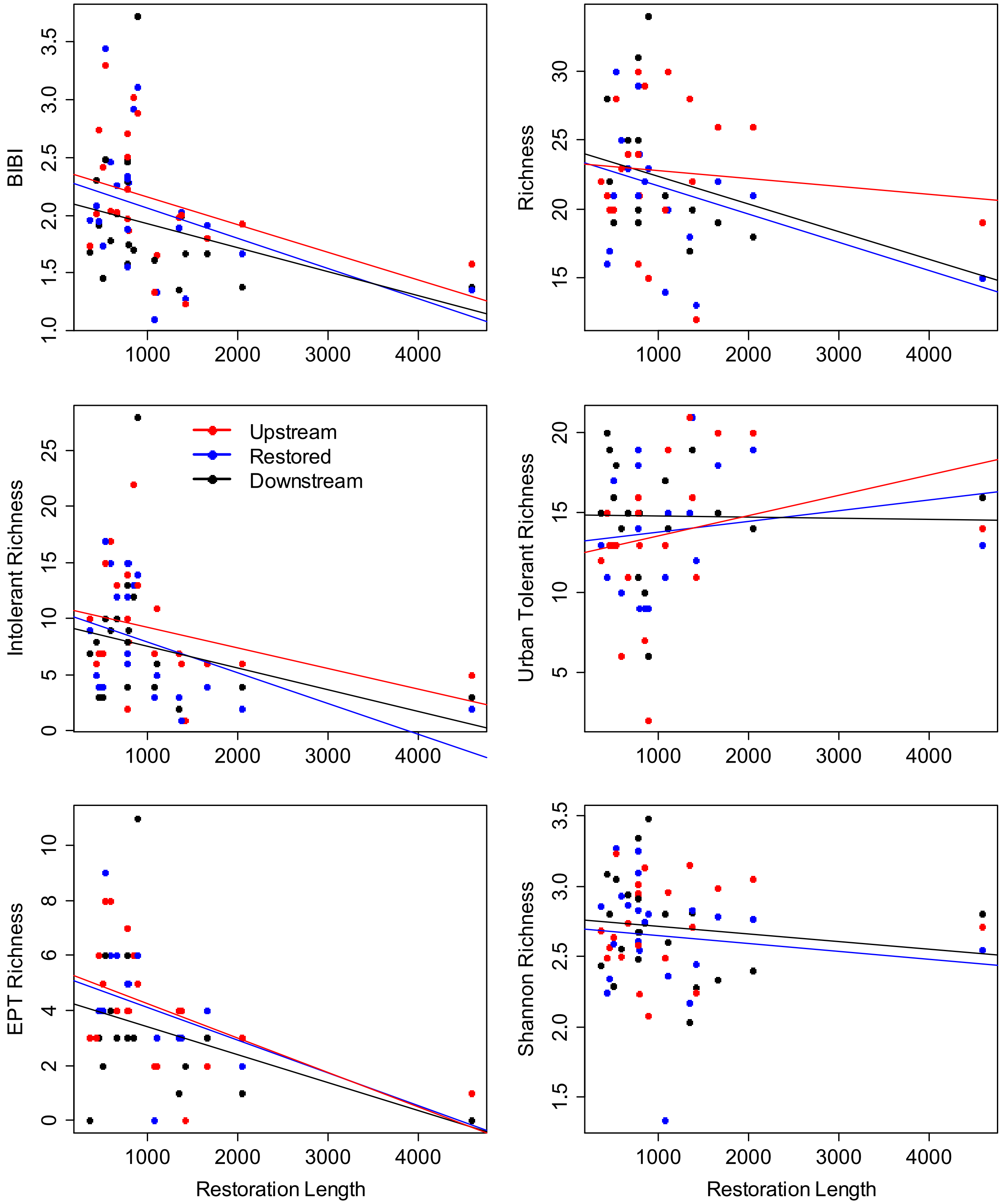
Appendix Figure C1. Relationships between ecological measures and Restoration Age for restored, upstream, and downstream sections in Piedmont streams.



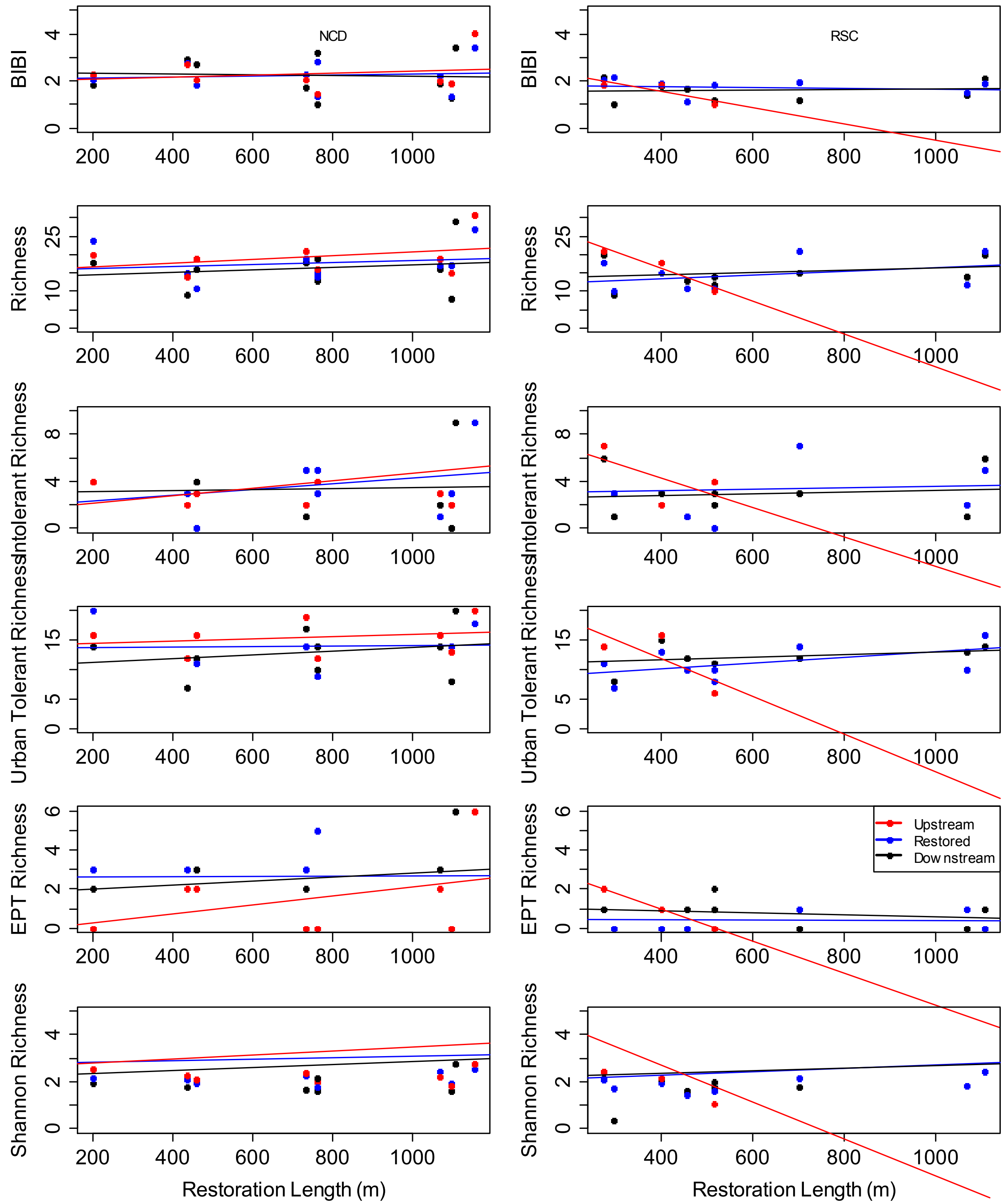
Appendix Figure C2. Relationships between ecological measures and Restoration Age for restored, upstream, and downstream sections in NCD (left column) and RSC (right column) restorations in Coastal Plain streams.



