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Urban stream restoration projects have been undertaken to improve physical, chemical, and biological integrity, but there has been little assessment of the effectiveness of these projects in restoring ecological function. I looked at the effect of restoration on improving water quality, periphyton, nutrient uptake, and macroinvertebrate communities compared to unrestored streams. When there was a restoration effect, I compared three types of restoration structures (riffle, cross vane, and step pool) in the restored streams to unrestored streams. Two years after restoration, restored streams did have a more oxygen rich environment. The structures provided hard substrate for algal growth which positively affected nutrient uptake length. There was also a strong trend toward faster uptake velocity and greater uptake rate in restored streams. There was a trend indicating riffles were more beneficial than cross vanes and step pools. The trend suggested that riffles allowed for more mean algal growth and had better water quality ratings. Despite the benefits of the restoration, there was little improvement in biotic integrity based on the North Carolina Biotic Index.

THE EFFECTS OF RESTORATION STRUCTURES ON NUTRIENT UPTAKE AND MACROINVERTEBRATE COMMUNITIES IN URBAN RESTORED STREAMS IN GREENSBORO, NORTH CAROLINA

by

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APPROVAL PAGE

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CHAPTER I

INTRODUCTION

Urban areas have been increasing in size over the last century. In 1870, approximately 5% of the United States population lived in urban areas; that percentage escalated to approximately 79% by 2000 (US Census Bureau). This trend towards urbanized living has overloaded streams' abilities to efficiently cycle the large influx of anthropogenic pollutants, nutrients, and sediments (Meyer et al. 2005).

As urban sprawl increases, so does the amount of impervious surfaces. Impervious surfaces in urban areas include parking lots, roads, sidewalks, and rooftops. Urban runoff from these surfaces contributes to loading of nutrients, ions, metals, pesticides, pet waste and sediments into streams (Paul and Meyer 2001). As the percentage of impervious surface increases, so does the amount of urban runoff. At 10-20 % impervious surfaces, runoff increases 2-fold; at 35-50 %, runoff increases 3-fold; and at 75-100 %, runoff increases more than 5-fold (Arnold and Gibbons 1996). The increasing amount of anthropogenic influx into urban streams alters the stream's geomorphology, hydrology, and biological communities (Paul and Meyer 2001).

Increased urbanization is detrimental to the natural geomorphology of streams. Bank erosion and increased channel incision are negative effects of stormwater runoff, which loads sediments into the streams and changes channel structure (Trimble 1997). As sediments enter the stream from runoff, the stream channel initially narrows and combined with increased flow from runoff, channel incision begins. This leads to bank erosion and, eventually, stream widening (Paul and Meyer 2001). Streams become shallow, wider, less sinuous, and the stream bed sediments become more homogeneous because of increased siltation (Brasher 2003).

The impacts of urban development on streams have been a concern for water resource managers since the 1960's (ASCE 2003). In 1972, goals to restore and maintain chemical, physical and biological integrity of our Nation's waters were outlined by Congress to amend the Water Pollution Control Act (Duda et al. 1982). Amendment 319 outlines standards to clean up impaired waterways. This amendment was adopted in North Carolina by the Environmental Management Commission in 1997. It was the first plan to control for both point and non-point pollutants (US EPA 2006).

Mitigation efforts include measures taken to reduce negative impacts to the environment through wetland or stream restoration (NCDENR 2001). The mitigation process was designed to have no net loss in wetlands (US EPA 2006). Historically, restoration efforts were undertaken to enhance habitat for fish species (Thompson 2006). Currently, restoration efforts attempt to return in-stream geomorphology and riparian vegetation to a pre-disturbed state (Cairns 1989). The restoration process for streams includes physical alteration of disturbed channels with hope that the biological community's health will also improve (Booth 2005). An important process in recovery of ecosystems after completion of physical restoration is re-coupling of aquatic-terrestrial environments (Haapala and Muotka 1998, Muotka et al. 2002).

Typical restoration projects for urban streams include bank stabilization, increased sinuosity to stream channels, dam and culvert removal, and habitat improvement (ASCE 2003). Replanting of riparian vegetation, laying geotextile fabrics, and installing structures such as boulders and root wads are useful to stabilize the stream bank and increase sinuosity. Live willow stakes are often used in riparian planting to hold layers of vegetation in the ground and, if installed properly, take root and grow. Geotextile fabrics are woven synthetic or natural fibers that are staked into the soil of the riparian area to help prevent erosion while the new vegetation becomes established. Boulders and root wads are placed in the banks of the stream to promote sinuosity by directing flow meanders. By increasing sinuosity, energy can be dissipated and flow reduced so sediment transport will not exceed what is needed for the stream system to maintain channel integrity (Riley 1998).

In-channel structures are installed in restoration projects for grade control and to provide in-stream heterogeneity. Drop structures, such as step pools, are installed to help dissipate the erosive force of an elevation drop of 2 or more feet. Channel constrictors, such as cross vanes, and deflectors, such as vanes, are designed to catch gravel behind them to create pools, riffles and meanders. Riffles are designed to create in-stream heterogeneity and oxygenate water. These structures also create habitat for in-stream biota. Rocks may be installed below a culvert to minimize the erosive force from water flowing out of the culvert. Restoration structures in streams, if installed properly, can help re-create a more natural grade to the stream (Riley 1998).

In-stream structures installed in restored streams have the potential to become habitat for macroinvertebrates. Lower order streams are more likely to undergo changes in macroinvertebrate species as urbanization increases (Quinn et al. 1992). Plant and animal species may be sensitive to metals or herbicides introduced into the stream, or may not be able to tolerate changes in the physical environment such as streambed sediment change, increased turbidity, removal of riparian vegetation, or nutrient enrichment (Brasher 2003, Paul and Meyer 2001). In urban areas, degraded streams become disconnected from their riparian zones, creating an unsuitable habitat for intolerant macroinvertebrate species (Riley 1998).

Standardized methods have been developed for sampling macroinvertebrates and interpreting the results (Whiles et al. 2000).Water quality rating systems have been developed using macroinvertebrate indices (Lenat 1993). The EPT taxa richness index was created to characterize the presence of intolerant macroinvertebrate species. The more intolerant species present, the better the stream quality assignment. However, this index may not be the best to use when comparing urban streams to each other. Robb (1980, 1992), as reported in Blakely et al. (2006), revealed that even after restoration was completed on urban streams in New Zealand, tolerant taxa had replaced the intolerant macroinvertebrate species that were present before disturbance to the streams. The North Carolina Biotic Index may be a better indicator when comparing urban streams because it includes more species that are likely to be found in urban streams (Lenat 1993).

