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Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems

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Abstract. Streams, as low-lying points in the landscape, are strongly influenced by the stormwaters, pollutants, and warming that characterize catchment urbanization. River restoration projects are an increasingly popular method for mitigating urban insults. Despite the growing frequency and high expense of urban stream restoration projects, very few projects have been evaluated to determine whether they can successfully enhance habitat structure or support the stream biota characteristic of reference sites. We compared the physical and biological structure of four urban degraded, four urban restored, and four forested streams in the Piedmont region of North Carolina to quantify the ability of reach-scale stream restoration to restore physical and biological structure to urban streams and to examine the assumption that providing habitat is sufficient for biological recovery. To be successful at mitigating urban impacts, the habitat structure and biological communities found in restored streams should be more similar to forested reference sites than to their urban degraded counterparts. For every measured reach- and patch-scale attribute, we found that restored streams were indistinguishable from their degraded urban stream counterparts. Forested streams were shallower, had greater habitat complexity and median sediment size, and contained less-tolerant communities with higher sensitive taxa richness than streams in either urban category. Because heavy machinery is used to regrade and reconfigure restored channels, restored streams had less canopy cover than either forested or urban streams. Channel habitat complexity and watershed impervious surface cover (ISC) were the best predictors of sensitive taxa richness and biotic index at the reach and catchment scale, respectively. Macroinvertebrate communities in restored channels were compositionally similar to the communities in urban degraded channels, and both were dissimilar to communities in forested streams. The macroinvertebrate communities of both restored and urban degraded streams were correlated with environmental variables characteristic of degraded urban systems. Our study suggests that reach-scale restoration is not successfully mitigating for the factors causing physical and biological degradation.

Key words: *benthic macroinvertebrate; biotic recovery; habitat restoration; species composition; stream restoration; urbanization.*

Introduction

The world's human population is primarily urban, and future population growth will occur predominantly in urban centers (United Nations 2008). Thus, an increasing proportion of our freshwater ecosystems will become impacted by urbanization, and a larger fraction of humanity will rely on waterways degraded by a common set of urban impacts. The physical, biogeochemical, and biological stream impairments that occur specifically in urbanized watersheds have been labeled the "urban stream syndrome" (Walsh et al. 2005*b*). Physical and hydrological consequences of watershed urbanization are well documented and include altered base flow and unstable hydrology with frequent, short-duration, high-peak floods (Booth and Jackson 1997, Paul and Meyer 2001, Meyer et al. 2005, Walsh et al. 2005*b*). These changes typically lead to channel incision and simplification (Shields et al. 2003, Niezgoda and Johnson 2005, Sudduth and Meyer 2006), and homogenization of benthic habitats (Federal Interagency Stream Restoration Working Group 1998, Malmqvist and Rundle 2002, Walsh et al. 2005*b*).

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Coincident with hydrological and geomorphological modification, urban streams have elevated nutrient and contaminant concentrations. Hyperconnectivity with the surrounding landscape through roads, storm drains, and leaky and overflowing sanitary sewers efficiently routes watershed contaminants into urban channels (Bernhardt et al. 2008). Pollutant concentrations increase not only due to increased inputs from point and non-point sources but also as a result of decreased nutrient removal efficiency in hydrologically disconnected riparian zones and streambeds (Groffman and Crawford 2003, Grimm et al. 2005, Meyer et al. 2005).

The inverse relationship between urbanization and native biodiversity and species composition is well established and persists across a range of taxa (Blair 1996, Germaine and Wakeling 2001, Clark et al. 2007, Grimm et al. 2008, McKinney 2008). Macroinvertebrate communities are strongly affected by land use patterns (Lenat and Crawford 1994, Sponseller et al. 2001, Allan 2004). Watershed impervious surface cover is generally associated with a decrease in invertebrate species richness and increasing dominance of highly tolerant taxa (Morse et al. 2003, Roy et al. 2003, Moore and Palmer 2005, Collier et al. 2009, Cuffney et al. 2010). Development that is within riparian areas or that is directly hydrologically connected to stream channels (e.g., road crossings and pipes) can be particularly detrimental to stream communities (Wang and Kanehl 2003, Moore and Palmer 2005, Walsh and Kunapo 2009), and there is thus great interest in riparian reforestation and management for urban stream ecosystem protection (Bernhardt and Palmer 2007). While the impacts of watershed urbanization on stream biota are well documented, it is far from clear what combination of reach and watershed scale management is necessary and sufficient to promote community recovery in urban streams.

In the face of channel incision and bank erosion, water quality degradation, and habitat and biodiversity loss, degraded urban waterways are often targeted for restoration. Stream restoration or rehabilitation encompasses a variety of strategies by which human impacts are mitigated and previous damage is addressed, with the overarching goal of returning the stream to as close to pre-impacted conditions as possible (National Research Council 1992). Urban stream restoration presents unique problems: there is minimal space for rehabilitation, and land acquisition is both expensive and complicated because it generally involves multiple landowners. These challenges typically lead to fewer linear feet being restored and higher per-project costs compared to rural and agricultural stream restoration projects (Bernhardt and Palmer 2007). In fact, for many regions of the United States, the majority of restoration dollars are invested in a small number of urban stream projects (Hassett et al. 2005, Sudduth et al. 2007). Stream restoration projects are customarily implemented with the specific goals of water quality improvement and provision of aquatic habitat (Bernhardt et al. 2007, Sudduth et al. 2007), yet few projects have been adequately evaluated to determine whether these goals are met (Charbonneau and Resh 1992, Palmer et al. 1997, 2005, Moerke et al. 2004, Moerke and Lamberti 2004, Bernhardt et al. 2005). Given the frequency with which urban stream restoration is employed to mitigate habitat and water quality degradation and the expenses and challenges involved, it is worth understanding whether these efforts are measurably improving habitat and community structure.

The underlying assumption of stream restoration is that altering channel geomorphology to resemble pre-degradation conditions will lead to the recovery of native aquatic organisms. This assumption is based on prior work demonstrating that fish or macroinvertebrate taxonomic richness and spatial heterogeneity are positively correlated (Gorman and Karr 1978, Angermeier and Winston 1998, Vinson and Hawkins 1998, Brown 2003). Although experimental manipulations have demonstrated that high substrate variability does not per se lead to higher species richness or faster recovery (Brooks et al. 2002, Spanhoff et al. 2006), stream restoration design employs habitat provision, or increased habitat heterogeneity as the primary mechanism for restoring biotic communities (Brooks et al. 2002). Evidence to support the assumption that successfully restoring physical structure is sufficient for community restoration (the "field of dreams" hypothesis) is lacking (Palmer et al. 1997, 2010, Moerke et al. 2004).

We set out to evaluate the effectiveness of four natural channel design (NCD) projects, a common urban stream restoration approach (sensu Rosgen 1994, 1996) in mitigating urban stream degradation. NCD reconfigures the pattern, profile, and dimensions of a degraded channel to emulate an unimpacted ideal (Rosgen 2007). This method utilizes heavy machinery to regrade and reshape a degraded channel and employs hard structures such as log vanes or cross vanes to control grade, installs root wads to stabilize banks, adds coarse bed material to create riffles, and revegetates reconfigured or newly created riparian areas.

Effective restoration should recapture the habitat structure and biological communities of forested streams, ideally approaching a stable approximation of "reference" conditions. We tested whether a series of urban restoration projects were achieving or moving toward this goal by examining whether habitat structure and macroinvertebrate community composition in the restored reaches of urban streams were different from similarly situated urban degraded stream reaches and whether the habitat and community structure of these restored reaches more closely matched conditions in nearby forested streams than their unrestored urban counterparts.

Methods

Site selection

Through consultation with staff of the North Carolina Ecosystem Enhancement Program (EEP) and the North Carolina Stream Restoration Institute (SRI) we selected four urban natural channel design restoration projects that practitioners and regulators felt were particularly well-designed and implemented. Our goal in selecting restoration projects was not to select a random sample, but rather to choose a set of projects that represented the best-case scenario for urban restoration based on expert practitioners' opinions. Each restored stream reach was then matched with a similarly situated unrestored urban stream and a forested stream in the Raleigh-Durham area in the Piedmont region of North Carolina. The full comparison thus included 12 study sites: four forested (F) sites, within small streams draining forested catchments; four urban restored (R) sites, within recently implemented natural channel design restoration projects; and four urban degraded (U) sites located in urban parks where future restoration activities are likely (Fig. 1; Appendix A). This suite of sites was selected to determine the potential for ecological restoration to restore the physical and biological structure and ecosystem function of stream ecosystems.