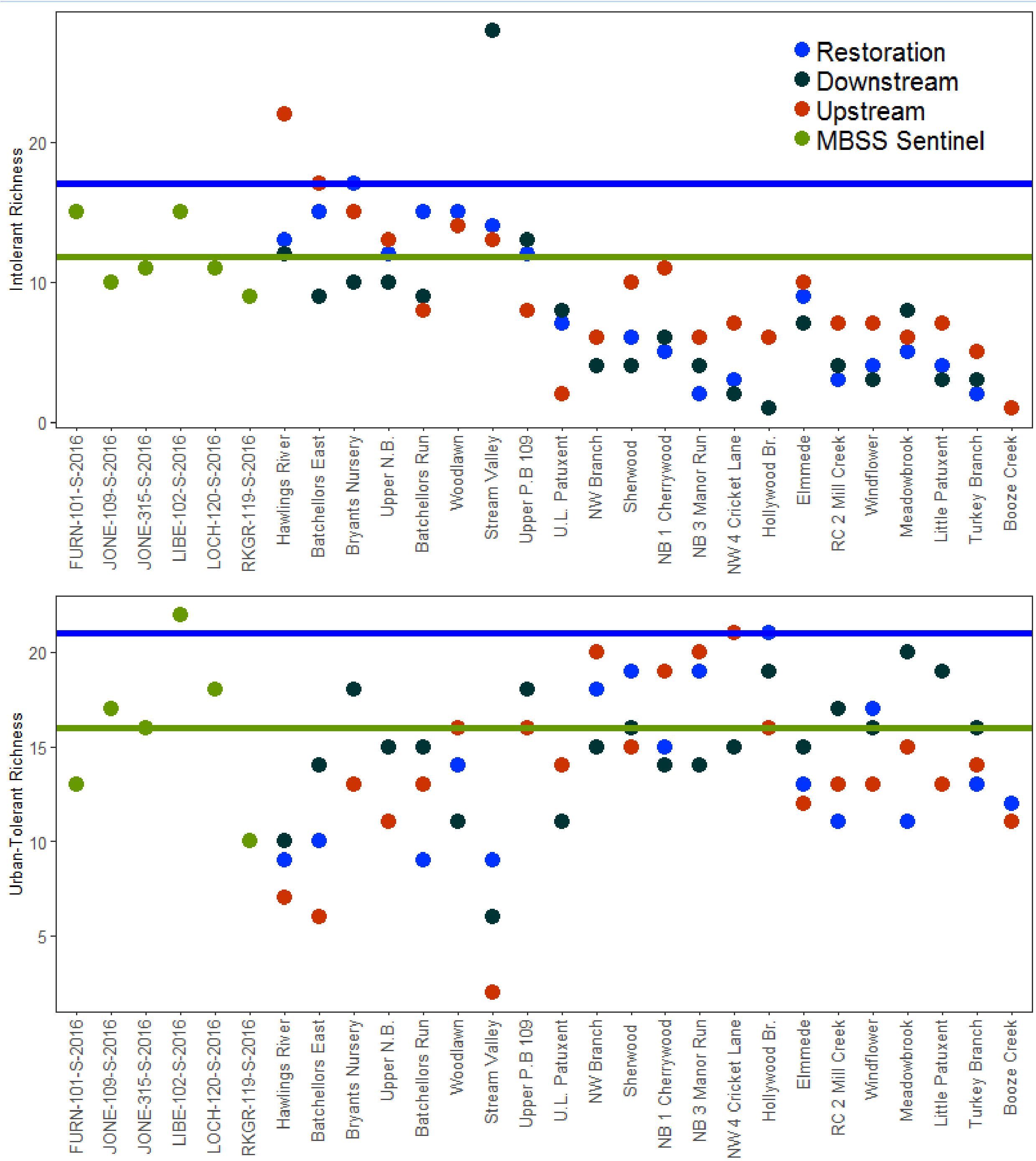
Appendix Figure D1. Relationships between ecological measures and Restoration Length for restored, upstream, and downstream sections in Piedmont streams.



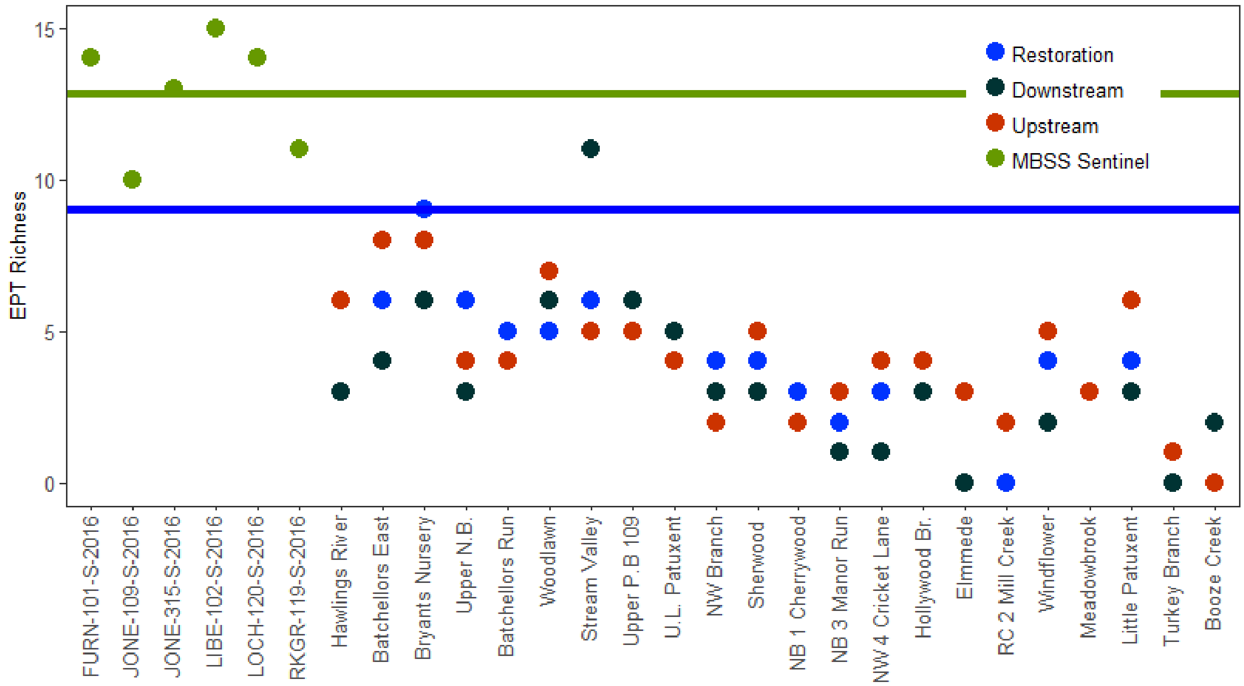
Appendix Figure D2. Relationships between ecological measures and Restoration Length for restored, upstream, and downstream sections in NCD (left column) and RSC (right column) restorations in Coastal Plain streams.

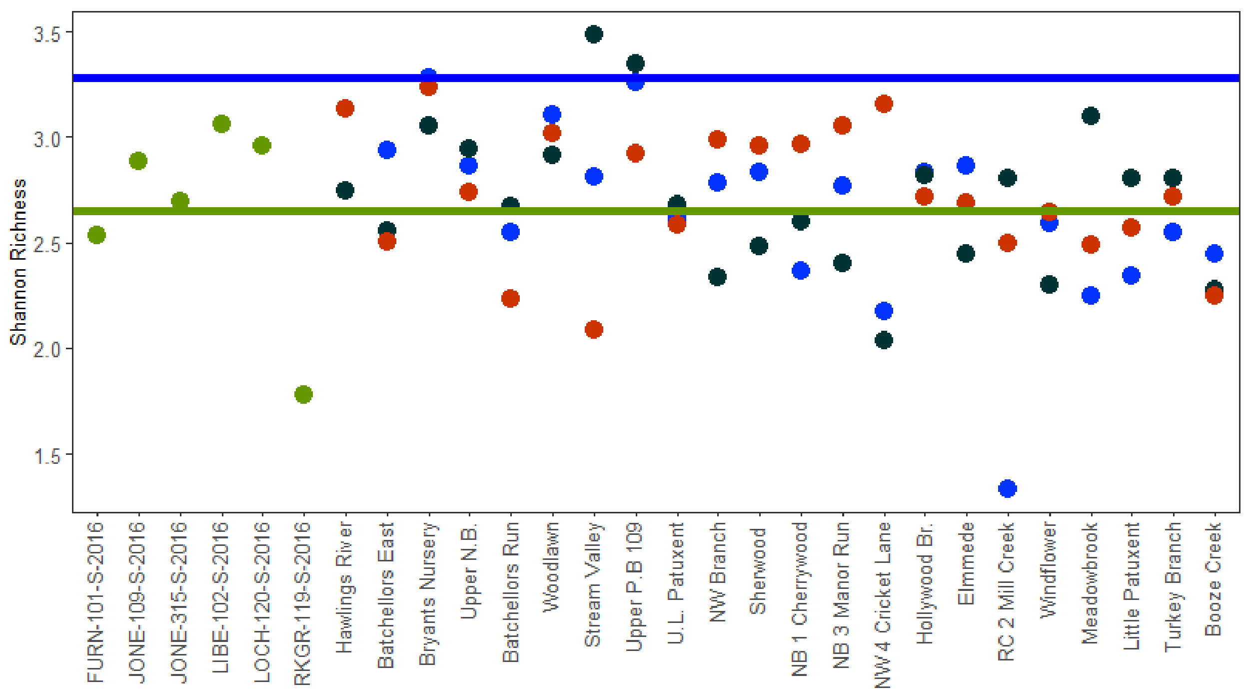


Appendix Figure E1. Intolerant Richness (top panel) and Urban-Tolerant Richness (bottom panel) for each section and ordered across the ISC gradient for PIEDMONT streams. Green line shows the average MBSS Sentinel Site value, while the blue line shows the Potential Ecological Uplift that can be expected from restorations.