Nutrient uptake and retention are important stream ecosystem functions, but it is unclear whether urban stream restoration results in improved uptake or retention of nutrients. Urban streams have higher concentrations of inorganic nitrogen than forested streams (Groffman et al. 2004, Grimm et al. 2005, Wollheim et al. 2005). Humans have significantly increased the input of nitrogen in terrestrial systems (Grimm et al. 2005). Inorganic nitrogen enters from nonpoint pollution from pet wastes, fertilizers, and waste water treatment leakage (Wollheim et al. 2005). Urban streams often lack hard substrate and channel heterogeneity, which can decrease contact time of nutrients with possible assimilation or removal sites. Urban streams normally have longer nutrient spiraling length, lower uptake rate and uptake velocity, which results in less nutrient retention capability compared to more pristine streams (Grimm et al. 2005, Meyer et al. 2005). High nutrient export rates can cause problems for downstream reaches (higher order streams, lakes, and the ocean), including eutrophication, declines in water clarity, toxic algal blooms, and taste and odor problems (Brett et al. 2005).

Nutrient uptake parameters are useful metrics for comparing nutrient processing among streams (Stream Solute Workshop 1990). Nutrient uptake length (S_w), the distance a molecule moves downstream before being removed from the water column, uptake velocity (v_f), the vertical removal of the nutrient from the water column to the uptake site, and uptake rate (U), the amount of the nutrient removed per unit area per unit time, are interrelated:

$$S_w = 1/k$$

$$v_f = \frac{(u*h)}{S_w}$$

$$U = v_f C$$

Where k is the distance specific uptake rate (m^{-1}), u is water velocity, h is depth, and C is nutrient concentration (Stream Solute Workshop 1990). Stable isotope enrichments can measure ambient N fluxes without significantly altering dissolved N concentrations (Hamilton et al. 2001, Peterson et al. 2001). Measurements of uptake parameters can be used to study restoration effectiveness of different structures at nutrient removal compared to unrestored streams.

In Greensboro, North Carolina, urban stream restoration projects have included bank stabilization, increased sinuosity of the stream channel, installation of restoration structures (riffles, cross vanes, step pools, root wads, j-hooks and vanes), and re-planting of the riparian zone (City of Greensboro 2006). In addition, in several unrestored urban streams, the riparian areas have been allowed to grow naturally along the stream. Post-restoration evaluation is essential to guide future projects for effective and ecologically successful restoration (Kondolf 1995, Palmer et al. 2005). It is important to monitor restored streams to determine if further improvements need to be made (Kondolf 1995).

This project assessed water quality parameters, algal biomass accumulation, ammonium (NH_4^+) uptake rate using ^{15}N-NH_4Cl, and NCBI to determine if restored streams were promoting a healthier stream environment in restored sections of Greensboro urban streams, and which restoration structure(s) within restored streams–

riffle, cross vane, or step pool were most beneficial. We hypothesize that the restored streams will have better water quality parameters (pH, dissolved oxygen, and in-stream temperature) compared to unrestored streams as a result of the restoration to mimic a pre-disturbed state. The restored streams had several in-stream structures installed, which provide more area for algal accumulation and habitat for biota. Thus, there should be more algal accrual, higher nutrient uptake, and a better NCBI score in the restored streams compared to the unrestored streams. We hypothesize the riffle structure will be superior in water quality, have more algal accumulation, higher nutrient uptake, and better NCBI score compared to the cross vane and step pool structures.

CHAPTER II

METHODS

Study Sites

The study sites included stream reaches in Benbow, Brown Bark, Spring Valley, O' Henry, Rolling Roads, and Shannon Woods (Figure 1). All sites were located in urban parks in Greensboro, North Carolina. Greensboro encompasses 2000 square miles, with a population of 223,891 (US Census Bureau 2000). Single family homes surrounded the study sites. Non-point contamination from urban runoff impacted these streams. The sites were headwater streams which were part of the Buffalo Creek Watershed. North and South Buffalo creeks were given a poor water quality rating based on the EPT metric completed by the Division of Water Quality in 2003 (NCDENR 2004).

Each restored site, corresponding to riffle, cross vane and step pool, were within reaches of each restored stream- Benbow (BB), Brown Bark (BK), and Spring Valley (SV). The restoration projects, which were mitigation for highway construction, were completed in 2004, and included revitalization of the riparian area, bank stabilization, and installation of rock structures to mimic natural streams. Three replicate structures (riffles, cross vanes, and step pools) examined in this study were used repeatedly during the renovation projects and were located within each of the three restored streams. Within each restored stream, there were three 180m study reaches (Figure 2). The three study reaches in each restored stream were chosen based on a 30 m reach, at a minimum, that was influenced by riffle, cross vane, or step pool structure series. Downstream of 30 meters included a variety of in-stream restoration structures which influenced nutrient uptake if not removed within 30 meters (distance including the structure series of interest), from the $^{15}N\text{-}NH_4Cl$ injection site (Table 1). To allow for mixing within each structure series, the first sampling site (10 meters) was 1.5 meters after the beginning of each structure series. The injection site (0 meters) was 10 meters upstream from the first sampling site. All other sampling, water quality, epilithon, and macroinvertebrates, were completed within the structure series of interest (riffle, cross vane and step pool) in each of the restored streams.

The three unrestored streams were O'Henry (OH), Rolling Roads (RR), and Shannon Woods (SW) (Figures 5-7). No structures were installed in these streams. 180m reaches of the unrestored streams included in the study had protected riparian areas. These riparian areas were not reinforced or planted, but allowed to grow naturally. The injection sites (0 meter) in each stream were chosen based on access to the stream.

Site description measurements of streambed material, bank incision, riparian conditions, and canopy cover were taken in the summer of 2006 for all 12 study sites (3 riffles, 3 cross vanes, 3 step pools, and 3 unrestored) according to the LINX II Protocol (2004). USGS guidelines (2006) were followed when determining streambed material size. The height of the channel incision was measured at 2.5 and 5 meters into the riparian area from the edge of the wetted channel. Two by five meter blocks of the riparian area were evaluated from the wetted channel edge extending 25 meters on both sides of the stream at sample locations along the stream transect (0, 10, 20, 30, 80, 130, and 180 meters). The riparian area was assessed for condition (stable or unstable), vegetation type, and width of continuous vegetation before being interrupted by modified vegetation (mowed grass), buildings, or invasive species (only at Benbow- Japanese grass *Microstegium vimineum*). Riparian vegetation type was categorized as being forested (large trees with little understory), shrubs (small trees, shrubs, and grasses), mowed (cut grass), bare (no vegetation), rock (including boulders and rip rap), or paved (any impervious surfaces including rooftops). Canopy cover was measured using a densiometer (Forest Densiometers, Bartlesville, OK).