Site descriptions

Four sampling blocks were created from the group of 12, each containing one urban degraded, one urban restored, and one forested stream of similar catchment sizes and underlying geology (Table 1). The study area spans the Northern Outer Piedmont, Slate Belt, and Triassic basin ecoregions, and many sites drain multiple ecoregions (Table 1). Soil characteristics affect baseflows and consequently stream size and permanence. Triassic Basin and Slate Belt streams have low summer baseflows due to low clay permeability and low water yield from slate substrate (Griffith et al. 2002). Reduced summer baseflows are not seen in Northern Outer Piedmont streams where streams tend to be larger and less prone to drying. For physical and functional metrics, all streams within a sampling block were sampled within one week with no intervening major storm events. In this way, the blocking factor accounts for both differences in watershed size, and staged timing of field analyses.

Our study included four restored stream reaches, each of which was restored using NCD between 1999 and 2005. The Abbott stream restoration project was implemented in 1999 on a tributary to Walnut Creek, in Raleigh, North Carolina. The goal of this restoration project was "to restore the stream to the stable dimension, pattern, and profile for a C4 stream type as classified using Rosgen's stream classification methodology (Rosgen 1996). ... This type of restoration will reestablish the channel on a previous floodplain, or in this case, the basin of an old pond. Appropriate channel dimensions (width and depth), pattern (sinuosity, belt width, riffle–pool spacing), and profile (bed slope) of the new channel will be determined from reference reaches" (North Carolina Department of Transportation 1999). Rocky Branch is a stream located on the urban North Carolina State University campus in Raleigh and was restored in 2001. The goals of this restoration project included, "Restore a stable self-maintaining morphological pattern in the stream channel; Stabilize stream banks using vegetation; Create and improve habitat for fish and aquatic invertebrates; Improve the quality of stormwater entering the creek through restoring and enhancing riparian buffers and establishing stormwater control within the creek's watershed; Provide safe and enjoyable access to the stream and passage through the campus by completing the greenway path adjacent to the creek." (Doll 2003). Restored in 2004, Sandy Creek flows through the urban Duke University campus in Durham. The Sandy Creek project goals were to, "Recontour and restore more than 600 meters of degraded stream to hydrologically reconnect the stream with the adjacent floodplain to improve biogeochemical transformations and stream water quality" (Richardson and Pahl 2005). Third Fork Creek is a stream flowing through an urban park near downtown Durham and was restored in 2005. The goals of this project were to, "Restore stable channel morphology that is capable of moving the flows and sediment provided by its watershed; reduce sediment-related poor water quality impacts resulting from lateral bank erosion and bed degradation; improve aquatic habitat diversity through the reestablishment of riffle–pool bed variability and the use of in-stream structures; restore vegetative riparian buffers utilizing native plant species; and improve natural aesthetics in an urban park setting." (KCI Associates 2003).

All of our urban stream reaches were located in urban parks or protected areas to facilitate access, and are similar to the pre-restoration conditions of our restored study sites. Two of our urban stream sites (reaches of Goose Creek and Ellerbe Creek) were chosen because the North Carolina Ecosystem Enhancement Program listed them as priority stream restoration sites (both were restored after this research effort). Our study reach on Upper Mud Creek is located within the protected Duke Forest, immediately downstream of a 1980s-era subdivision. Cemetery Creek is located on city property in Raleigh and drains an older, high-density, urban neighborhood.

Forested sites were selected from "reference" sites previously used for stream restoration projects as well as sites within Duke Forest. Lower Mud Creek and the Tributary to Mud Creek are located in Duke Forest, in Durham. Stony Creek is located in Duke Forest near Hillsborough. Pot's Branch is located in Umstead State Park near Raleigh. Because of the land use history of the North Carolina Piedmont, these are not pristine reference sites, but rather post-agriculture reforested streams with primarily forested watersheds; thus there may be legacy effects of prior agricultural land use on geomorphology, vegetation, sediment, and biota (Maloney et al. 2008). There are no primary growth forests of sufficient size to have a permanent stream,

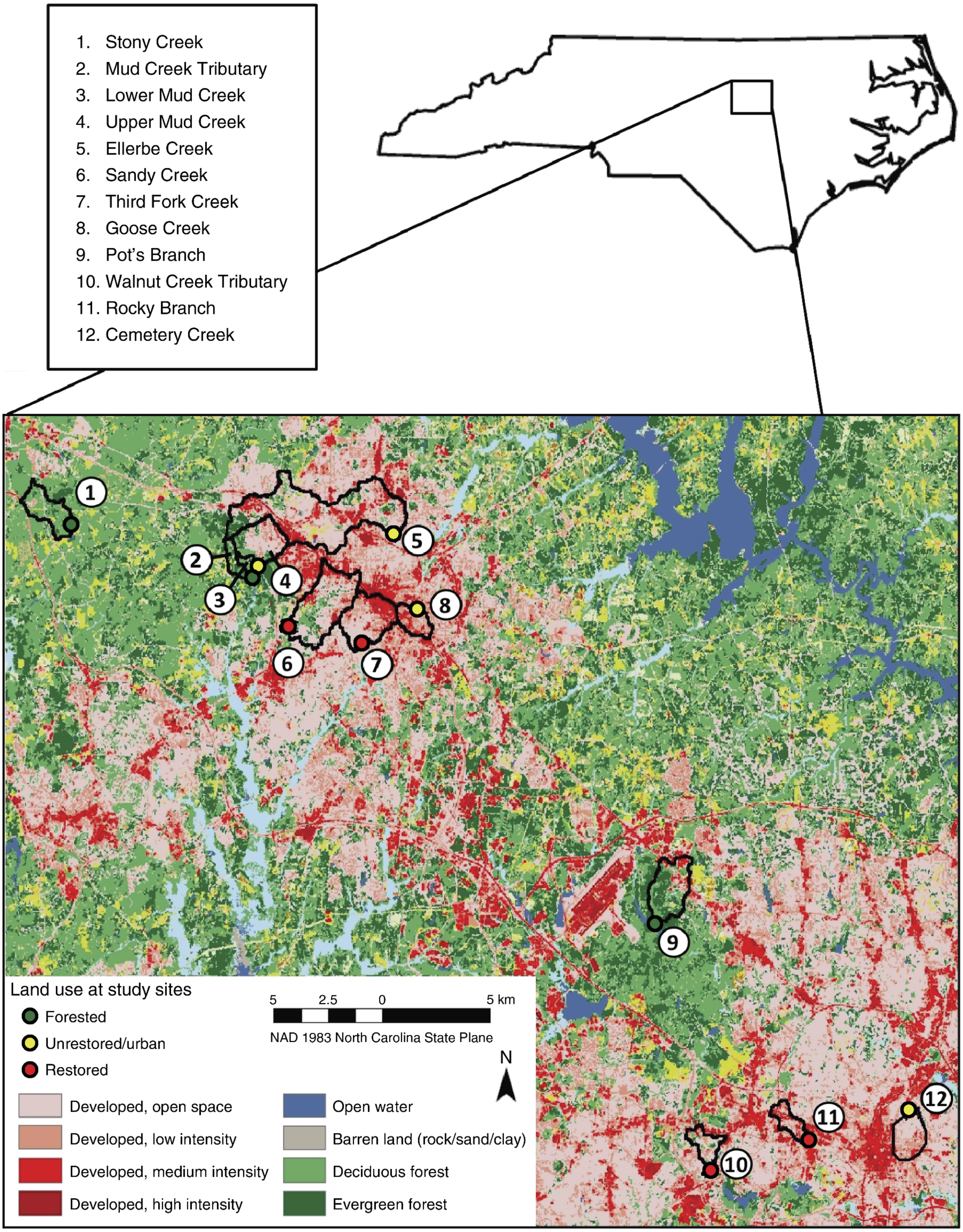

FIG. 1. Study site locations and watershed boundaries in the Raleigh–Durham area in the Piedmont region of North Carolina.

rather our "forested" streams are secondary growth and represent the post-agricultural, pre-urban landscape. Due to the lack of undisturbed Piedmont streams, we included one forested stream reach, Lower Mud Creek that had urban development more than 1.5 km upstream of the study reach, and for which the entire watershed within that 1.5 km was ~100-year-old mixed deciduous forest. Because Lower Mud Creek is far from an ideal reference stream, we performed all statistical comparisons both with and without this stream.

TABLE 1. Study sites listed by block, stream type, EPA level-IV ecoregion, channel, and catchment characteristics.