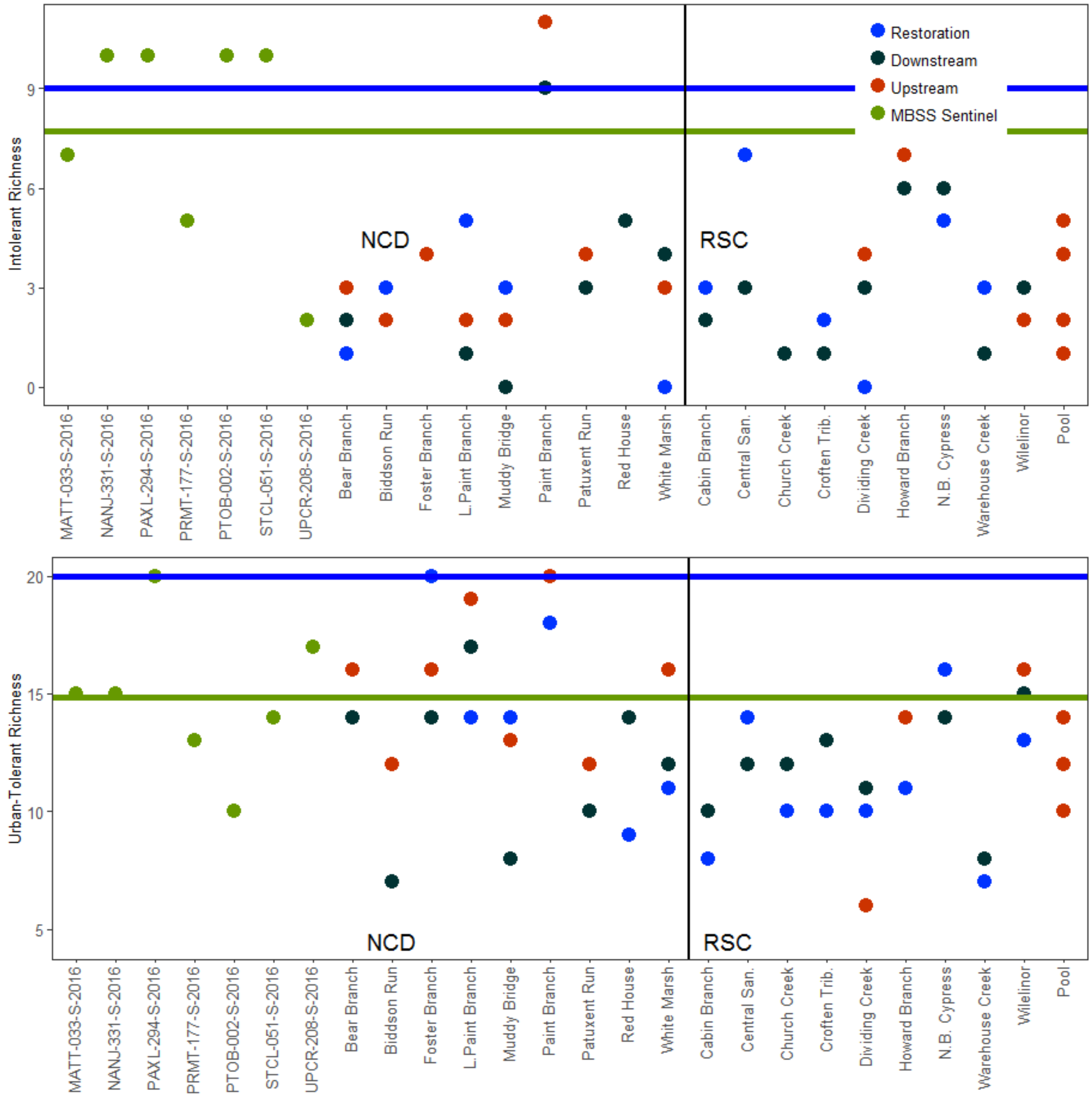


Appendix Figure E2. EPT Richness (top panel) and Shannon-Wiener Richness (bottom panel) for each section and ordered across the ISC gradient for PIEDMONT streams. Green line shows the average MBSS Sentinel Site value, while the blue line shows the Potential Ecological Uplift that can be expected from restorations.

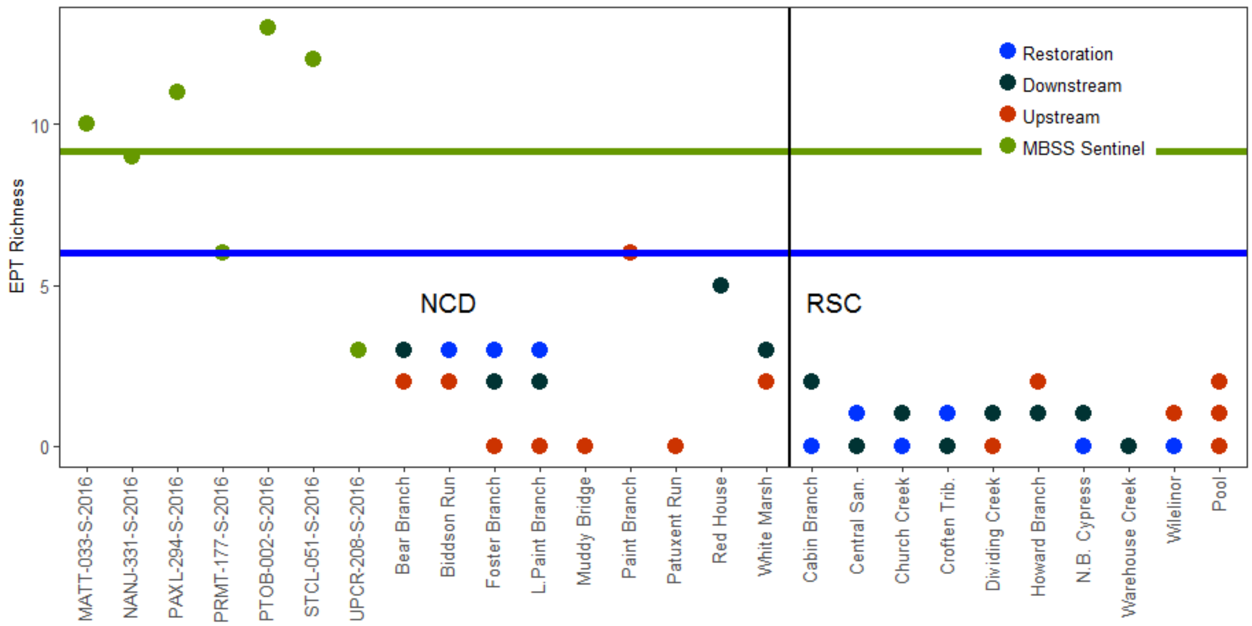


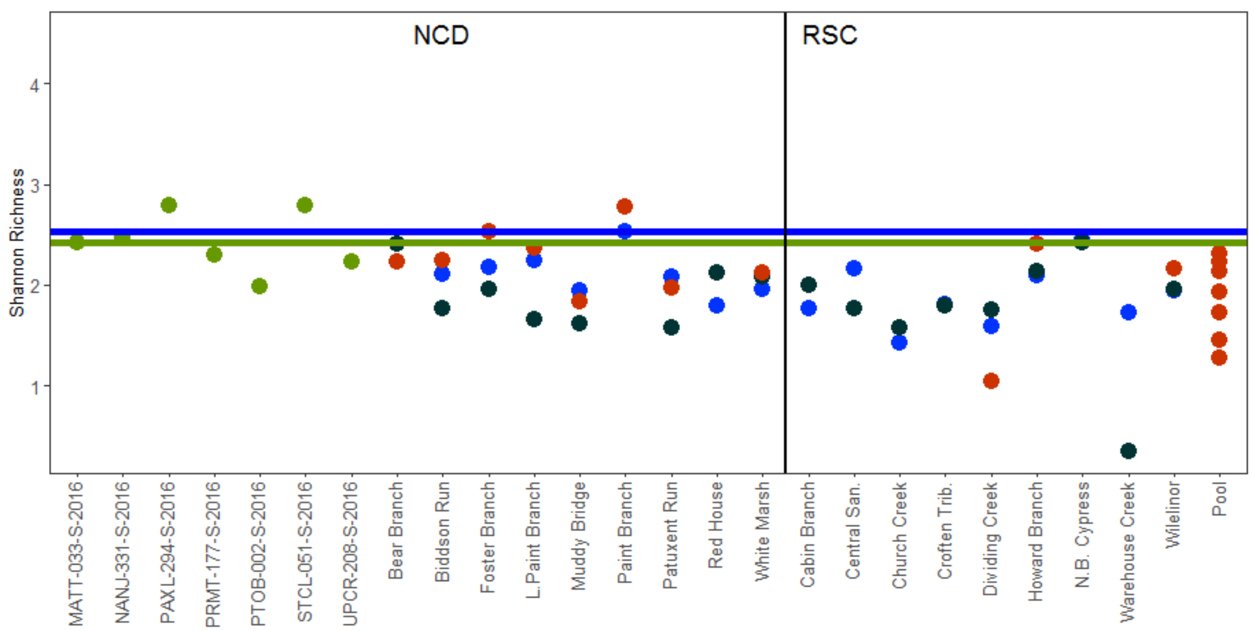


Appendix Figure E3. Intolerant Richness (top panel) and Urban-Tolerant Richness (bottom panel) for each section and ordered by Sentinel, NCD, and RSC groups respectively, for COASTAL PLAIN streams. Green line shows the average MBSS Sentinel Site value, while the blue line shows the Potential Ecological Uplift that can be expected from restorations.