Preliminary analysis of sites revealed that streambeds in restored sites tended to have a greater percentage of larger substrates compared to unrestored sites. Almost 50% of the streambed in unrestored sites was sand/silt (Table 2). Riffles tended to have a larger percentage of boulders compared to all other sites. The mean bank incision at unrestored sites was higher, 1.5 to 2 m, compared to restored sites, 0.8 to 1.2 m (Table 2). The riparian area at the restored sites had more shrubs, 34%, compared to 21% at unrestored sites. Over half of the riparian area at unrestored sites was mowed grass (Table 2). Shannon Woods had the greatest percentage of canopy cover, 89%, compared to the rest of the sites (Table 2).

Sampling Period

All sampling took place during the summer of 2006, which was two years after the streams were restored. Bricks were installed within each restoration structure and at unrestored sites for epilithic biomass accrual during the third week in June. The ^{15}N-NH_4Cl pulse releases occurred from July 12 to August 1. Macroinvertebrates were collected during August. The weekly mean rain fall for North and South Buffalo Creeks during the experimental period was 0.9 cm ± 0.3 (USGS, 2006). The weekly mean low temperature for Greensboro, NC was 21 °C ± 0.7 and weekly mean high temperature was 31 °C ± 1.1 during the experimental period (National Weather Service, 2006).

Water Quality Parameters

Weekly measurements of pH, dissolved oxygen and in-stream temperature were taken using a YSI (YSI 6600, Yellow Springs, OH) over a four week period. Three measurements were taken per study site. At the restored study sites, measurements were taken within the riffle, cross vane, and step pool structure series. At the unrestored study sites, measurements were taken at 0, 80 and 180 meters.

Epilithic Biomass Procedures

The bricks were incubated for three to four weeks to allow for epilithic biomass formation. Epilithic biomass samples were scraped with a bristle brush from a known area, using an 8.05 cm^2 slide template, from natural rock substrata and incubated bricks. Three samples were collected from brick and natural rock substrata within each study reach at 10, 20, and 30 meters. Epilithic biomass slurries were put in pre-acid washed bottles, wrapped in aluminum foil and put on ice, then filtered immediately in low light conditions at the lab.

For the chlorophyll *a* analysis, the slurries were filtered onto precombusted Whatman GF/F glass fiber filters in low light conditions. The filters were immediately placed in 10 mL of 95% ethanol, re-wrapped in aluminum foil, and placed in the freezer until analyzed. The hot ethanol extraction technique was utilized to remove the chlorophyll *a* from the filters (Sartory and Grobbelaar 1984 as cited in the LINX II Protocol, 2004). The hot ethanol extraction technique does not require grinding to extract the chlorophyll *a* and does not require possible exposure to toxic methanol or acetone (LINX II Protocol 2004). The chlorophyll *a* concentration was analyzed using a spectrophotometer (Genesys ™ 10 Series Thermo Spectronic, Rochester, NY).

For ash-free dry mass (AFDM), the slurries were filtered onto pre-weighed, precombusted filters and dried as described above. The filters were re-weighed to determine dry mass. After the dry mass was determined, the filters were ashed at 500 °C for two hours, and then placed back into the dessicator for 24 hours. The filters were re-weighed to determine AFDM.

Nutrient Uptake Procedures

NH_4^+ has a faster uptake rate compared to nitrate, so a $^{15}N\text{-}NH_4Cl$ addition was used to examine nutrient uptake (Peterson et al. 2001). Initial ammonium concentrations were measured to determine the amount of 99% $^{15}N\text{-}NH_4Cl$ needed to enrich each site to about 100 ‰ of pre-enrichment level. Six water samples were collected per stream. The water was sampled using a syringe with a Whatman GF/F precombusted filter attached, then placed in acid washed bottles and put on ice until arrival at the lab. The water samples were frozen until analyzed. Duplicate assays were performed for each of the water samples following standard ammonium testing techniques at 640nm on a spectrophotometer (Genesys ™ 10 Series Thermo Spectronic, Rochester, NY) (Steve Whalen, UNC-Chapel Hill personal communication). The average of two determinations for all six samples per stream was used to estimate initial ammonium concentration per stream (Appendix B).

The amount of 99% ^{15}N-NH_4Cl needed for 100 ‰ enrichment above pre-enrichment level was calculated for each site using equations according to Hershey et al. (2006). Discharge measurements for each study site used to calculate needed amount of 99% ^{15}N-NH_4Cl were taken within 3 days of the nutrient release at 0, 80, and 180 meters.

A pulse addition of 99% ^{15}N-NH_4Cl was released for approximately four hours at 0m at each study site. The required amount of 99% ^{15}N-NH_4Cl was mixed on site in pre-acid washed containers of distilled water. The mixture was added continuously into the stream via pump (Master Flex, Niles, IL) operated by a car battery. Since the first sample site was 10 meters downstream, artificial mixing was induced at the injection site using a small, battery operated trolling motor. Pre and post ^{15}N periphyton samples were collected by scrubbing epilithic biomass from natural rock substrata, and then placed in acid washed bottles. The rocks were located at 0, 10, 20, 30, 80, 130, and 180 meters.

Samples were taken just prior and immediately after the ^{15}N-NH_4Cl addition. To avoid possible contamination, between more enriched upstream and less enriched downstream sites, post injection samples were collected beginning at 180 meters and working upstream. All samples were placed on ice immediately and filtered in the lab within 4 hours. The filters were dried at 60 °C for 48 hours, and then placed in the desiccator for 24 hours. The filters were analyzed for $\delta\,^{15}N$ at the UC Davis Stable Isotope Lab in Davis, California.

The $\delta^{15}N$ value is expressed as $[(R_{sample}/R_{standard})-1] \times 10^3$, where R is the ratio of $^{15}N/^{14}N$ and the standard is atmospheric nitrogen, N_2, reported in ‰ (Peterson and Fry 1987). The post $\delta^{15}N$ values were background corrected for natural abundance of ^{15}N by subtracting the pre $\delta^{15}N$ values. The natural log of the background corrected $\delta^{15}N$ values were plotted verses distance downstream (10, 20, 30 80, 130, 180 m). The slope (k) is the distance specific uptake rate (m^{-1}). Uptake length (S_w) is the inverse of the distance specific uptake rate ($S_w = 1/k$). The uptake velocity (v_f) is calculated as $(u*h)/S_w$, where u is water velocity and h is depth. Uptake rate (U) is calculated as v_f*C, where C is solute concentration (Stream Solute Workshop 1990). $\delta\ ^{15}N$ uptake per algal unit was determined as the inverse of the slope of the regression of the tracer $^{15}N\text{-}NH_4^+$ flux on distance per unit biomass (Hamilton et al. 2001).