Status	Site name	Ecoregion	Reach length (m)	Estimated discharge (L/s)	Watershed size (km^2)	Developed (%)	% ISC
Block 1							
Forested	Stony Creek	45c	100	0.66	6.9	24.4	3.4
Urban restored	Third Fork Creek	45g	80	4.41	4.4	99.5	32.4
Urban degraded	Ellerbe Creek	45c/g	50	10.41	7.6	88.7	20.8
Block 2							
Forested	Pot's Branch	45f	140	5.83	4.2	27.4	9.9
Urban restored	Walnut Creek Tributary	45f	200	5.47	1.7	84.5	17.8
Urban degraded	Cemetery Creek	45f	100	11.54	2.2	98	19.1
Block 3							
Forested	Mud Creek Tributary	45c/g	54	2.08	0.9	4.4	0.5
Urban restored	Rocky Branch	45f	50	1.54	1.5	99.2	34.8
Urban degraded	Goose Creek	45g	35	3.72	1.7	100	39.4
Block 4							
Forested	Lower Mud Creek	45c/g	102.5	11.58	4.1	58.6	9.5
Urban restored	Sandy Creek	45g	60	12.00	6.7	76.9	16.8
Urban degraded	Upper Mud Creek	45c/g	140	4.86	3.5	66.9	11

Notes: Ecoregions are described in Griffith et al. (2002); %ISC is the percentage of the watershed with impervious surface cover.

Land use characterization

We acquired the 1/3 arc-second (10 m) digital elevation model for Durham, Orange, and Wake counties in North Carolina from the USGS Seamless Server and performed analysis using the ArcHydro extension of ArcGIS (ESRI, Redlands, California, USA) to calculate flow direction and flow accumulation, and define streams based on a 1000-pixel threshold and delineate watersheds for all sites. Land use and impervious surface cover within study watersheds were analyzed based on 2001 National Land Cover Dataset (NLCD), and the associated Impervious Surface Cover data set from the USGS Seamless Server (Homer et al. 2004). We classified riparian land use in a 30-m buffer around each stream segment using the same technique. NLCD was reclassified into four categories: developed, agriculture, undeveloped, and water and for each watershed we calculated the percentage of each land use type and percentage of impervious surface cover. The percentage of catchment developed and catchment impervious cover (ISC) were used as predictor variables in subsequent analyses.

Habitat surveys

In each stream, we delineated experimental reaches encompassing at least one hour of travel time under June 2006 base-flow conditions. We selected the upstream end of each reach by locating an area of constricted flow with the greatest downstream extent of channel uninterrupted by tributary inputs or road crossings. Reach travel time was determined by calculating water travel times using a rhodamine dye release. We used rhodamine tracers because traditional salt tracers proved problematic in several of our urban streams due to high spatial and temporal variation in stream water chloride concentrations. Our study reaches were standardized by water residence time and varied in length from 35 to 200 m. We delineated our study reaches in this manner in order to correctly measure ecosystem function variables (see Sudduth et al. 2011). Habitat surveys were performed in July and August of 2006. We created habitat maps (see Appendix B for examples) of all experimental reaches by determining the longitudinal boundaries and channel widths of riffle, run, pool, and debris dam habitats within each reach (Vermont Water Quality Division 2009). We used a stadium rod and level to survey longitudinal slope for the entire reach and to generate cross-sectional profiles for five randomly selected points within the reach. Reach canopy cover was measured at each cross section using a spherical densiometer. We conducted pebble count surveys of 100 randomly selected sediment particles spaced evenly throughout the study reach (Wolman 1954) to estimate variation in sediment grain size within each stream reach.

Hydrologic data

We created fine-scale flow-habitat maps by measuring velocity and depth values at five evenly spaced points across the active channel, with a sixth measurement in the thalweg, at 30 cross-section locations evenly spaced longitudinally in each reach. In October 2006, we deployed Solinst leveloggers (Solinist Canada, Georgetown, Ontario, Canada) in each stream reach to collect continuous measurements of water level. We used HEC-RAS (US Army Corps of Engineers) to estimate discharge water level and surveyed channel dimensions (software *available online*).[7] We used these data to create a flashiness index (Baker et al. 2004) for use as a predictive variable in macroinvertebrate community analyses.

[7] ⟨www.hec.usace.army.mil/software/hec-ras/⟩

Functional measures

Nutrient and organic matter dynamics were measured concurrently in the same study reaches (for methods and results, see Sudduth et al. 2011). Functional measures were used as potential predictor variables in ordination analyses.

Macroinvertebrate sampling

Macroinvertebrate sampling was conducted at the 12 study sites between May and September 2006 ("summer" sample) and February and March 2007 ("winter" sample) from the same reach as physical and functional measurements were taken. Macroinvertebrate communities were sampled once each season using the North Carolina Department of Water Quality Qual 4 semi-quantitative protocol (North Carolina Division of Water Quality 2006 [hereafter NC DWQ]). This sampling protocol is designed to assess macroinvertebrate diversity in small streams (drainage area <7.7 km^2) and is conducted so that sampling effort is consistent among study sites. Each sample consisted of one 2–3 minute, 1-m^2, 1-mm mesh, kick-net sample from a characteristic riffle; one 500-μm mesh triangular sweep net of stream marginal habitats such as root mats and bank vegetation; an approximately 500-g leaf pack sample collected from rock or snag habitats; and visual assessments of habitats not easily sampled with the above methods (e.g., large rocks or logs). Samples were field sorted and specimens were preserved in 95% ethanol. Non-chironomid taxa were identified at 45× magnification to the lowest possible taxonomic level, typically species (Pennak 1953, Brigham et al. 1982, Merritt et al. 2008). Chironomidae were mounted on slides in CMC-10 medium (Master's Chemical Company, Wood Dale, Illinois, USA), and identified at 400× magnification to genus or species (Epler 2001). Following the NC DWQ protocol, we classified taxa as abundant (>10 individuals), common (three to nine individuals), or rare (one or two individuals). One of the winter urban degraded samples, Goose Creek, was lost, however field notes conclusively indicate the absence of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa in this sample. Thus, this sample was included in EPT richness analyses.

Data analyses

Physical data analyses.—Habitat complexity was determined by counting the number of transitions between different aquatic habitats (riffle, run, pool, and debris dam classifications) for each experimental reach. The transition counts were normalized for all reaches by converting the counts to number of transitions per 100 m reach length. The average number of transitions and standard error was determined for each stream type (F, R, and U). Velocity and point depth measurement averages for each reach were calculated and used to obtain an average and coefficient of variation for each stream type.

The ratio of active channel width to the active channel depth at the thalweg was determined from the field survey cross-section data for each experimental reach. Also, the maximum (smallest width : depth ratio value for each stream) and minimum (largest width : depth ratio value for each stream) incision value from the field survey data were calculated. The average and coefficient of variation of percent canopy cover were determined from spherical densiometer measurements. Physical metrics were compared among stream types using one-way ANOVA with stream type as a single factor (GraphPad Prism v4; GraphPad Software Inc., La Jolla, California, USA). Where the overall effect was significant, we performed post hoc pairwise comparisons (Student-Newman-Keuls) to test for differences among stream types and calculated the magnitude of effect as ω^2, the variance component of the factor in the ANOVA relative to the total variance (Graham and Edwards 2001).

Macroinvertebrate data analyses.—In addition to total species richness, we calculated richness of the orders Ephemeroptera, Plecoptera, and Trichoptera, (EPT) as a measure of pollution-sensitive taxa richness. We also calculated the biotic index (BI) for each site as a measure of overall macroinvertebrate community pollution tolerance. BI was calculated as a weighted mean of taxa tolerance values relative to their abundance, and higher BI values indicate a more pollution-tolerant assemblage (NC DWQ 2006). Individual taxon tolerance values were taken from the NC DWQ benthos standard operating protocol (Lenat 1993, NC DWQ 2006). Taxa for which BI information was not available represented a small minority of taxa and were excluded from the BI calculations. We compared community metrics among stream types using one-way analysis of variance (ANOVA) with stream type as a factor. Where the overall affect of stream type was significant, post-hoc pairwise comparisons and effect size calculations were performed as for physical metrics.

We used least-squares linear regression to quantify correlative relationships between macroinvertebrate metrics and environmental physical and functional variables. As sites were grouped into sampling blocks a priori according to watershed and geological variables, all analyses should include sampling block as a variable. However, as block was not found to be an important predictor of any of our habitat or macroinvertebrate community metrics (data not shown), it was not included in our analyses in order to maximize our power to detect differences among site types.

We examined seasonal macroinvertebrate species compositional similarity among sites using nonmetric multidimensional scaling (NMS) ordination of sites in species space, using Bray-Curtis similarities of square root transformed abundance values (PC-ORD v. 5; McCune and Mefford 2006). Solutions were obtained from 500 runs (250 randomized, 250 with real data) using random starting coordinates. We created joint plots incorporating a second matrix of physical and

TABLE 2. Mean values (±SE) of habitat complexity, flow heterogeneity, floodplain connectivity, and canopy cover of forested, urban restored, and urban degraded stream types.