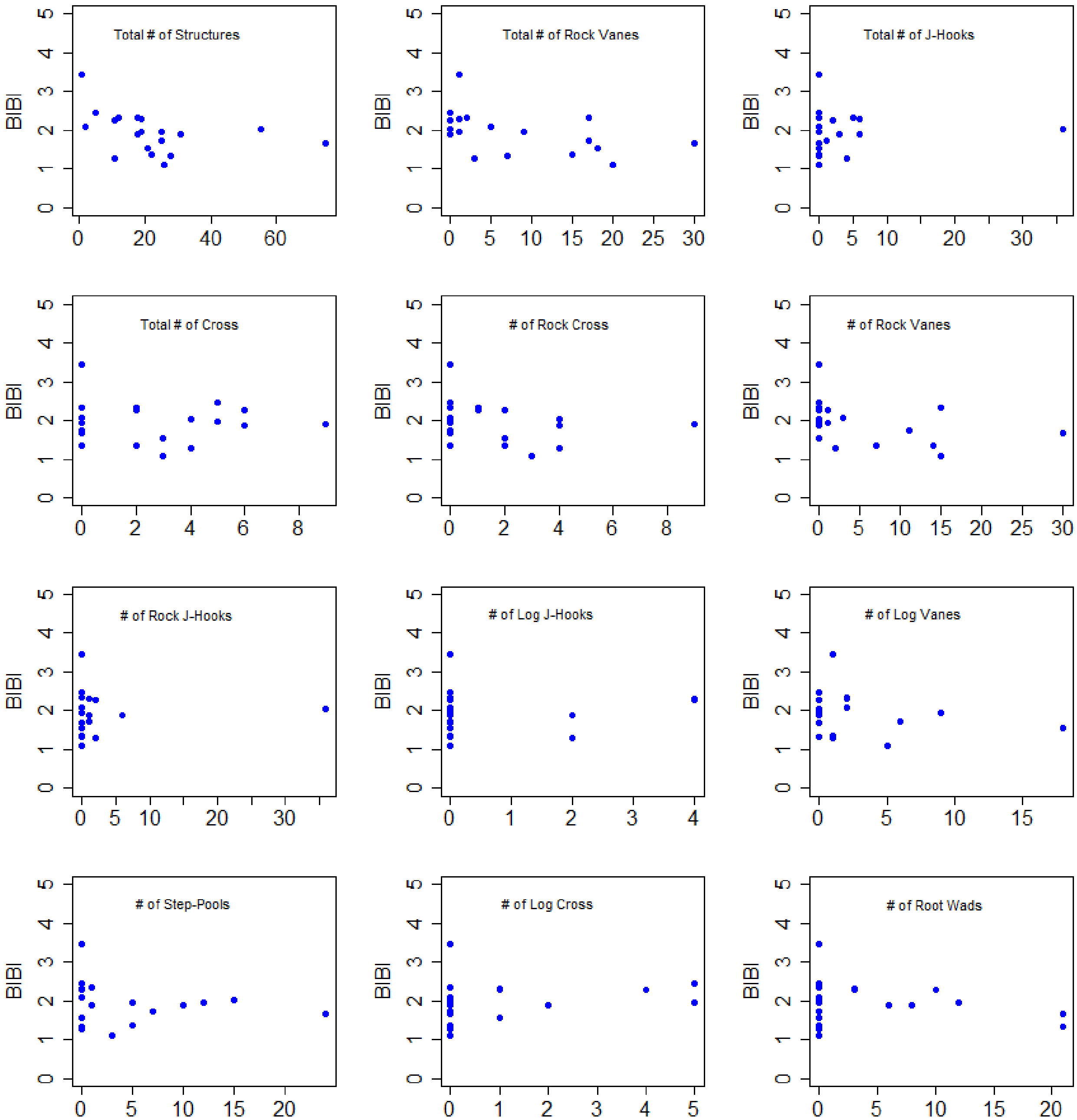


Appendix Figure E4. EPT Richness (top panel) and Shannon-Wiener Richness (bottom panel) for each section and ordered by Sentinel, NCD, and RSC groups respectively, for COASTAL PLAIN streams. Green line shows the average MBSS Sentinel Site value, while the blue line shows the Potential Ecological Uplift that can be expected from restorations.

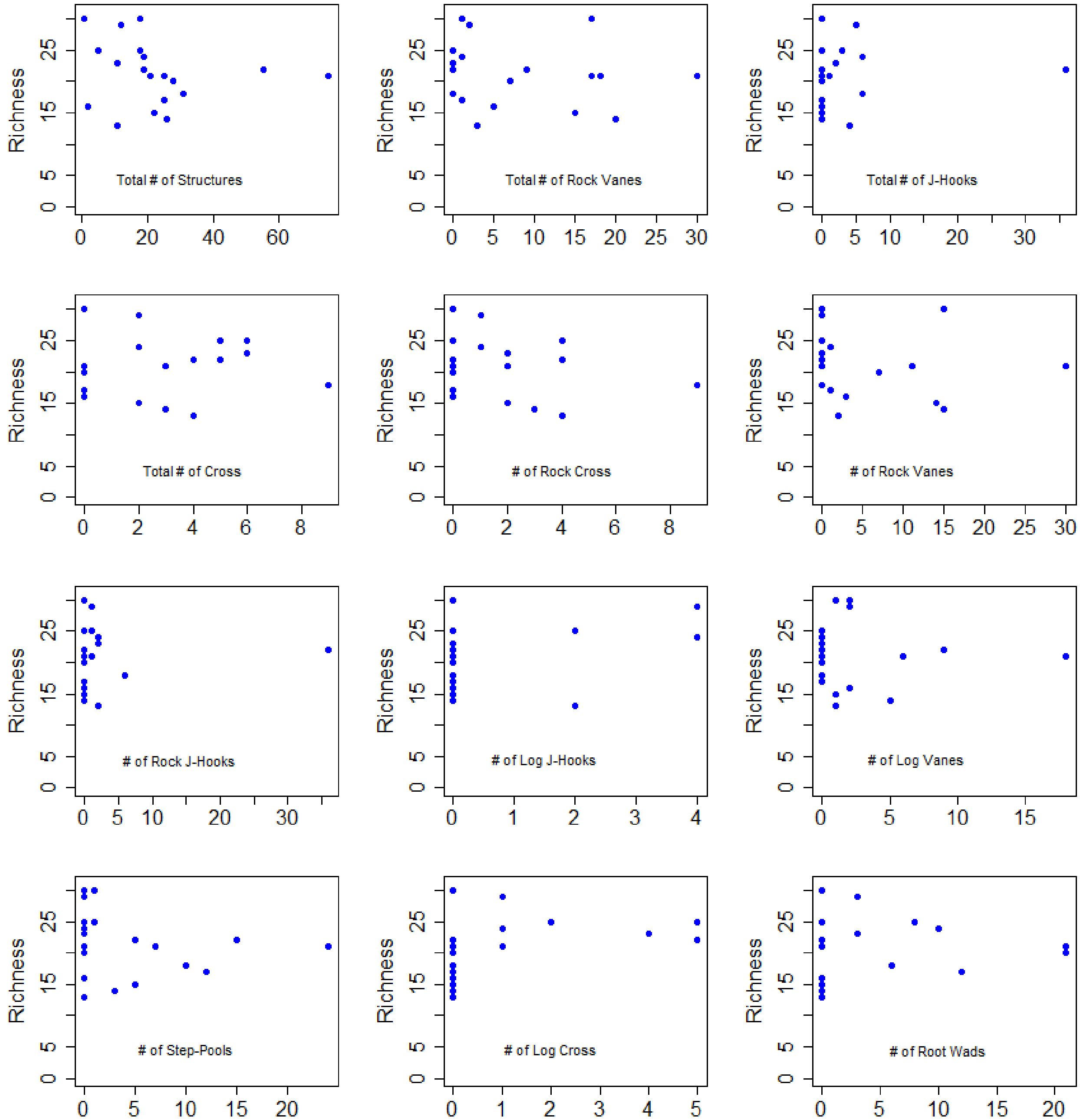




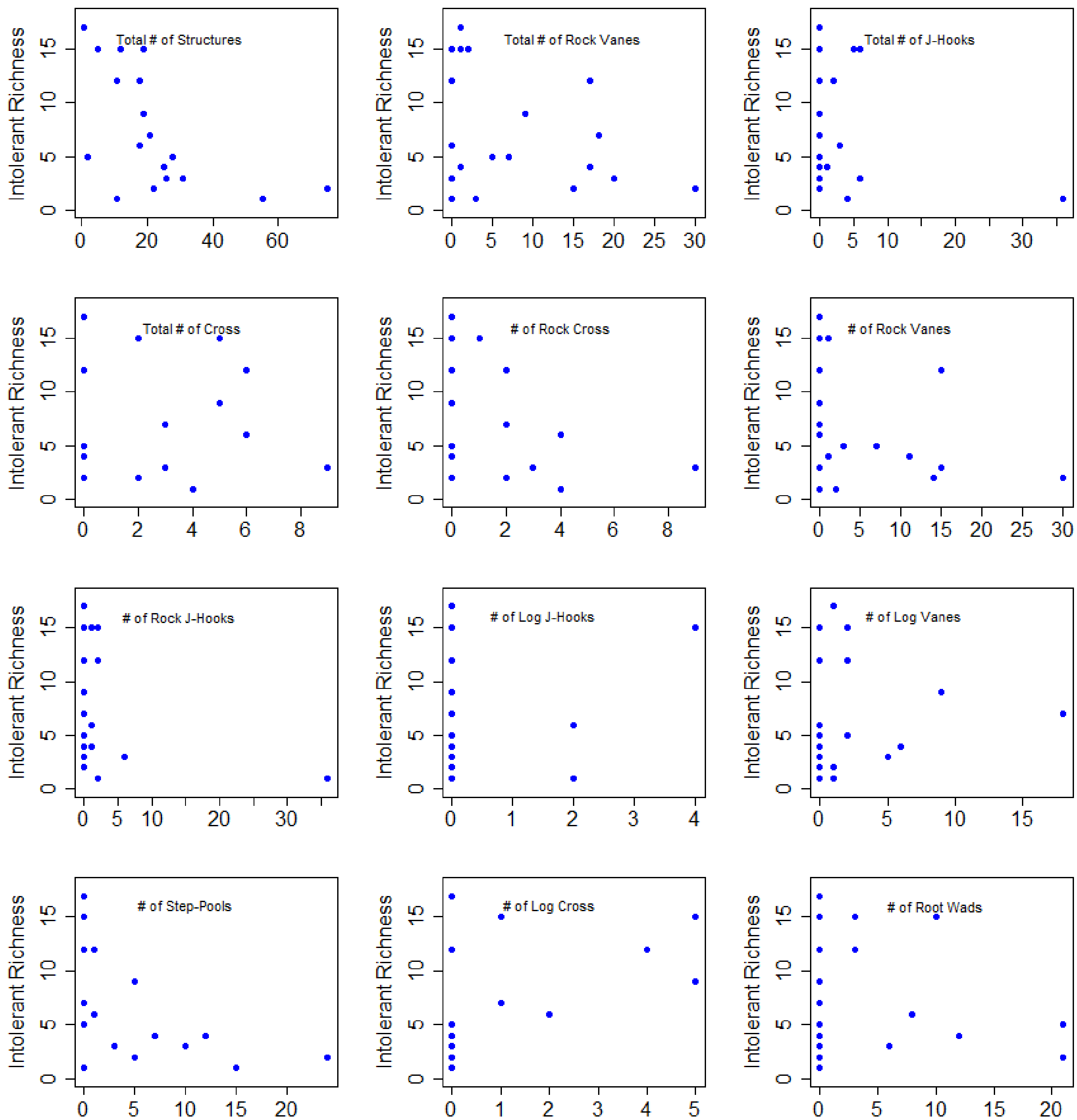
Appendix Figure F1. Relationships between BIBI and installed structures in PIEDMONT restorations.