Macroinvertebrate Procedures

Macroinvertebrates were gathered using kick-net, sweep net, leaf pack collection, and rock scrapings according to techniques of Eaton and Lenat (1991). In sections where rocks were not available, the stovepipe collection technique was substituted for rock scrapings. Two people sampled the 10 to 30 meter reach for each study site for a timed period of 1.5 hours. All samples were sorted in the field and placed in 95% ethanol. Sorted samples were identified to the lowest possible taxonomic level. The North Carolina Biotic Index (NCBI) was used to rate water quality of the study sites based on the macroinvertebrates collected.

Macroinvertebrate species were assigned scores based on pollution tolerance ranging from 0-10, with 0 being most intolerant species according to Lenat (1993). The number of individuals per species collected was incorporated into the index: rare species given a score of 1 (1-2 individuals per site), common species given a score of 3 (3-9 individuals per site), and abundant species given a score of 10 ($\geq$ 10 individuals per site):

$$\text{NCBI} = \frac{\text{Sum } TV_i N_i}{\text{Total } N}$$

where TV_i is the tolerance value of the *i*th taxa, N_i is the abundance value of the *i*th taxa, and Total N is the sum of all abundance values for every species in the sample. The final NCBI value was compared to the specific region value and a water quality class was assigned (Lenat 1993). The water quality classes are as follows: Excellent (5), Good (4), Good-Fair (3), Fair (2), and Poor (1) (NCDENR 2003).

Statistical Analysis

For each week, one-way ANOVAs, with restoration as a factor, were used to determine significant differences for each water quality parameter and discharge rate between unrestored and restored streams ($\mu = \beta_0 + \beta_{1\text{restored}}$). Two-way ANOVAs, with week and restoration as factors, were used to determine differences over the entire sampling period for temperature and dissolved oxygen between unrestored and restored sites ($\mu = \beta_0 + \beta_{1\text{restorated}} + \beta_{2\text{week}}$). When the one-way ANOVA indicated significant differences for each week, two-way ANOVAs, with restoration treatment and stream as factors, were used to determine significant differences for each water quality parameter between unrestored sites and restoration structure types (riffle, cross vane, and step pool) in the restored sites ($\mu = \beta_0 + \beta_{1\text{restoration treatment}} + \beta_{2\text{stream}}$). When the two-way ANOVA indicated significant differences, Tukey's HSD or Tamhane pairwise comparisons (based on equality of variances) were used to determine significant differences between unrestored sites and structure types.

One-way ANOVAs, with restoration as a factor, were used to determine differences for brick AFDM, brick and rock chlorophyll *a*, uptake metrics (uptake length, velocity, and rate and uptake rate per chlorophyll *a*), and NCBI score between unrestored and restored sites. To normalize variances, AFDM and brick and rock chlorophyll *a* data were log transformed. Uptake length, velocity and rate data were log transformed to pass Levene's equality of variances. Due to interference from a neighborhood pet in the stream during the $^{15}N\text{-}NH_4Cl$ pulse, Brown Bark riffle was excluded from the analysis.

A two-way ANOVA, with restoration treatment and stream as factors, was used to determine differences for brick AFDM, brick and rock chlorophyll *a*, uptake parameters and NCBI scores between unrestored sites and structure types in the restored sites. To normalize variances, AFDM and brick and rock chlorophyll *a* data were log transformed. Uptake velocity and rate data were log transformed to pass Levene's equality of variances. When the ANOVA indicated significant differences, Tukey's HSD pairwise comparison was used to determine significant differences between unrestored sites and structure types.

CHAPTER III

RESULTS

Water Quality

Across streams, the unrestored sites had lower mean pH values in week 1 compared to all structures in restored streams ($F_{7,28}$=40.6, p-value$\leq$ 0.001, Table 3). There were no significant differences for weeks 2-4. There were no significant differences in pH between structure types in restored streams.

Across the sampling period, the mean DO of restored sites was 2.7 mg/L higher than unrestored sites ($F_{2,139}$=27.8, p-value$\leq$ 0.001). Across streams, all restoration structures had a higher DO level compared to unrestored sites (Table 3). For weeks 1 and 2, the difference was statistically significant ($F_{7,4}$=8.4, p-value$\leq$ 0.001 and $F_{7,4=}$88.5, p-value$\leq$ 0.001 respectively) between unrestored sites and restoration structures, but not for weeks 3 and 4. There were no significant differences in DO between structure types in restored streams.

Across the sampling period, the temperature in restored sites was 1.8 °C higher compared to unrestored sites ($F_{2,139}$=64.7, p-value$\leq$ 0.001). During week 2, the temperature difference was statistically significant between unrestored sites and restoration structures ($F_{7,4}$=18.4, p-value $\leq$ 0.001). Step pools were significantly warmer than the unrestored sites during weeks 2 and 4 (p-value$\leq$0.004). During week 4, the temperature difference was statistically significant between unrestored sites and among restoration structures (F7,4=11.3, p-value$\leq$ 0.001). During week 4, riffles were significantly warmer than the unrestored sites, and step pools were significantly warmer than cross vanes (p-value$\leq$ 0.002, Table 3). There were no significant differences in temperature for weeks 1 and 3.

Epilithic Biomass

There was no statistical difference in epilithic biomass as AFDM accumulation on bricks between the unrestored and restored sites, 0.58±.08 and 0.66±.06mg m^{-2} respectively ($F_{1,106}$=0.063, p-value= 0.802). There were significant differences between restoration structures ($F_{7, 100}$=4.3, p-value$\leq$0.001). Within the restored streams, step pools and riffles had about 2 times more AFDM accumulation compared to cross vanes (p-value$\leq$ 0.004, Figure 3).

The restored sites had about 3 times more chlorophyll *a* on the bricks during the 3-4 week incubation period compared to unrestored sites, 7.6±2.2 and 2.1±.9 respectively ($F_{1,106}$= 11.2, p-value= 0.001). There were significant differences between unrestored sites and restoration structures ($F_{7,100}$=2.3, p-value=0.022). Riffles and step pools had about 3 times more chlorophyll *a* on the bricks compared to unrestored sites (p-value$\leq$ 0.034, Figure 4). There was a trend suggesting that cross vanes had more chlorophyll *a* on the bricks compared to the unrestored sites, although the difference was not statistically significant (p-value= 0.065).

The restored sites had 4.6 times more chlorophyll *a* on rock substrates compared to unrestored sites, 14.3±6.6 and 1.8±.8 mg m^{-2} respectively ($F_{1,106}$=18.05, p-value< 0.001). There were significant differences between unrestored sites and restoration structures ($F_{7,100}$=4.7, p-value$\leq$ 0.001). Within restored streams, rocks in cross vanes and step pools supported about 8 and 5 times more chlorophyll *a*, respectively, than rocks in unrestored sites (p-value$\leq$ 0.002), but riffle rocks had similar chlorophyll *a* to rocks in unrestored sites (p-value=0.12, Figure 4). There was also a trend toward more chlorophyll *a* on rocks in cross vanes than in riffles (p-value= 0.059).