Parameter	Forested	Urban restored	Urban degraded	df	F	ω^2
Number of habitat transitions per 100-m reach	20.75 ± 1.89	9.250 ± 2.14	9.75 ± 1.11	2, 9	13.55*	41.1
Depth from point measurements (m)	0.065 ± 0.0164	0.175 ± 0.0131	0.158 ± 0.012	2, 9	17.97**	48.5
%CV for depth point measurements	109.3 ± 12.21	73.73 ± 7.59	83.03 ± 9.30	2, 9	2.29	
Velocity from point measurements (m/s)	0.035 ± 0.008	0.023 ± 0.007	0.026 ± 0.012	2, 9	0.47	
%CV for velocity point measurements	209.2 ± 46.43	139.4 ± 5.38	237.0 ± 46.43	2, 9	2.29	
Degree of incision	6.15 ± 0.37	7.14 ± 1.39	4.96 ± 0.77	2, 8	1.64	
Maximum degree of incision	4.74 ± 0.31	5.02 ± 1.06	4.40 ± 0.57	2, 8	0.23	
Longitudinal slope (%)	0.93 ± 0.49	0.51 ± 0.49	0.29 ± 0.11	2, 8	0.78	
Canopy cover (%)	87.54 ± 2.50	53.71 ± 8.28	81.35 ± 4.36	2, 9	10.37*	34.2
Median substrate size (mm)	35.75 ± 11.35	8.0 ± 6.35	4.75 ± 3.75	2, 9	4.75*	17.3

Notes: Degree of incision is the width-to-depth ratio; the maximum degree of incision is the smallest width-to depth ratio. Results and effect sizes are from one-way ANOVAs with stream type as a factor. Where the overall effect was significant, we performed post hoc pairwise comparisons (Student-Newman-Keuls) to test for differences among stream types and calculated the magnitude of effect as ω^2, the variance component of the factor in the ANOVA relative to the total variance.

* $P < 0.05$; ** $P < 0.01$.

functional variables. We set a minimum r^2 of 0.30 to identify geomorphological and functional parameters correlated with macroinvertebrate community structure at different sites.

We assessed the importance of time since restoration to macroinvertebrate recovery by evaluating separately collected macroinvertebrate monitoring data from Rocky Branch both within the restoration and at an unmanipulated upstream reference, and from Sal's Branch, a forested reference site in Umstead Park, North Carolina. Monitoring data were collected using the same NC DWQ Qual-4 protocol as for this study. Pre-restoration samples were collected for Rocky Branch in December 2000 and post-restoration data were collected in December 2003, November 2004, December 2005, and December 2006. Reference data were collected from Sal's Branch in March 2002, March 2003, and May 2004. We evaluated the importance of time since restoration to total species richness, EPT richness, and community BI for Rocky Branch. We calculated the change in each community metric by subtracting the pre-restoration value from the post-restoration value for each monitoring year ($\Delta_{\mathrm{Rest}} = \mathrm{Metric}_{\mathrm{post[yr}\ i]} - \mathrm{Metric}_{\mathrm{pre}}$). We accounted for community structure changes due to factors other than restoration by performing the same calculation for the upstream reference ($\Delta_{\mathrm{Up}} = \mathrm{Metric}_{\mathrm{up[yr}\ i]} - \mathrm{Metric}_{\mathrm{up\ pre}}$) and then calculated the effect of restoration by taking the difference of the two (Restoration Response $= \Delta_{\mathrm{Rest}} - \Delta_{\mathrm{Up}}$). This is similar to the "raw effect score" for taxon abundance calculations from impact assessment studies (Weiss and Reice 2005), but applied to community-level metrics. We evaluated species compositional similarity among these samples using the same NMS ordination protocol as above.

RESULTS

To test our overarching hypothesis that positive restoration outcomes would lead urban restored streams to become more similar to minimally impacted sites, we compared physical and biological attributes among the three stream types. Excluding Lower Mud Creek did not change the conclusions of any of our relationships of physical metrics among stream types and there was no consistent pattern in the effect of removing this site. However, in every case, removing Lower Mud Creek from biological analyses increased the strength of the observed relationship (Fig. 3), and for some analyses, resulted in a stronger overall effect of stream type (Tables 3–4). For all analyses, we show comparisons with and without Lower Mud Creek included as a forested site.

Habitat

Urban streams had significantly deeper channels, smaller substrate sizes, and less reach-scale habitat variation (transitions between riffles, runs, and pools) than their forested counterpart (Table 2, Fig. 2). For each of these metrics, urban restored streams were indistinguishable from their urban degraded counterparts and significantly different from the forested streams. We found a significant difference between urban degraded and urban restored reaches in only a single habitat metric; restored urban streams had significantly lower riparian canopy cover than their unrestored counterparts.

Our hydrologic metrics did not differ between stream types. Stream velocities and flow heterogeneity were highly variable within stream types. There was no difference in either average or maximum degree of incision among stream types (Table 2).

Biological structure

Macroinvertebrate community richness was similar across stream types in summer, while in our winter sampling our three forested sites (excluding LMC) had significantly higher taxa richness than their restored or urban counterparts (Table 3, Fig. 3). In both seasons, species of Chironomidae made up 56.6% (±4.5% [mean ± SE]) and 44.9% (± 2.6%) of the taxa found in urban and restored streams respectively, and only 26.7% (± 5.0%) of the taxa in the forested streams (Appendix C).

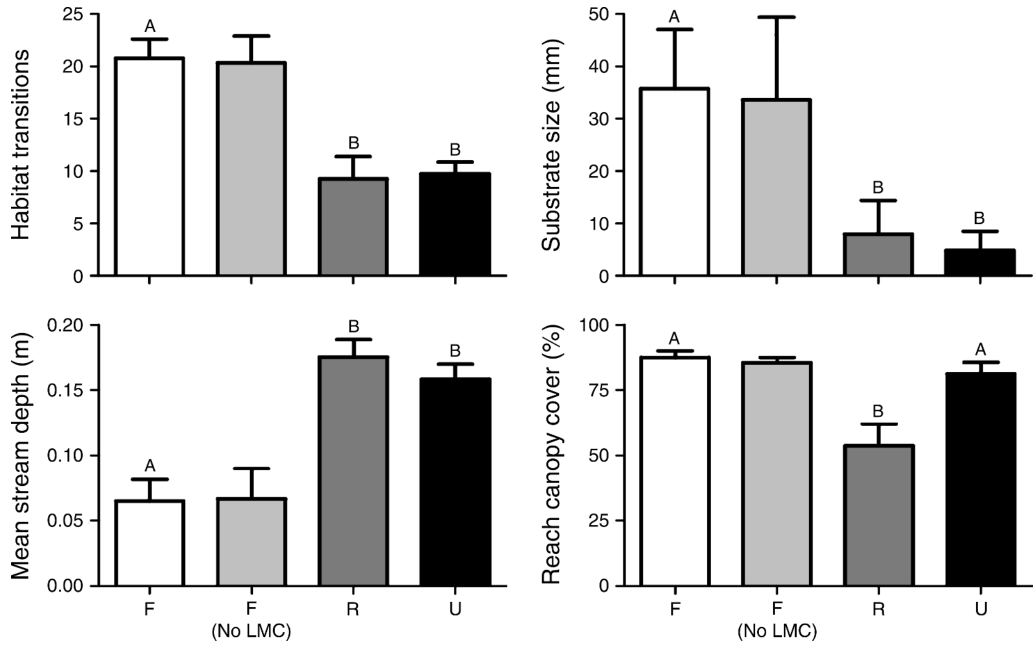

FIG. 2. Mean values (and SE) of habitat transitions (number of transitions per 100 m reach length), substrate size (mm), mean stream depth (m), and reach canopy cover (% covered) for forested sites (F), forested sites excluding Lower Mud Creek (LMC), urban restored (R), and urban degraded (U) sites (one-way ANOVA, $P < 0.05$). Differences among stream types are indicated by different letters (Student-Newman-Keuls post hoc multiple comparisons test).

The three forested sites had higher mean EPT richness than urban and restored sites in both summer and winter (Table 3, Fig. 3). Summer and winter biotic integrity scores were lower (higher number of sensitive taxa) in these three forested streams than in the urban restored and urban degraded streams (Table 3, Fig. 3).

Among quantified watershed variables, watershed imperviousness was found to be the best single predictor of EPT richness, although the trend was significant only in winter ($r^2 = 0.54$, $P < 0.01$). Biotic index was positively correlated with watershed imperviousness in both summer ($r^2 = 0.50$, $P < 0.01$) and winter ($r^2 = 0.40$, $P < 0.05$) (Table 4, Fig. 4).