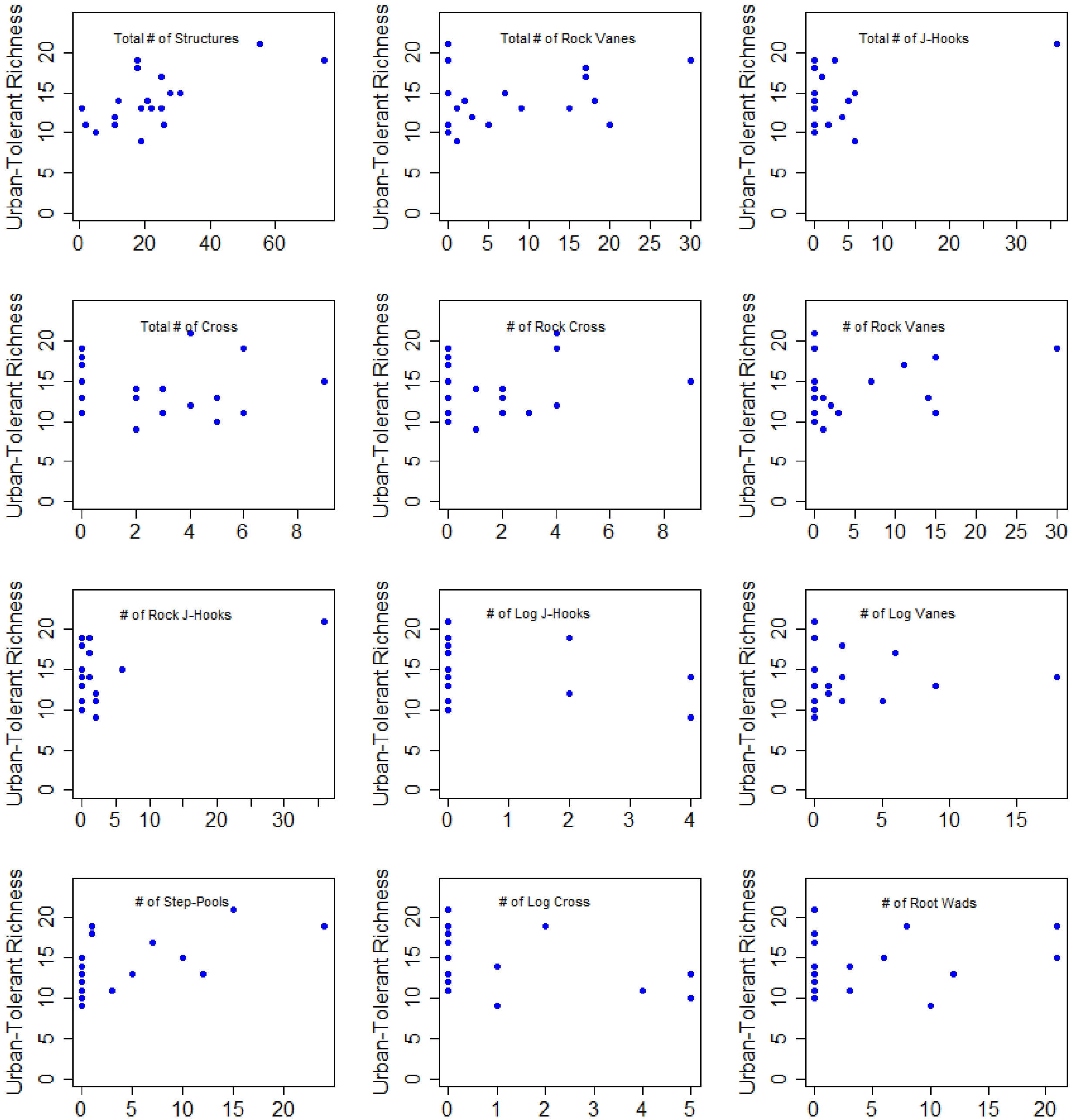
Appendix Figure F2. Relationships between Total Richness and installed structures in PIEDMONT restorations.



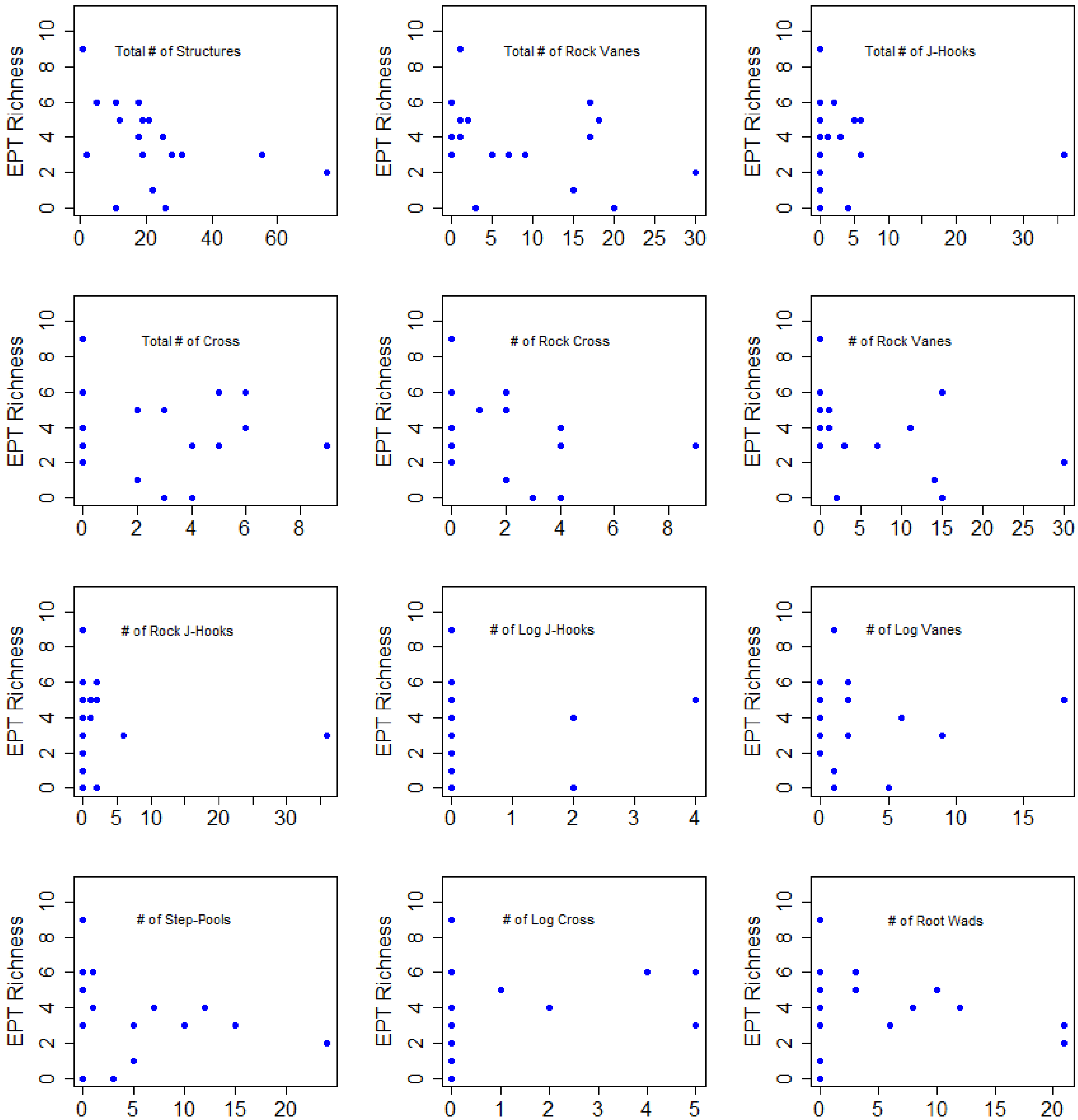
Appendix Figure F3. Relationships between Intolerant Richness and installed structures in PIEDMONT restorations.



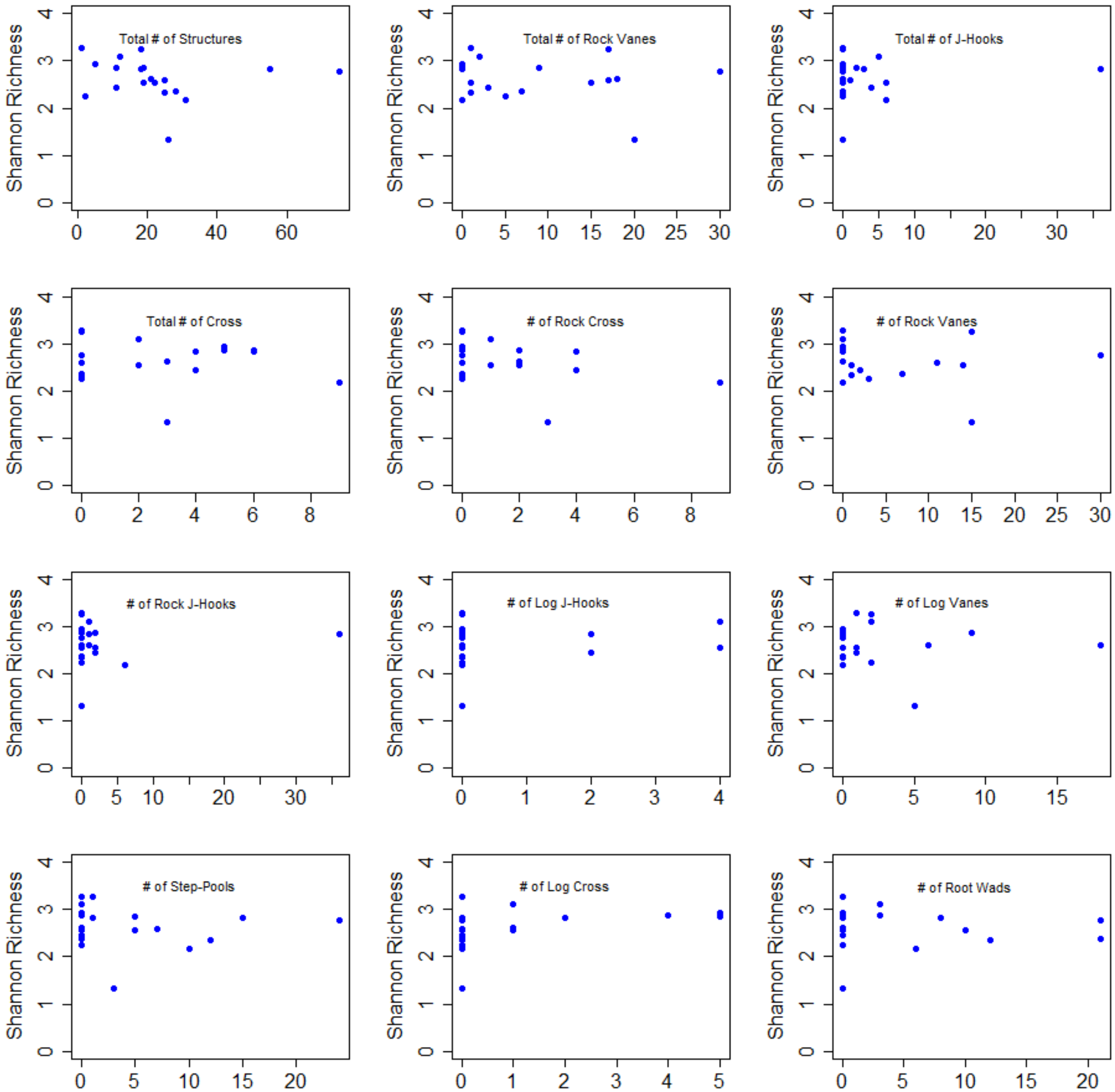
Appendix Figure F4. Relationships between Urban-Tolerant Richness and installed structures in PIEDMONT restorations.