Nutrient Uptake

Initial ammonium concentrations were high in all streams, ranging from 63.7 to 112.3 μg/L in unrestored streams and from 78.2 to 217.2 μg/L in restored streams (Table 4).

Uptake of ^{15}N by periphyton (background corrected) varied among restoration structures and unrestored sites. Riffles and step pools, with the exception of Benbow step pool, tended to have the highest $\delta\ ^{15}N$ periphyton at 10m with an exponential decline downstream. Peak $\delta\ ^{15}N$ periphyton at cross vane sites was variable ranging from 30 to 80 meters, with an exponential decline downstream. In two of the unrestored sites, Rolling Roads and Shannon Woods, uptake of ^{15}N followed the same pattern; $\delta\ ^{15}N$ periphyton peaked at 30 meters with an exponential decline further downstream. $\delta\ ^{15}N$ periphyton at O'Henry, an unrestored stream, was the most variable of all the sites (Figure 5).

Unrestored sites had an uptake length (S_w) that was about 3 times longer, 197±118 meters, than that of restored sites, 70±28 meters. Unrestored sites had 127 m longer uptake length compared to riffles, 117 m longer compared to cross vanes, and 135 m longer compared to step pools ($F_{7,3}$=39.5, Table 5). Uptake length between structure types was not statistically different. Uptake length does not take into account discharge differences between streams. However, over the study period, there were no statistical differences in discharge between unrestored and restored sites (Figure 6).

There was a trend indicating restored sites had 0.7 m/h faster uptake velocity (v_f) compared to unrestored sites ($F_{1,9}$=4.3, p-value= 0.069). Mean uptake velocity for restored sites was 1.1±.2 m/hr compared to 0.4±.3 m/hr at unrestored sites (Table 5). There was no significant difference for uptake velocity between restoration structures.

There was a trend suggesting that restored sites had an uptake rate (U) that was about 7 times faster compared to unrestored sites, 25.1±8.2 and 3.5±2.1 mg m^{-2} min^{-1} respectively ($F_{1,9}$=4.9, p-value= 0.055). However, uptake rate was highly variable between restoration structures, and did not differ significantly. There was no significant difference for uptake rate per chlorophyll *a* between unrestored and restored sites ($F_{1,9}$=0.3, p-value=0.596) or restoration structures ($F_{4,3}$=1.2, p-value=0.473, Table 5).

Macroinvertebrates

There was no statistical difference in NCBI scores between unrestored and restored sites (p-value=0.803) or among restoration structures ($F_{7,4}$=1.6, p-value= 0.337). The average NCBI score for unrestored sites ranged from 1 to 1.6, which indicates Poor to Fair water quality. The average NCBI score for restored sites was 2, corresponding to a quality rating of Fair. Riffles were the only structure that did not receive a water quality rating of Poor at any site, and one of the riffles had a rating of Good-Fair. Among unrestored sites, cross vanes, and step pools, at least one of the replicates received a water quality rating of Poor, and none of the ratings were above Fair (Table 6).

Riffles had the least percentage of Chironomini species collected compared to the other sites. Chironomini species accounted for 12% (48 of 384 individuals) of collector-gatherers in riffles, 57% (139 of 246 individuals) in cross vanes, 54% (226 of 417 individuals) in step pools, and 18% (27 of 154 individuals) in unrestored sites (Appendix C).

CHAPTER IV

DISCUSSION

Algal abundance and water quality parameters were important to evaluate because they potentially affected nutrient uptake parameters and macroinvertebrate species in this study. The installed restoration structures provided hard substrate for algal growth, which positively affected nutrient uptake parameters in restored streams. All study sites were within tolerable ranges, regarding pH, DO, and in-stream temperature, to support native macroinvertebrate species, but most sites were given Poor or Fair water quality ratings based on the NCBI. Two years after restoration, improvements were seen, but changes could be made to reduce nutrient loading and improve the stream habitat for intolerant macroinvertebrate species.

Effect of Restoration on Epilithic and Algal Biomass

The similar accumulation of epilithon, measured as AFDM, at all sites could best be attributed to high nutrient concentrations (Taylor et al. 2004). Lynam (2004) found that headwater streams in Greensboro had elevated levels of NO_3^- and PO_4^{3-}. This study revealed all sites had high NH_4^+ levels. Therefore, the similar accumulation of benthic epilithon in all unrestored and restored sites most likely resulted from high nutrient concentrations in all streams; it is unlikely that nutrient abundance limited algal growth (Stelzer and Lamberti 2001).

Within restored streams, riffles and step pools accumulated more epilithic biomass on the incubated bricks, measured as AFDM, compared to cross vanes. Some of the major differences between structures were percent canopy cover, percentage of fine streambed material, and current speed. It is unlikely that canopy cover was a factor because epilithon accumulation was not correlated to canopy cover (p-value= 0.954), and there was more epilithon accumulation in riffles despite the 20% greater canopy cover over riffle habitats. Almost 50% of the streambed composition in cross vanes was sand/silt, compared to 27% in riffles and 14% in step pools. Cross vanes had the slowest mean current of all the restoration structures. Incubated substrates with less sediment cover allow for more epilithon accrual (Pringle et al. 1993). The largest percentage of fine streambed material combined with the slowest current in cross vanes probably resulted in the bricks being partially covered. Thus, there was less surface area for epilithon accrual over the incubation period in cross vanes compared to riffles and step pools.

Accrual of algae, measured as chlorophyll *a*, on bricks was greater in restoration structures than in unrestored streams, which probably resulted from less canopy cover. Amount of benthic light is a contributing factor to benthic algal growth (Vannote et al. 1980, Quinn et al. 1997, and Bis et al. 2000). Restored sites averaged about 27% less canopy cover than unrestored sites. Therefore, more benthic light in restored streams partially explains the higher chlorophyll *a* biomass in the epilithon at restored sites.

Within restored streams, riffles had the highest mean algal growth on bricks compared to the other structures despite the highest percentage of canopy cover of the structures, although this difference was not statistically significant (Figure 9). Busse et al. (2006) found that benthic algal biomass was positively correlated to current speed. Riffles had the fastest current, 0.114 m/sec, compared to cross vanes and step pools, 0.097 and 0.108 m/sec respectively. Thus, the faster current in riffles very likely had offset the negative effect of greater canopy on algal accumulation.