Among the many in-channel structural and functional variables measured, number of habitat transitions per 100 m was the only reliable predictor of EPT richness, with habitat complexity positively correlated with the number of EPT in both summer ($r^2 = 0.54$, $P < 0.01$) and winter ($r^2 = 0.46$, $P < 0.05$). Habitat complexity was strongly negatively correlated with BI scores in both summer ($r^2 = 0.70$, $P < 0.001$) and winter ($r^2 = 0.67$, $P < 0.01$). Removing Lower Mud Creek from the analyses increased the strength of the observed relationships, but had a stronger affect on EPT richness than BI (Table 4, Fig. 4).

NMS ordination results revealed large differences in community composition between stream types. Two-dimensional NMS solutions were best for both summer and winter. The summer NMS ordination had a final stress of 0.13 and explained 78.2% of compositional similarity, 40.2% along axis 1 and 37.8% along axis 2 (Fig. 5a). The winter NMS had a final stress of 0.078 and explained 88.2% of compositional similarity among sites, 58.6% along axis 1 and 29.7% along axis 2 (Fig. 5b). With the exception of Lower Mud Creek, forested sites clustered closer to one another in winter than in

TABLE 3. Mean values (±SE) of macroinvertebrate community metrics with and without Lower Mud Creek (LMC).

Parameter	Forested	Forested (no LMC)	Urban restored	Urban degraded	df	F	df (no LMC)	F (no LMC)	ω^2 (no LMC)
Summer species richness	20.0 ± 2.3	22.3 ± 0.3	15.3 ± 3.9	15.0 ± 2.6	2, 9	0.86	2, 8	1.70	
Summer EPT richness	5.8 ± 2.0	7.3 ± 1.8	1.8 ± 0.6	1.5 ± 0.9	2, 9	3.28	2, 8	8.79*	32.1
Summer BI	5.8 ± 0.8	5.4 ± 1.0	7.7 ± 0.5	8.0 ± 0.4	2, 9	4.35*	2, 8	5.19*	20.2
Winter species richness	29.3 ± 3.4	32.3 ± 1.9	15.5 ± 4.7	14.7 ± 3.8	2, 8	4.19	2, 7	5.88*	24.5
Winter EPT richness	10.3 ± 2.4	12.3 ± 1.7	1.5 ± 0.5	1.3 ± 0.9	2, 9	11.46**	2, 8	34.47***	67.0
Winter BI	5.6 ± 0.3	5.4 ± 0.3	7.0 ± 0.4	7.3 ± 0.5	2, 8	5.55*	2, 7	6.04*	25.1

Notes: Abbreviations are: EPT, Ephemeroptera, Plecoptera, and Trichoptera; BI, biotic index. Results and effect sizes are from one-way ANOVAs performed with and without LMC, with metric as a factor.
* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

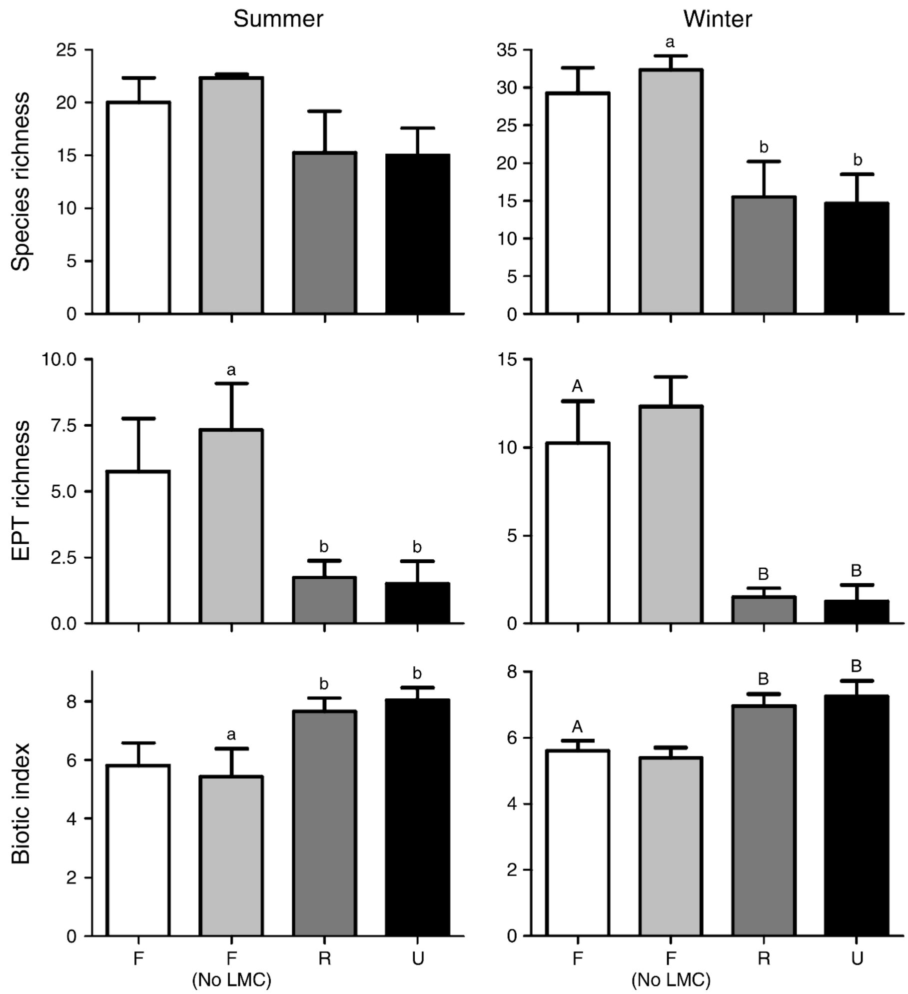

FIG. 3. Mean values (and SE) of summer and winter species richness (number of species); Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness (number of species); and biotic index for forested sites, forested sites excluding LMC, urban restored, and urban degraded sites (one-way ANOVA, $P < 0.05$). Differences among stream types is indicated by different letters (Student-Newman-Keuls post hoc multiple comparisons test). Different uppercase letters indicate differences among stream types for the entire data set; different lowercase letters indicate differences among stream types excluding LMC.

summer (Fig. 5). In spite of their close proximity (<50 m of forest between the reaches, Fig. 6), Lower Mud Creek and Mud Creek Tributary did not cluster together in either season.

In analyses of long-term data from Rocky Branch, we found no significant effect of time since restoration on total species richness, EPT richness, or BI. A three-dimensional NMS solution was best for explaining compositional similarity (final stress = 0.09, cumulative $r^2 = 0.866$) among Rocky Branch macroinvertebrate communities collected as part of this study, upstream and restored samples collected for restoration monitoring, and reference data from Sal's branch and the winter forested samples from this study. Regardless of year, restoration monitoring samples from Rocky Branch clustered more closely to restored samples collected as part of this study and impacted upstream reference samples than to forested samples (Fig. 7).

DISCUSSION

We hypothesized that, if restoration is effective at improving degraded urban stream ecosystems, both the geomorphology and biota at restored sites would more closely resemble forested sites than would their urban counterparts. While it would be overly optimistic to expect restored stream reaches to become identical to reference sites, successful restoration ought to lead to stream habitat and biological communities that are distinguishable from unrestored urban streams. In this survey, urban restored streams differed significantly from their unrestored urban counterparts in only a single metric: having reduced canopy cover as a direct result of project implementation (Fig. 2). These results

TABLE 4. Results from least-squares linear regression analyses of macroinvertebrate community metrics with and without LMC.

Metric	With LMC r^2	With LMC df	With LMC F	No LMC r^2	No LMC df	No LMC F
Watershed imperviousness						
Summer EPT	0.26	1, 10	3.44	0.33	1, 9	4.43
Summer BI	0.50	1, 10	10.18**	0.51	1, 9	9.55*
Winter EPT	0.54	1, 10	11.85**	0.58	1, 9	12.23**
Winter BI	0.40	1, 9	6.03*	0.40	1, 8	5.24
Habitat transitions						
Summer EPT	0.54	1, 10	11.72**	0.86	1, 9	56.62***
Summer BI	0.70	1, 10	22.91***	0.81	1, 9	39.36***
Winter EPT	0.46	1, 10	8.61*	0.58	1, 9	12.60**
Winter BI	0.67	1, 9	18.01**	0.74	1, 8	22.41**

Note: Watershed imperviousness is the percentage of the watershed with impervious surface cover.