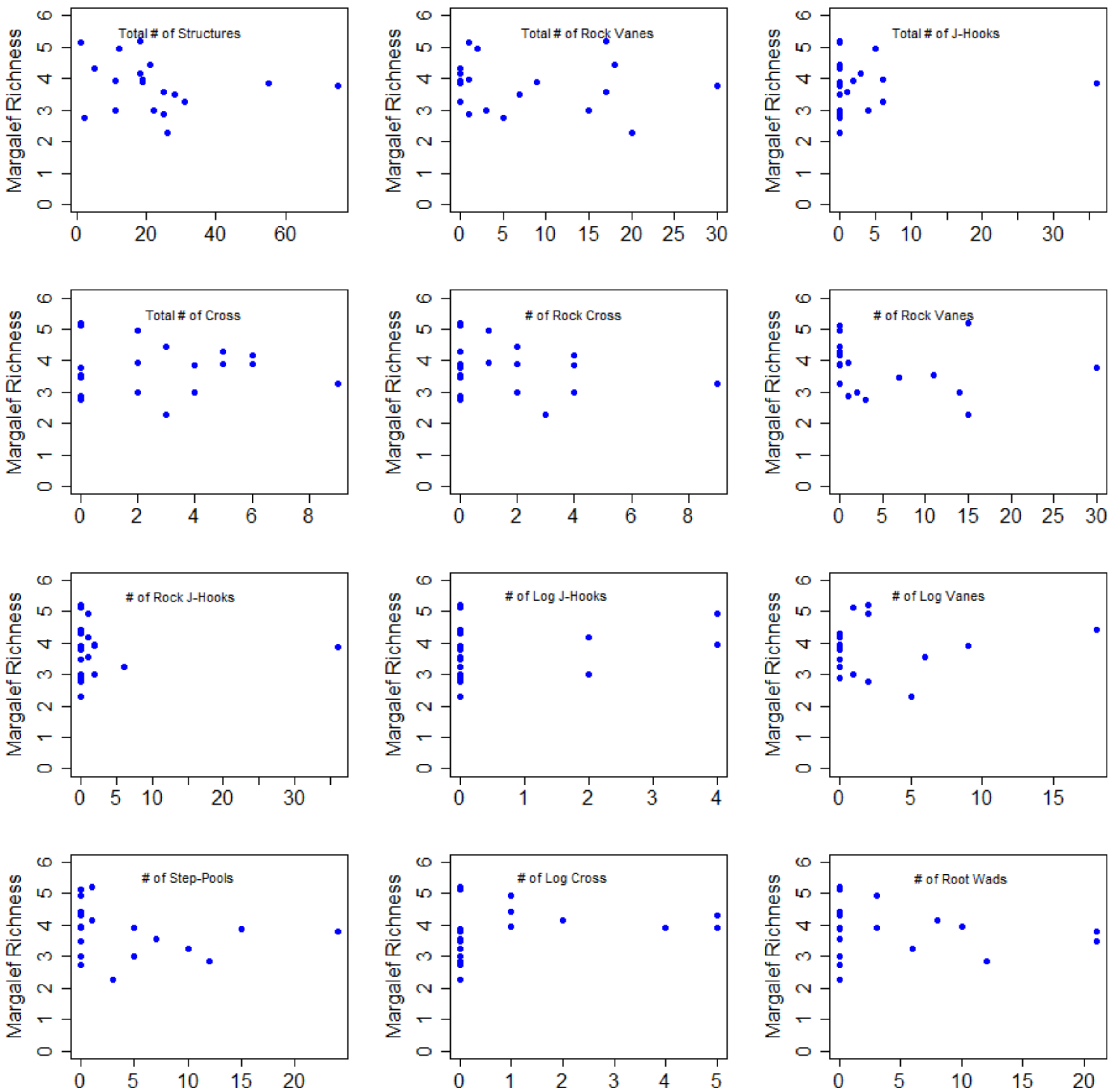
Appendix Figure F5. Relationships between EPT Richness and installed structures in PIEDMONT restorations.



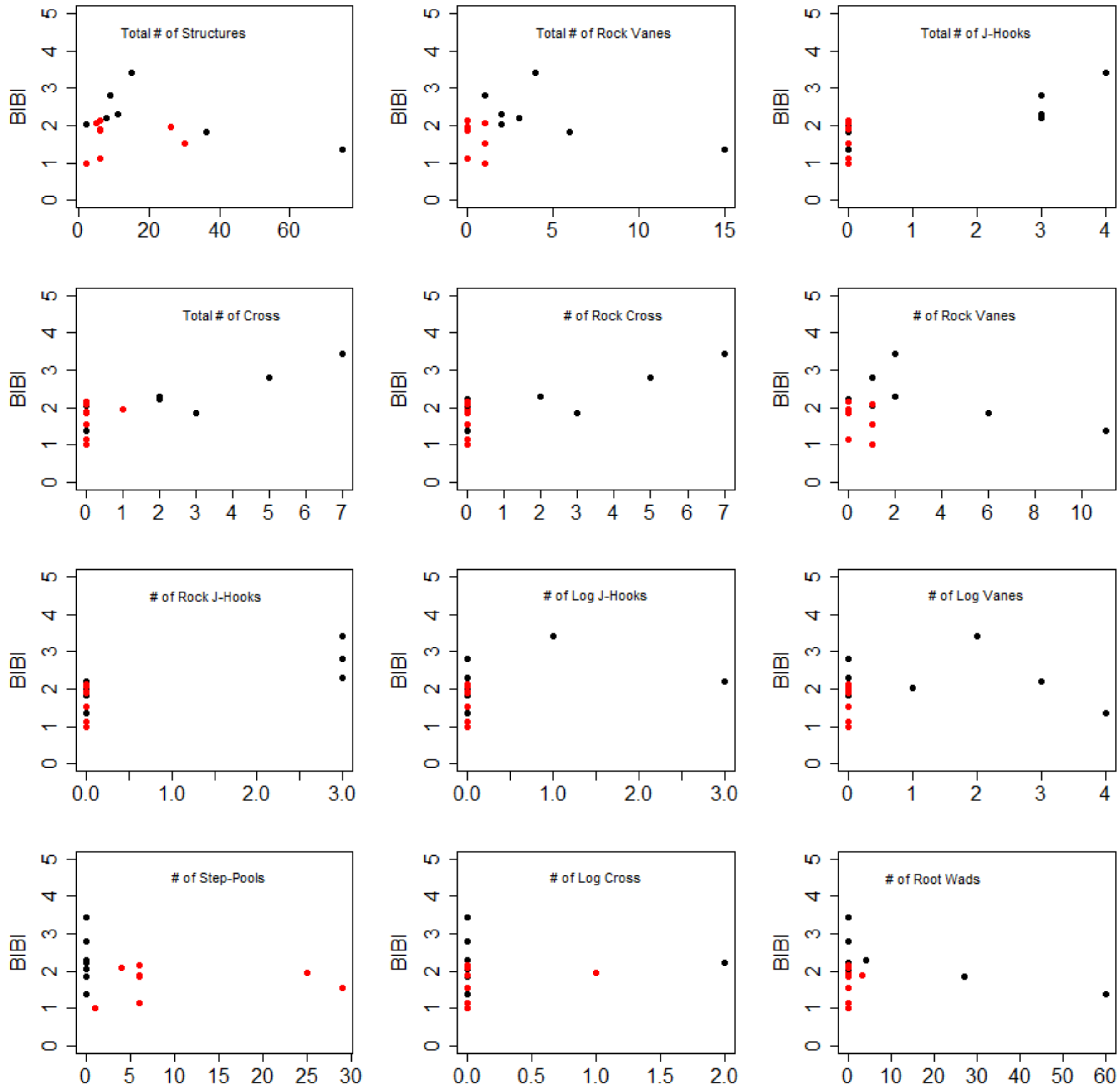
Appendix Figure F6. Relationships between Shannon-Wiener Richness and installed structures in PIEDMONT restorations.