Restored streams had significantly more algae on rocks than unrestored streams, especially in cross vanes and step pools, but not riffles. Although algal growth on bricks in riffle habitats was significantly higher than unrestored streams, there was no significant difference in algal abundance on rocks between riffles and unrestored sites (p-value= 0.12). Riffles had 13% less canopy cover than unrestored sites, but cross vanes and step pools had at least 30% less canopy cover than unrestored sites. Therefore, the significantly higher algal abundance on rocks in cross vanes and step pools compared to riffles and unrestored sites probably resulted from less canopy cover.

Effect of Restoration on Nutrient Uptake

All the urban streams in this study had high NH_4^+ levels, which is typical of urban streams (Brett et al. 2005, Groffman et al. 2005, and Meyer et al. 2005). All study sites were upstream from the waste water treatment plant, so treated waste water can not be implicated. However, stormwater pipes drained directly into all streams from surrounding single family homes. Thus, fertilized lawns and pet waste were the most likely factors contributing to the high NH_4^+-N levels in the study streams.

Unrestored sites had a ^{15}N uptake length that was approximately 2.5 to 3 times longer compared to the restoration structures, probably due to less hard substrate for periphyton. Uptake length is influenced by algal biomass and percent surface area covered by algae (Sabater et al. 2000), stream discharge (Hall et al. 2002), and ammonium concentrations (Webster et al. 2003). Stream discharge was not significantly different between study sites (p-value= 0.157), and all sites had similarly high NH_4^+-N concentrations. Therefore, discharge and NH_4^+-N concentration did not influence differences in uptake length seen in this study. Almost 50% of the streambed composition at unrestored sites was sand/silt, compared to less than 40% at restored sites. Therefore, it is very likely that the higher percentage of hard substrates and more algal accumulation per unit area in restored sites allowed for shorter uptake lengths.

Within restored streams, uptake length was not statistically different between restoration structures, suggesting all restoration structures are approximately equally efficient at removing nutrients from the water column. Removing nutrients from the water column is important in urban headwater streams because of high nutrient loading and concentration in urban streams (Brett et al. 2005, Groffman et al. 2005, and Meyer et al. 2005). If there is greater uptake, there may also be more denitrification. Lofton et al (2007) found that denitrification rate could be significant relative to loading at one urban stream site. Restored streams removed nutrients from the water column faster, potentially reducing nutrient export (Warren et al. 2007), which would be beneficial in reducing negative impacts of eutrophication to downstream reaches (Wollheim et al. 2005).

Uptake velocity in urban restored streams was not only faster than the unrestored sites, but was also faster than forested tributary streams (Meyer et al. 2005), probably due to installation of hard substrates for periphyton accumulation. Uptake velocity is not affected by differences in stream discharge (Webster et al. 2003), so it is a useful metric to compare streams to one another. Uptake velocity in unrestored sites was comparable to that observed in urban tributary streams in Georgia (Meyer et al. 2005). The uptake velocities of the urban restored sites in my study (v_f=1-1.3 m/hr) exceeded urban and agricultural influenced streams in the Midwestern US, ($v_f \leq$ 0.198 m/hr) (Bernot et al. 2006), restored headwater streams in Georgia (restored by adding coarse woody debris) (v_f=0.05-0.11 m/hr) (Roberts et al. 2007), and forested tributary streams in Georgia (v_f =0.36-0.58 m/hr) (Meyer et al. 2005). Hence, the restoration structures appear to provide more substrate for algal growth, which allows for faster removal of nutrients from the water column.

Due to higher uptake rates in cross vane and step pool structures, there was a trend indicating uptake rate was higher in restored sites than unrestored sites (p-value=0.055). However, uptake rate for riffles was very similar to unrestored sites, 5.3 and 3.5 mg m^{-2} min^{-1} respectively. Transient storage can increase nutrient uptake rates (Ensign and Doyle 2005, Thomas et al. 2003). Both cross vane and step pool structures create pool habitats, while riffles are very shallow. Nutrient molecules would be expected to take longer and/or complete more cycles while moving through a pool habitat, thereby increasing transient storage time. Unrestored sites did have some pool habitats, but lacked hard substrates for algal accumulation within these pools. Nutrient uptake rates are also influenced by periphyton (Parkyn et al. 2005). Cross vanes and step pools had significantly more algae than riffles and unrestored sites. Therefore, higher uptake rates in cross vanes and step pools most likely resulted from a combination of increased transient storage time and more algae.

The same trend was not seen for uptake rate per chlorophyll *a*. There were no significant differences in uptake rate/chlorophyll *a* between unrestored sites and any of the restoration structures. Despite similar accumulation of algae in both cross vane and step pool structures, step pools had the highest mean uptake rate per chlorophyll *a* over all structures, even though this difference was not statistically significant. Step pools are completely enclosed pools (square rock structure) with large substrates on the bottom of the pool, while cross vanes are not enclosed. The design of step pools could increase the transient storage time of a nutrient molecule, which might be reflected in increased uptake rates per chlorophyll *a* unit. The step pool structure at Spring Valley had the highest uptake rate per chlorophyll *a*, 32.28 $mg/m^2/min$, compared to the other step pools, 0.34-6.58 $mg/m^2/min$. Spring Valley had the largest, deepest pool with the largest boulders on the pool bottom compared to other step pools in this study. In enclosed larger, deeper pools, nutrient molecules would be expected to travel across more slowly and/or complete more cycles. Hence, step pools structures, designed similar to the step pool at Spring Valley, could increase transient storage time allowing for higher nutrient uptake rates per chlorophyll *a* unit. Installing more of these types of structures in urban areas, where nutrient concentrations are already elevated, and step pools are deemed essential for grade control, could benefit downstream reaches because it has the potential to remove more nutrients from the water column.

Effect of Restoration on Water Quality

Unrestored sites had significantly cooler in-stream temperatures during weeks 2 and 4, but the small difference, not seen during other weeks, was most likely due to differences in canopy cover. Solar and thermal radiation, riparian shading, local air temperature, and groundwater inflow are some of the factors that can affect in-stream temperature (Gravelle and Link 2007). Since all temperature measurements each week were taken on the same day and within two hours, local air temperature was not a factor in this study. All sites were located in the same watershed and in close proximity to each other, so it is unlikely groundwater inflow was a significant factor. However, unrestored sites did have 20% more canopy cover compared to restored sites. Thus, cooler temperatures at unrestored sites most likely resulted from more riparian shading.