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

suggest that despite expenditures of >US$1 million per project, these restored streams did not have improved habitat complexity or detectable changes in their macroinvertebrate communities. The deep, sandy, simplified channels in urban catchments suggest that hydrological differences, particularly storm events, are the major habitat structuring force in our study channels. Stormwater is rarely, if ever, addressed by NCD, therefore this is likely a significant barrier to urban stream restoration success (Walsh et al. 2005*a*, Bernhardt and Palmer 2007).

The similarity in summer total species richness among stream types (Table 3, Fig. 3) is likely due to high richness of more-tolerant non-EPT taxa in urban and urban restored sites (Appendix C). Higher winter EPT richness probably accounts for the significant effect seen in winter (Table 3, Appendix C). Higher EPT richness at forested sites (Fig. 3) is consistent with the expectation that urbanization typically results in the loss of these sensitive taxonomic groups (Morse et al. 2003, Roy et al. 2003, Cuffney et al. 2010). Urban restored channels did not have higher EPT richness than urban degraded channels (Fig. 3), suggesting that natural channel design is not mitigating the factors responsible for sensitive taxa loss at these locations. The difference in biotic integrity between urban restored and forested channels, and their similarity to urban degraded channels (Fig. 3) indicates that in addition to having lower sensitive taxa richness (i.e., lower EPT richness), these channels contain more tolerant assemblages across all invertebrate groups.

Regression analyses revealed a strong relationship between EPT richness and watershed ISC (Table 4, Fig. 4). Watershed imperviousness is not something easily addressed by reach-scale restoration, thus prioritizing projects with lower ISC or evaluating the spatial arrangement of ISC (Moore and Palmer 2005) during the planning stages may increase the likelihood of successful restoration. However, although ISC cannot be easily altered, its effects may be mitigated by catchment-based stormwater retention efforts (Walsh et al. 2005*a*).

Urban degradation leads to compositionally distinct macroinvertebrate communities, which is not successfully mitigated by reach-scale restoration. NMS plots revealed that species composition of restored streams were more similar to each other and to urban degraded streams than to forested streams (Fig. 5). The lack of grouping of forested sites in summer illustrates that although forested sites possess multiple sensitive EPT taxa that primarily delineate them from urban sites (Fig. 5a), there are inter-site compositional differences across all taxonomic groups (Appendix C). While this could be due to the fact that sites were sampled over several summer months, it is also likely local-scale habitat filters differed among forested sites and influenced community composition (Poff 1997). Additionally, although these sites are best-case scenarios of minimally impacted Piedmont streams, they are still subject to human impacts, the extent of which varies among catchments. The dissimilarity between Lower Mud Creek and Mud Creek Tributary in spite of their proximity further suggests that Mud Creek still experiences urban influences at the lower site. In fact, Lower Mud Creek is more similar to the urban degraded site Upper Mud Creek in both seasonal NMS plots (Fig. 5).

Joint plots suggest that summer species composition is explained first by underlying geology (axis 1), and second by catchment and reach-scale urban stressors (axis 2). Axis 1 represents a gradient of high percent dilution to high chloride ion concentration (Fig. 5a). Higher percent dilution is characteristic of streams with higher groundwater and hyporheic exchange (Griffith et al. 2002) and this variable correlated mainly with Northern Outer Piedmont sites. High chloride concentration is probably caused by low groundwater and hyporheic exchange and low summer baseflows characteristic of Triassic basin and Slate Belt streams (Griffith et al. 2002). Axis 2 represents an urban vector that encompasses differences in habitat and water quality,

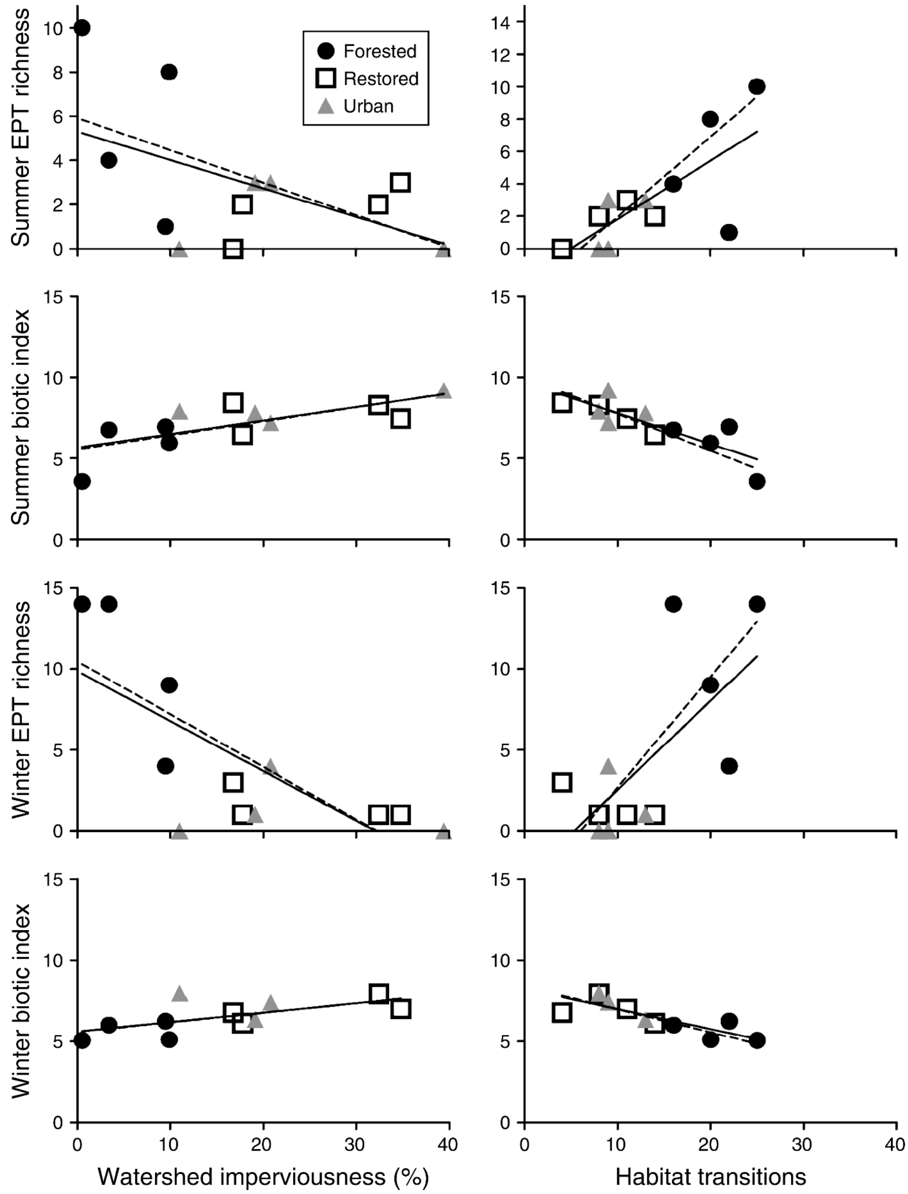

FIG. 4. Linear regression of seasonal macroinvertebrate community metrics vs. watershed imperviousness (percentage of impervious cover in watershed; left panels) and number of habitat transitions per 100-m reach (right panels) with (solid line) and without (dashed line) Lower Mud Creek. Table 4 gives r^2 values.

canopy cover, and hydrological differences and consequently separates forested sites from those in urban catchments (Fig. 5a). This axis best represents our original hypothesis that if urban restoration effectively addresses factors responsible for sensitive taxa loss, restored sites would be at least intermediate between forested and urban degraded endpoints. Our analysis finds no evidence of directional change in composition due to restoration.