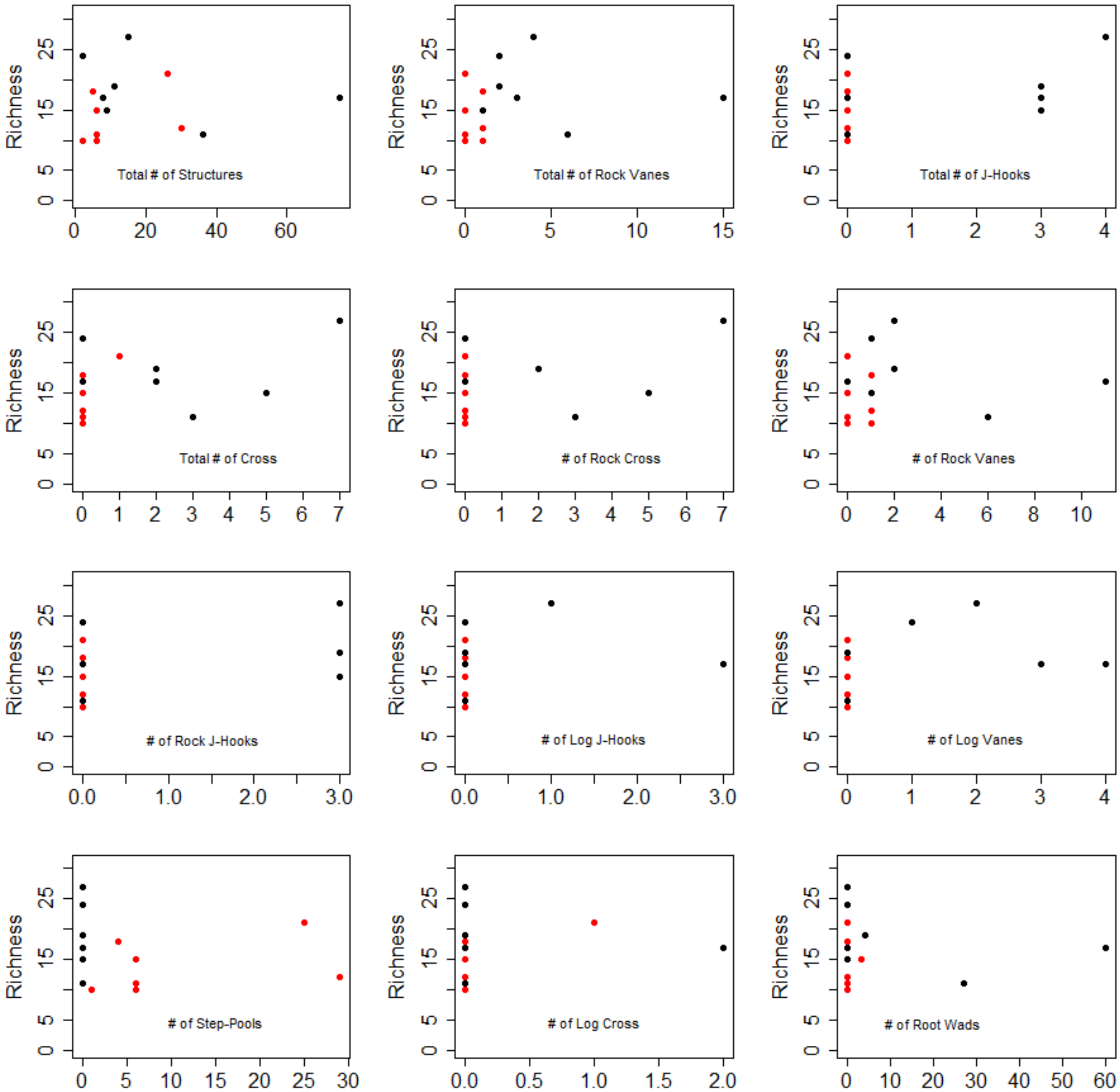
Appendix Figure F7. Relationships between Margalef Richness and installed structures in PIEDMONT restorations.



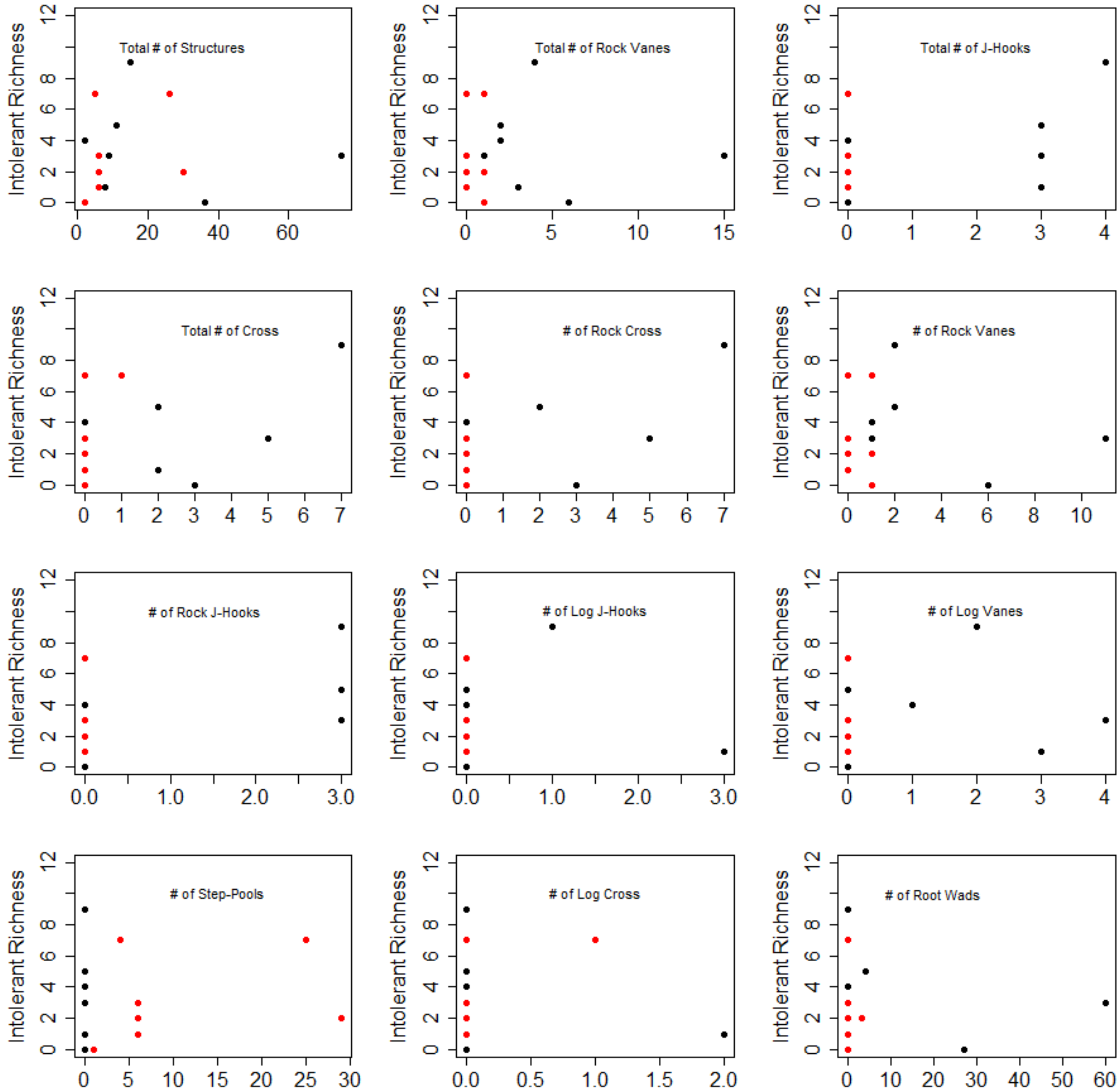
Appendix Figure F8. Relationships between BIBI and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.



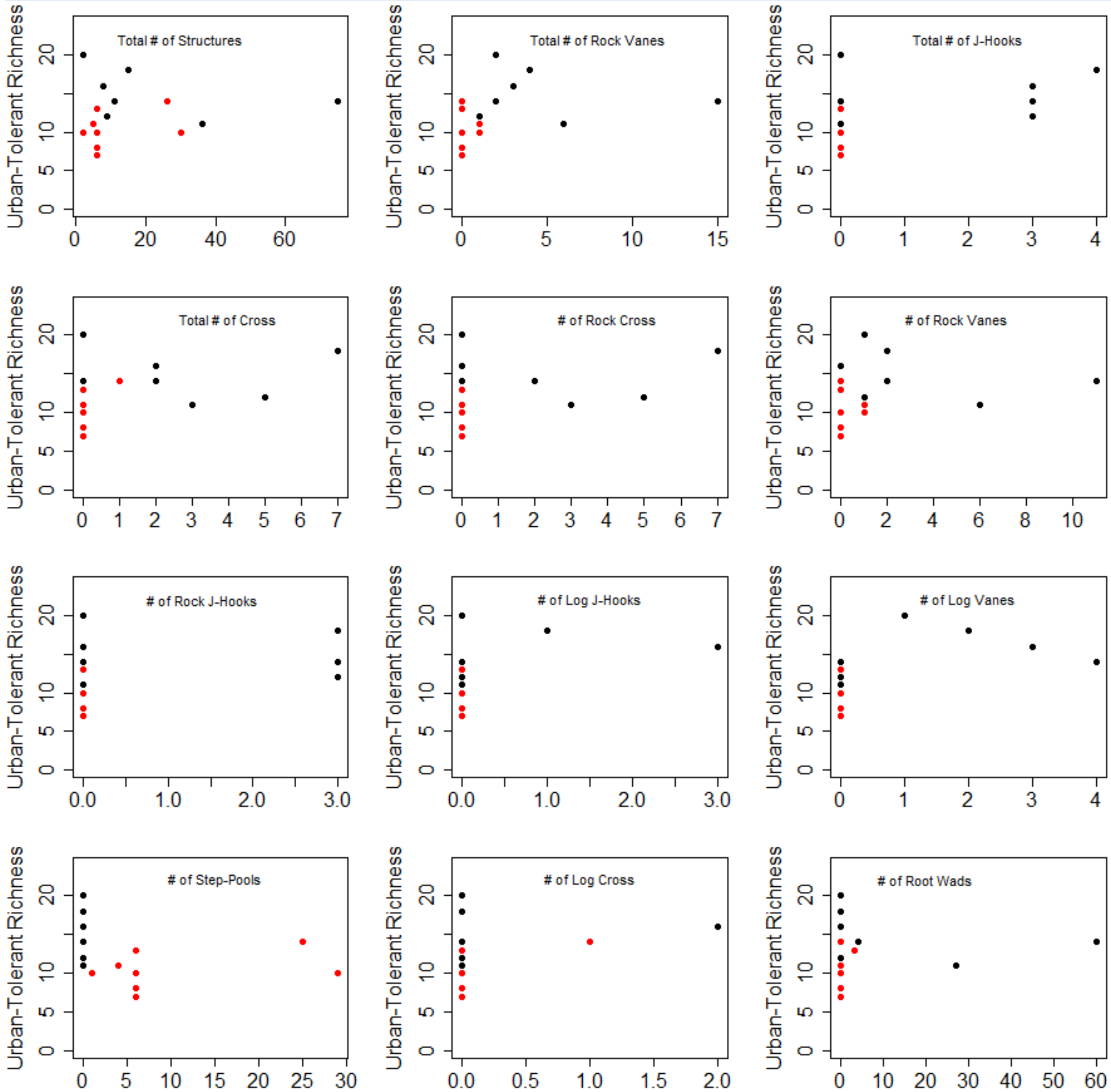
Appendix Figure F9. Relationships between Total Richness and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.