The observed higher DO during weeks 1 and 2 at restored sites can best be attributed to greater turbulence associated with the restoration structures. DO in streams can be affected by many factors, including stream temperature and mixing across the air-water interface (Michigan Department of Environmental Quality 2007). Although oxygen is more soluble in colder water, colder temperature cannot explain the observed higher DO in restored sites because these sites were slightly warmer. Furthermore, lack of significant differences between weeks 3-4 may have reflected an effect of warmer temperatures in restored sites offsetting beneficial effect of turbulence. This study revealed there were no statistical differences in DO between restoration structures suggesting all structures were comparably efficient at creating turbulence. Pretty et al. (2003) found similar results concerning DO levels between riffles and deflectors (which create pool habitats). Despite the differences among site types, all sites had DO levels above the 5-6 mg/L needed to support healthy communities (Michigan Department of Environmental Quality 2007).

Although there was a significant difference in pH between restored and unrestored sites, the difference was small and not biologically significant. All sites were within tolerance levels for invertebrate and fish species native to the region. Across all weeks, the restored sites had a slightly higher pH compared to unrestored sites, which may have been due to greater algal production (Wetzel et al. 1985). However, the difference was small and only statistically significant for week 1 between unrestored sites and the restoration structures (p-value ≤ 0.009). Griffiths (1992) found that at pH 5.9, there was no significant decline in macroinvertebrate species. Thus, it is unlikely that pH could have contributed to the observed rating of Fair for invertebrate communities at most sites.

Effect of Restoration on Macroinvertebrates

While water quality ratings were not significantly different between unrestored and restored sites, riffles did receive better ratings compared to all sites. None of the riffle structures received a rating below Fair, and one received a rating of Good-Fair. Cross vane and step pool ratings were very similar to unrestored sites. These results are similar to previous studies. Harrison et al. (2004) found there was slightly more macroinvertebrate diversity in riffles than in pool habitats created by deflectors in restored streams. Northington and Hershey (2006) found that there was a trend indicating macroinvertebrate abundance and richness was better in restored streams compared to unrestored streams in NC. The restored urban streams in this study received a better water quality rating of Fair compared to a past study of NC urban streams, when they were rated as Poor (Lenat and Crawford 1994).

Even though riffles were in the same stream as cross vanes and step pools, riffle structures had the smallest percentage of Chironomini for all sites, which implies that riffles may better counteract the negative impacts of urbanization to streams than other types of structures. A higher percentage of Chironomini are found in areas with poor water quality due to organic pollution (Hellawell 1986 as cited in Harper et al. 1997), and are positively correlated with watershed imperviousness (Wang and Kanehl 2003). Therefore, installing more riffle structures in restored streams may promote the habitat necessary for macroinvertebrate species which are less tolerant to the negative effects of urbanization to streams.

CHAPTER V

CONCLUSIONS AND IMPLICATIONS

Overall, there were some consistent improvements to urban restored streams resulting from the installation of the restoration structures. The restored streams did have higher DO levels, probably due to greater turbulence associated with restoration structures. The structures provided hard substrate for algal growth, and positively affected nutrient uptake length. Nutrient uptake length was much shorter in restored streams. There was also a strong trend toward faster uptake velocity and greater uptake rate in restored sites compared to unrestored sites. Riffles appeared to be more beneficial than cross vanes and step pools. There was a trend indicating riffles allowed for more mean algal growth and, overall, showed a trend toward better water quality ratings based on NCBI. Thus, restoration did seem to offset some of the negative effects of urbanization to streams by creating a more oxygen rich environment and increasing nutrient removal.

Despite the benefits of the restoration, there was little improvement in biotic integrity at restored sites, based on NCBI, probably due to several factors associated with urban streams. The numbers of EPT taxa were very low, congruent with past studies of urban streams (Lenat and Crawford 1994, Wang and Kanehl 2003, and Suren and McMurtrie 2005). Plecoptera were absent and few Ephemeroptera and Trichoptera taxa were collected. Scraper density is negatively correlated with increasing urbanization (Roy et al. 2003), and no scrapers were collected in this study. Declines in the quality of macroinvertebrate species in the urban restored streams in this study could have resulted from a decline or absence of species not tolerant to water pollution resulting from high nutrients (Harrison et al. 2004), and less upstream macroinvertebrate colonization due to disconnectivity caused by culverts (Blakely et al. 2006). Thus, improvements are needed in restoration approaches if biotic integrity is to be enhanced.

From observations in this study, improvements to the standard method of restoring streams could be made. Cross vanes may be a useful structure as a channel constrictor, but making improvements to the design, such as adding large boulders in the pool, could be beneficial as habitat for macroinvertebrates and increase nutrient removal. Diverting runoff, which is high in inorganic nutrients, through the riparian areas instead of directly into streams could remove a large amount of nutrients before they reached the stream. This study was completed 2 years post restoration, so continual monitoring of restored streams would be beneficial to evaluate longer term effects of restoration. Furthermore, additional studies investigating transient storage time and nutrient retention of restoration structures would be valuable in determining effectiveness of various structures.

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APPENDIX A: Tables and Figures

Table 1. Description of the 180m study reaches within each restored stream, Benbow (BB), Brown Bark (BK) and Spring Valley (SV), and sampling stations for $^{15}N\text{-}NH_4Cl$ injections. The description only includes in-stream structures, some reaches of the streambed were not influenced by in-stream structures. Study reach indicates the distance downstream from the injection site where the structure listed in the description was located. Structure length indicates the total length of the study reach influenced by the structure listed in the description.

Site	Description	Study Reach (m)	Structure Length (m)
BB Riffle	Injection Site	0	
	Riffle	8.5	65
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Step Pool	78	79
	Sample Station 4	80	
	Sample Station 5	130	
	Concrete Culverts	166	13
	Sample Station 6	180	
BK Riffle	Injection Site	0	
	Riffle	8.5	31
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Sample Station 4	80	
	Boulder	92	7
	Sample Station 5	130	
	Boulder	161	10
	Sample Station 6	180	
SV Riffle	Injection Site	0	
	Riffle	8.5	55
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Sample Station 4	80	
	Step Pool	85	5
	Cross Vane	103	61
	Sample Station 5	130	
	Sample Station 6	180	

Table 1 Continued.

Site	Description	Study Reach (m)	Structure Length (m)
BB Cross Vane	Injection Site	0	
	Cross Vane	8.5	25
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Vane	34	27
	Sample Station 4	80	
	Riffle	104	7
	Vane	118	5
	Sample Station 5	130	
	Riffle	150	24
	Sample Station 6	180	
BK Cross Vane	Injection Site	0	
	Cross Vane	8.5	30
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Step Pool	40	5
	Cross Vane	56	4
	Step Pool	67	8
	Sample Station 4	80	
	Step Pool	86	7
	Riffle	106	31
	Sample Station 5	130	
	Sample Station 6	180	
SV Cross Vane	Injection Site	0	
	Cross Vane	8.5	61
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Sample Station 4	80	
	Step Pool	123	8
	Sample Station 5	130	
	Cross Vane	158	22
	Sample Station 6	180	

Table 1 Continued.