The closer clustering of forested sites in winter (Fig. 5b) likely reflects the widespread winter prevalence of shredder taxa such as *Tipula*, *Gammarus*, and *Amphinemura*. An urban vector along axis 1 similar to that found in summer was the primary axis separating forested sites from urban and restored sites, and once again we found no evidence that the restored stream benthos were distinct from their unrestored urban counterparts (Fig. 5b). Additionally, stream nitrogen concentrations (TN, NO_3-N) were correlated with urban and restored species composition. Urban catchments deliver more nutrients to streams than undeveloped ones, and nutrient pollution has long been known to influence macroinvertebrate community structure and impair aquatic communities (Bernhardt et al. 2008). There was no clear effect of underlying geology in the winter ordination, in this season chloride concentrations

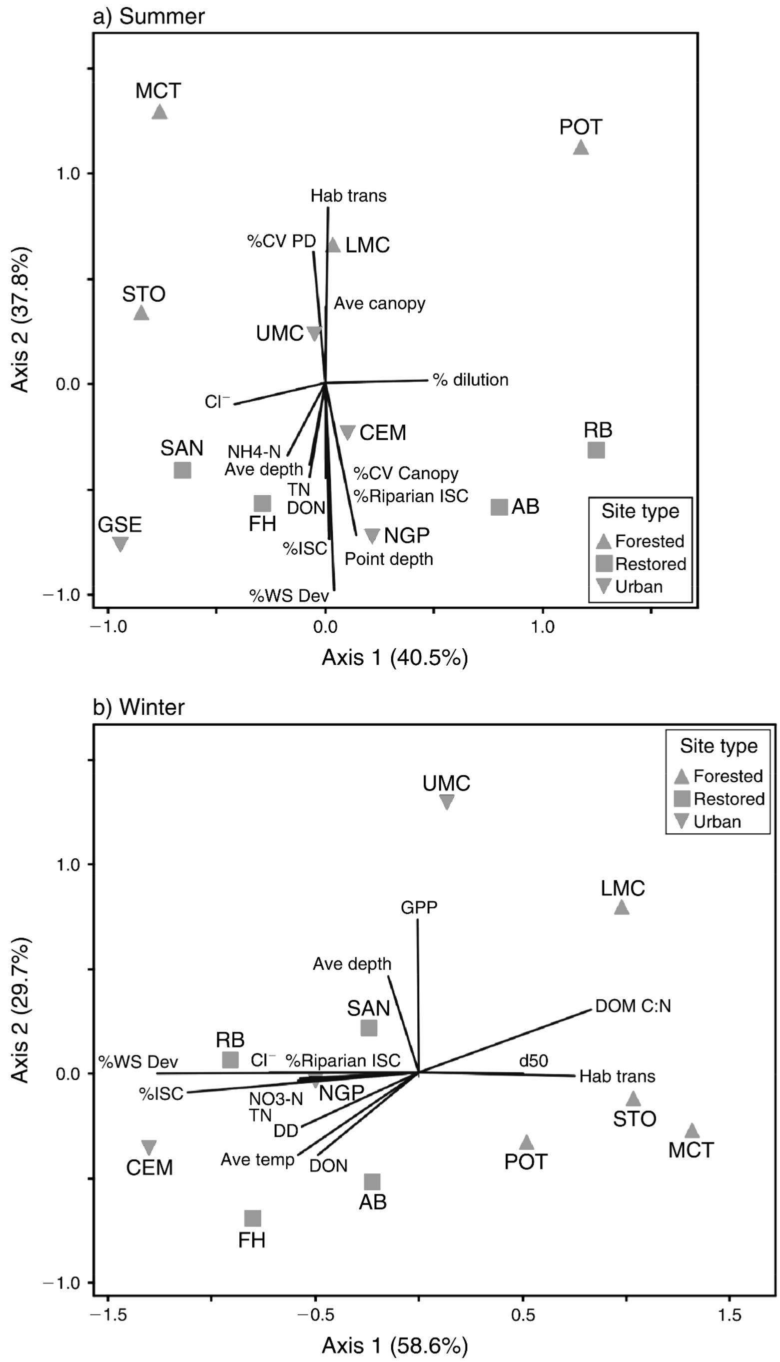

FIG. 5. Two-dimensional joint plots of nonmetric multidimensional scaling (NMS) ordination of (a) summer and (b) winter square-root transformed Bray-Curtis similarities with environmental variables. Minimum explanatory r^2 for environmental variables = 0.3. Final stress = 0.13 for summer and 0.078 for winter. Cumulative r^2 = 0.782 and 0.882 for summer and winter, respectively. Site abbreviations are: AB, Walnut Creek Tributary; CEM, Cemetery Creek; FH, Forest Hills; GSE, Goose Creek; LMC, Lower Mud Creek; MCT, Mud Creek Tributary; NGP, Ellerbe Creek; POT, Pot's Branch; SAN, Sandy Creek; STO, Stony Creek; UMC, Upper Mud Creek. Environmental variable abbreviations (followed by the units in which they were measured) are: Hab trans, habitat transitions; %CV PD, percent coefficient of variation of point depth; Ave canopy, average canopy cover; Cl^-, chloride concentration (ppm); NH4-N, ammonium-N concentration (ppm); Ave depth, average water depth (m); DON, dissolved organic nitrogen concentration (ppm); %ISC, percentage of watershed with impervious surface cover; %WS Dev, percentage of watershed developed; Point depth, stream point depth (m); %Riparian ISC, percent riparian buffer impervious surface cover; %CV Canopy, percent coefficient of variation of canopy cover; GPP, gross primary production ($g \cdot m^{-1} \cdot d^{-1}$); NO3-N, nitrate-N concentration (ppm); TN, total nitrogen (ppm); DD, degree days; Ave temp, average stream temp (°C); d50, median particle size (mm); DOM C:N, dissolved organic matter C:N.

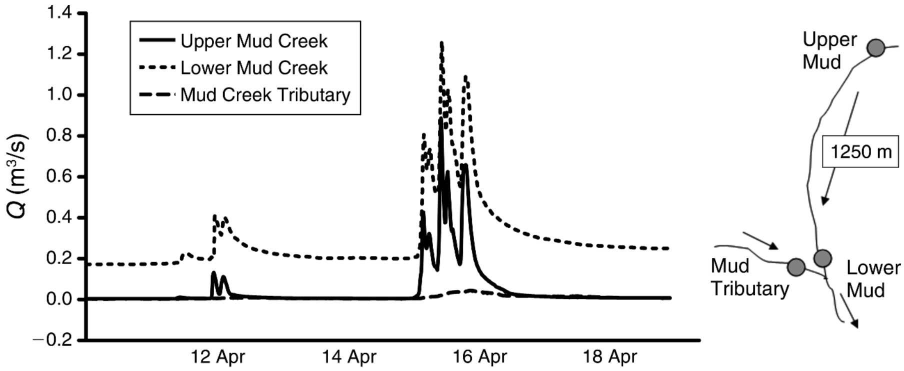

FIG. 6. Hydrographs (flow [Q] over time) from April 2007 and station map of Upper Mud Creek, Lower Mud Creek, and Mud Creek Tributary. Flow readings were made in 10-minute increments.

were highly correlated with impervious cover, suggesting that winter chloride concentrations are dominated by road salt use (e.g., Kaushal et al. 2005). Deep streams with high gross primary production (GPP) were delineated from other sites along axis 2 (Fig. 5b). Together the two axes appear to separate the open canopy urban streams with high nitrogen loading and higher temperatures from closed-canopy forested streams with the cooler temperatures and high streamwater C:N ratios characteristic of forested heterotrophic streams (Fisher and Likens 1973).

The restored streams we studied more closely resembled urban rather than forested endpoints both structurally and biologically, suggesting that restoration activities have not yet led to the recovery of sensitive macroinvertebrate taxa in these streams. The large number of metrics measured in the context of this study provides an unprecedented opportunity to explore what factors are most important for community recovery.

Lower Mud Creek: an unacceptable reference site proves an effective case study of the field of dreams hypothesis

Due to the extreme difficulty of locating watersheds without significant urban or agricultural activity in the North Carolina Piedmont, we made the decision to include Lower Mud Creek (LMC) as one of our forested streams. Based on its geomorphology and its location within ~100-year-old mixed deciduous forest within the protected Duke Forest, the segment we selected on Mud Creek (LMC) appeared to be an acceptable forested stream reach for inclusion in our study. Despite the high habitat heterogeneity, connected floodplain and high canopy cover of LMC, it supports a very depauperate faunal community. Although LMC proved to be a less than ideal forested stream replicate, the mismatch between physical habitat and biological community structure make it an ideal case study for investigating the limitations of reach-scale restoration. Lower Mud Creek was indistinguishable from other forested sites in most measured geomorphic variables, including habitat complexity. Among forested sites however, Lower Mud Creek had the fewest EPT taxa, and the most tolerant macroinvertebrate assemblage (highest BI value).

The positive correlation between habitat complexity and species richness is well documented (Macarthur and Macarthur 1961, Minshall 1984, O'Connor 1991, Downes et al. 1998, Allan 2004) but may not be causal (Palmer et al. 2010). We speculate that our measure of habitat complexity serves as an indicator of hydrologic disturbance as well as a direct measure of habitat suitability. As such, we must caution that the observed strong positive correlation between habitat complexity and sensitive invertebrate taxa (Table 4, Fig. 4) does not necessarily support the assumption that an increase in habitat complexity will improve biological communities. Indeed, our findings suggest that habitat restoration will prove ineffective if urban stormwaters rapidly rehomogenize restored stream segments, as seen in previous urban restorations (Larson et al. 2001, Booth 2005).