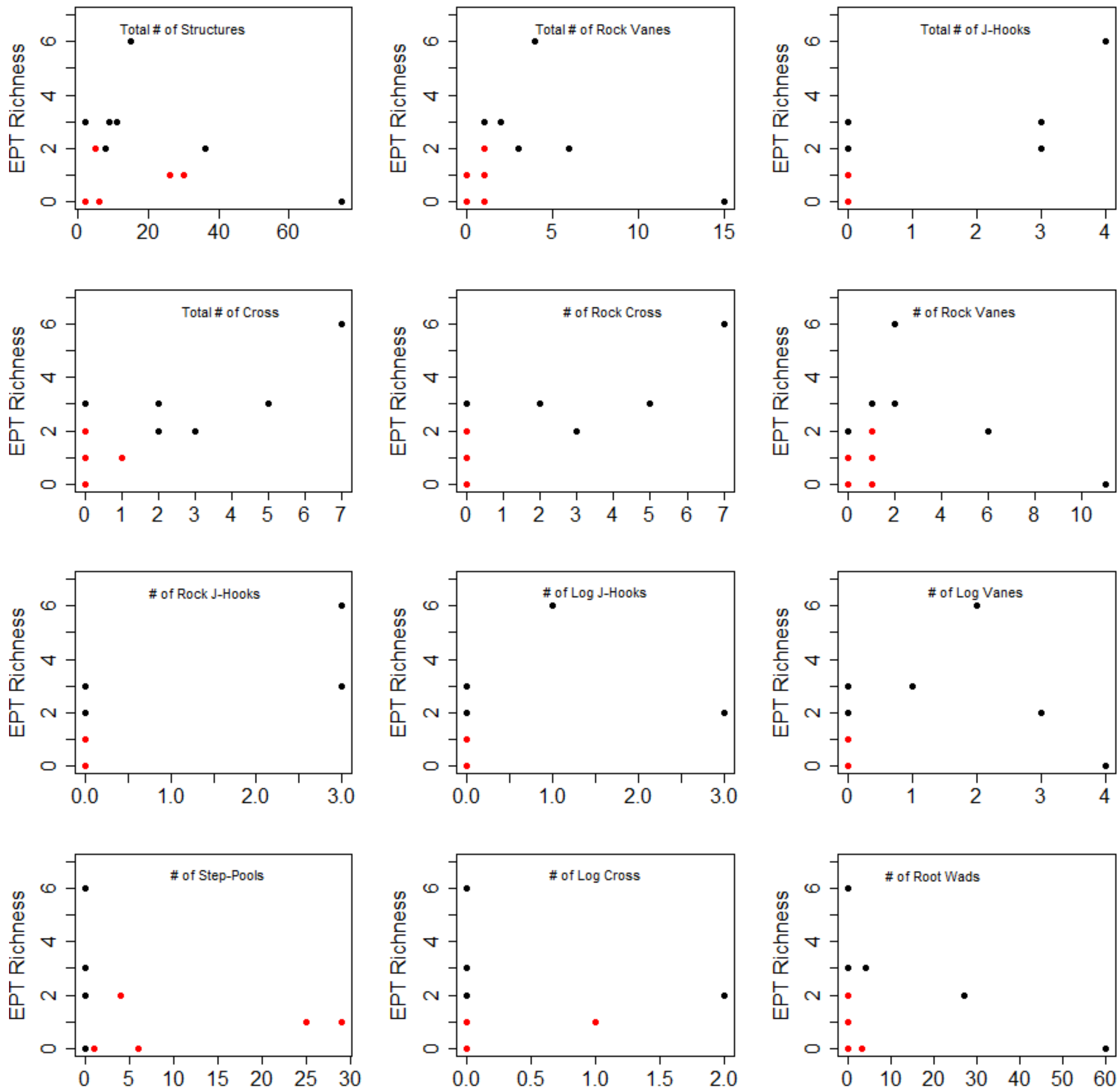
Appendix Figure F10. Relationships between Intolerant Richness and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.



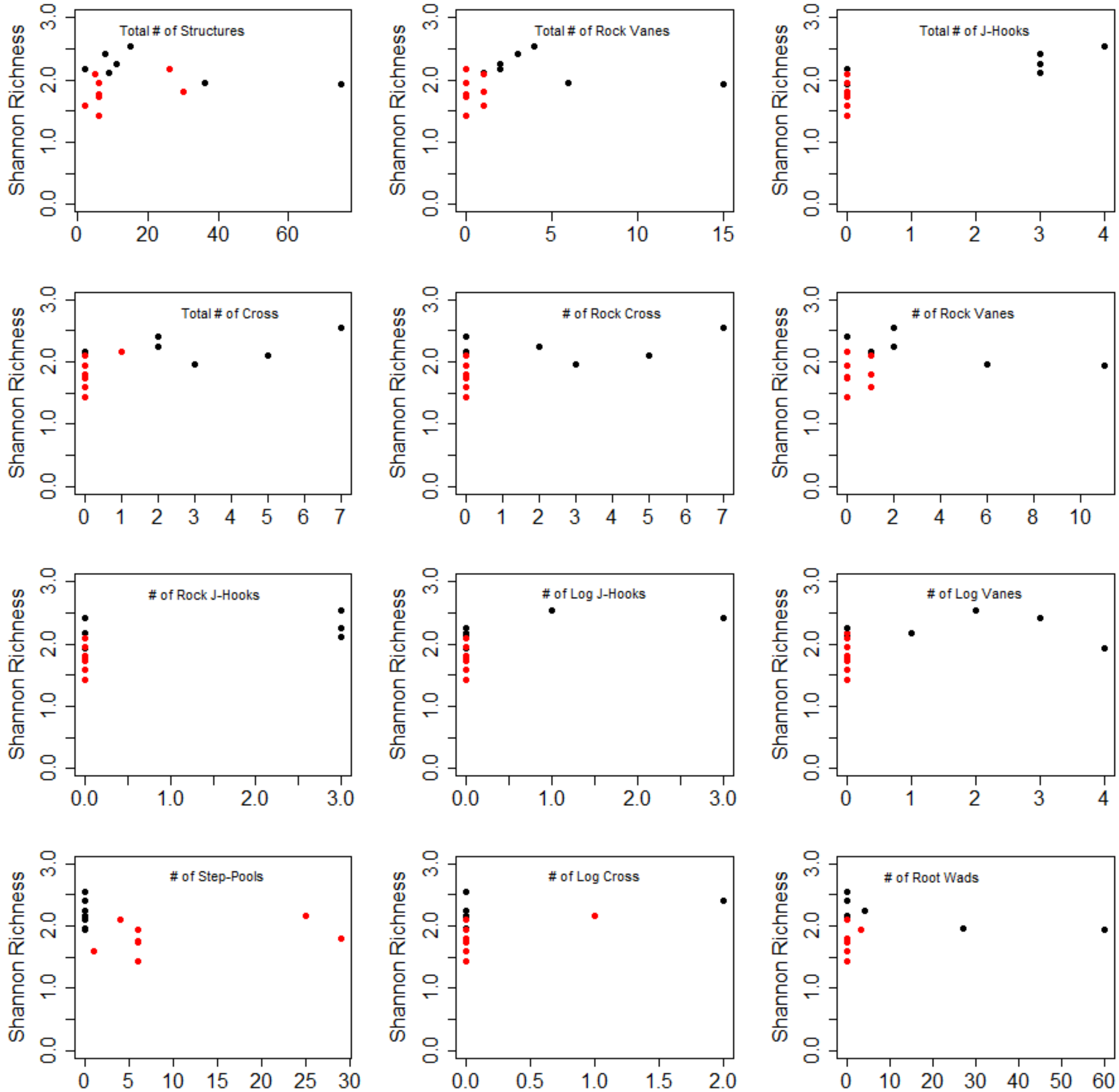
Appendix Figure F11. Relationships between Urban-Tolerant Richness and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.



Appendix Figure F12. Relationships between EPT Richness and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.



Appendix Figure F13. Relationships between Shannon-Wiener Richness and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.



Appendix Figure F14. Relationships between Margalef Richness and installed structures in COASTAL PLAIN restorations. Red points show RSC restorations and black show NCD.