Prior work has suggested that landscape or stream network fragmentation or habitat homogenization may represent an important barrier to macroinvertebrate dispersal in urban catchments and thus may inhibit community recovery in restored urban systems (Blakely et al. 2006, Urban et al. 2006, reviewed in Smith et al. 2009). The proximity of LMC to Mud Creek Tributary (<50 m, Fig. 6), the forested site within our data set with the least impervious cover and the highest diversity of sensitive macroinvertebrate taxa suggests that LMC is not dispersal limited. Indeed, several of the EPT taxa found in the tributary but not in LMC have been sampled in the riparian vegetation surrounding LMC suggesting that dispersal is likely (C. R. Violin, *unpublished data*). The apparent structural integrity and impaired macroinvertebrate community suggests that while habitat complexity is important to faunal diversity, it is, on its own insufficient to support the recovery of biotic communities. This is an important caveat to the utility of the "field of dreams" hypothesis. Mud Creek experiences urban influences along its length, and although the kilometer preceding our LMC study site is entirely forested, this site has a characteristic urban hydrograph due to upstream catchment urbanization

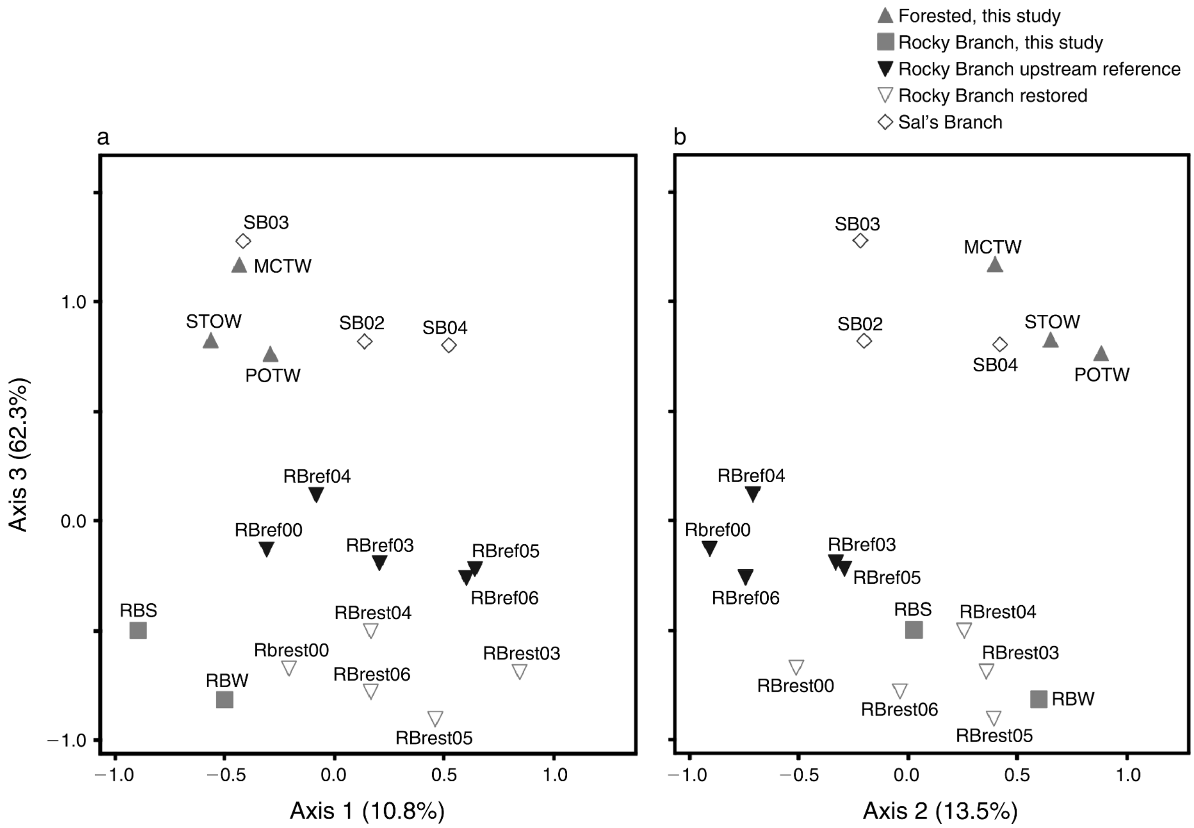

FIG. 7. Two-dimensional representations of (a) axis 3 vs. axis 1 and (b) axis 3 vs. axis 2 of three-dimensional NMS of all Rocky Branch and forested site data (final stress = 0.09, cumulative $r^2 = 0.866$). Axis 3 explained the majority of compositional differences ($r^2 = 0.623$); therefore two-dimensional plots are shown relative to this axis. Site abbreviations are: RBS, Rocky Branch summer sample (from this study); RBW, Rocky Branch winter sample (from this study); RBref00, Rocky Branch monitoring, upstream reference 2000 (pre-restoration); RBref03, Rocky Branch monitoring, upstream reference 2003; RBref04, Rocky Branch monitoring, upstream reference 2004; RBref05, Rocky Branch monitoring, upstream reference 2005; RBref06, Rocky Branch monitoring, upstream reference 2006; RBrest00, Rocky Branch monitoring, restored 2000 (pre-restoration); RBrest03, Rocky Branch monitoring, restored 2003; RBrest04, Rocky Branch monitoring, restored 2004; RBrest05, Rocky Branch monitoring, restored 2005; RBrest06, Rocky Branch monitoring, restored 2006; MCTW, Mud Creek Tributary winter; POTW, Pot's Branch winter; STOW, Stony Creek winter; SB02, Sal's Branch 2002; SB03, Sal's Branch 2003; SB04, Sal's Branch.

(Fig. 6). Hydrologic disturbance is a major driver of macroinvertebrate community structure, as species adapt to local hydrologic conditions (Resh et al. 1988, Townsend et al. 1997, Lake 2000). Previous work has shown that the effective discharge(s) responsible for in-stream habitat structure are not necessarily those responsible for ecological processes such as invertebrate dislodgement (Doyle et al. 2005). Thus, while the urban hydrology of LMC does not cause significant scouring or substrate/habitat homogenization, it may be sufficient to impair aquatic fauna. It is also possible that periods of storm flow introduce urban derived contaminants (e.g., Makepeace et al. 1995, Beasley and Kneale 2002, Kolpin et al. 2002) that may further stress sensitive aquatic taxa. The special case of Mud Creek suggests that even if a restoration project could build intact channels with high floodplain connectivity surrounded by 100-year-old trees and in close proximity to source populations; such an effort would be unsuccessful at promoting the recovery of a diverse macroinvertebrate assemblage containing sensitive taxa unless the project also was able to mitigate storm flows or the associated pulses of sediments and contaminants through catchment-based efforts (Walsh et al. 2005*a*, 2007).

Time since restoration

Restoration itself is a catastrophic disturbance to already impaired stream ecosystems (Tullos et al. 2009), and as a result, we expect time lags between restoration implementation and community recovery. One possible explanation for the lack of significant recovery of habitat or biological communities within our restored streams is that insufficient time was allowed for recovery between the restoration implementation and our sampling effort. Our restored study sites were restored one to seven years prior to our sampling effort, however the small sample size and the lack of pre-restoration data precluded us from evaluating the potential role of time lags in our initial data set. To address this question, we were able to examine long-term data from one of our study sites (Rocky Branch, Raleigh, North Carolina). For this data set we found no significant effect of time since restoration on total species richness, EPT species richness, or biotic index for the five-year post-restoration monitoring period. Three-dimensional NMS ordination of long term monitoring data for the restored reach of Rocky Branch revealed that macroinvertebrate communities from the restored reach of Rocky Branch remained similar in composition to the unrestored upstream urban reach during the five years of post restoration monitoring and remained consistently different from benthic communities in the closest reference stream Sal's Branch (a tributary to Pot's Branch) (Fig. 7). Thus, long-term monitoring of Rocky Branch further supports the conclusions of our synoptic sampling effort, with no evidence of directional change in restored stream reaches either away from the pre-restoration composition or toward reference stream conditions. All available evidence suggests that merely waiting longer prior to evaluating a restoration project is unlikely to lead to different conclusions.

Conclusions

Our results demonstrate the limited utility of reach-scale restoration to combat the overwhelming effects of watershed urbanization. Within this study, the only demonstrable effect of restoration activities was to remove riparian trees, a practice that may impede recovery. In our study, restoration failed to improve habitat over impaired urban channels, suggesting that watershed level hydrologic processes are degrading restoration efforts. Expanding urban restoration planning beyond the reach scale to include watershed-scale impacts will lead to better restoration design and more positive restoration outcomes.

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APPENDIX A

Study site UTM coordinates (*Ecological Archives* A021-087-A1).

APPENDIX B

Example habitat transition maps and photos for study site block 1 (*Ecological Archives* A021-087-A2).

APPENDIX C

Study site species lists (*Ecological Archives* A021-087-A3).