Site	Description	Study Reach (m)	Structure Length (m)
BB Step Pool			
	Injection Site	0	
	Step Pool	8.5	23
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Vane	32	8
	Cross Vane	42	20
	Vane	65	27
	Sample Station 4	80	
	Sample Station 5	130	
	Riffle	137	7
	Vane	150	5
	Sample Station 6	180	
BK Step Pool			
	Injection Site	0	
	Step Pool	8.5	26
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Riffle	47	31
	Sample Station 4	80	
	Sample Station 5	130	
	Boulder	131	7
	Sample Station 6	180	
SV Step Pool			
	Injection Site	0	
	Step Pool	8.5	32
	Sample Station 1	10	
	Sample Station 2	20	
	Sample Station 3	30	
	Cross Vane	44	20
	Riffle	66	7
	Sample Station 4	80	
	Cross Vane	83	56
	Sample Station 5	130	
	Step Pool	151	14
	Sample Station 6	180	

Table 2. Mean (± 1 SE) site description measurements for all 12 study sites and for all unrestored (UN) and all restored (RES) sites. Streambed material was gauged following USGS (2006) guidelines. Bank incision, is the height of the incision at 2.5 and 5 meters into the riparian area. Riparian condition was measured 25 m on both sides of the stream and was evaluated as being stable or unstable, vegetation type (see description in methods section), and width of natural vegetation before being interrupted by modified vegetation or structures. Percent canopy cover was measured according to LINX II Protocol (2004).

	Unrestored			Restored Riffle			Restored Cross Vane			Restored Step Pool				
Site Description	**OH**	**RR**	**SW**	**BB**	**BK**	**SV**	**BB**	**BK**	**SV**	**BB**	**BK**	**SV**	**UN**	**RES**
Streambed (%)														
Boulder (> 25cm)	10 ± .04	59 ± .09	12 ± .08	48 ± .10	27 ± .09	53 ± .08	19 ± .06	7 ± .02	17 ± .06	39 ± .10	8 ± .04	17 ± .08	27 ± 16	26 ± 6
Cobble (6.5-25 cm)	10 ± .04	25 ± .08	7 ± .05	3 ± .02	22 ± .09	19 ± .04	6 ± .02	22 ± .06	26 ± .08	9 ± .03	21 ± .08	11 ± .06	14 ± 6	15 ± 3
Gravel (.2-6.5 cm)	26 ± .08	2 ± .01	8 ± .04	11 ± .05	24 ± .06	11 ± .03	10 ± .04	32 ± .09	27 ± .06	15 ± .05	35 ± .07	17 ± .06	12 ± 7	20 ± 3
Sand (.1-.2 cm)	35 ± .08	8 ± .03	54 ± .09	21 ± .09	16 ± .05	13 ± .03	42 ± .06	22 ± .06	23 ± .07	31 ± .08	20 ± .07	17 ± .06	32 ± 13	23 ± 3
Silt (< .1 cm)	19 ± .07	6 ± .03	19 ± .06	17 ± .10	11 ± .05	4 ± .03	23 ± .06	17 ± .05	7 ± .05	6 ± .04	16 ± .08	38 ± .11	15 ± 4	16 ± 4
Bank Incision (m)														
2.5 m	1.7 ± .10	1.4 ± .63	1.3 ± .10	1.0 ± .07	0.5 ± .05	1.0 ± .07	0.8 ± .06	0.7 ± .08	1.1 ± .08	0.8 ± .10	0.7 ± .07	0.9 ± .08	1.5 ± .21	0.8 ± .03
5.0 m	2.3 ± .08	1.7 ± .64	1.9 ± .12	1.3 ± .08	1.0 ± .11	1.4 ± .08	1.1 ± .09	0.9 ± .08	1.4 ± .09	1.4 ± .15	1.2 ± .11	1.3 ± .06	2.0 ± .22	1.2 ± .03
Riparian Condition (%)														
Stable	91 ± 2	94 ± 2	89 ± 2	96 ± 2	98 ± 2	91 ± 2	96 ± 2	98 ± 2	89 ± 2	96 ± 2	95 ± 2	95 ± 2	91 ± 2	95 ± 1
Unstable	9 ± .7	6 ± .5	11 ± .8	4 ± 1	2 ± .5	9 ± 1	4 ± .5	2 ± .6	11 ± 2	4 ± .8	5 ± .8	5 ± .8	9 ± 2	5 ± 1
Riparian Vegetation (%)														
Forested	6 ± 3	1 ± 1	15 ± 4	20 ± 5	2 ± 1	36 ± 5	9 ± 3	0	27 ± 5	4 ± 2	1 ± 1	17 ± 4	7.5 ± 4	13 ± 4
Shrubs	17 ± 4	25 ± 5	21 ± 4	41 ± 6	45 ± 5	21 ± 4	34 ± 5	50 ± 5	16 ± 3	40 ± 5	41 ± 5	22 ± 4	21 ± 2	34 ± 4
Mowed	63 ± 5	47 ± 5	61 ± 6	28 ± 5	42 ± 5	24 ± 4	39 ± 5	32 ± 4	37 ± 5	34 ± 5	45 ± 5	52 ± 6	57 ± 5	37 ± 3
Bare	2 ± .9	10 ± 3	0.6 ± .6	0	1 ± .7	2 ± 1	1 ± 1	1 ± .5	1 ± .8	2 ± .7	1 ± .4	2 ± 1	4 ± 3	2 ± .4
Rock	1 ± .6	0	0.4 ± .4	0	0	1 ± .5	1 ± .5	0	2 ± .9	1 ± .5	2 ± 1	2 ± .7	0.5 ± .3	1 ± .3
Paved	11 ± 3	17 ± 4	2 ± 2	11 ± 3	10 ± 3	16 ± 4	16 ± 4	17 ± 4	17 ± 4	19 ± 4	10 ± 3	5 ± 2	10 ± 4	13 ± 2
Natural Veg. Width (m)	6 ± .7	7 ± .8	10 ± 2	11 ± 1	12 ± .9	7 ± 1	10 ± 2	12 ± 2	6 ± .4	11 ± .9	10 ± 1	8 ± 2	8 ± 1	10 ± .7
Canopy Cover (%)	47 ± 10	41 ± 9	89 ± 8	56 ± 6	34 ± 6	59 ± 7	22 ± 8	26 ± 4	56 ± 7	18 ± 6	35 ± 6	41 ± 10	59 ± 15	39 ± 5

Table 3. Weekly mean pH, dissolved oxygen (DO), and in-stream temperature (± 1 SE) for unrestored sites (UN), restored sites (RES), riffles, cross vanes, and step pools. For each week, one-way ANOVAs were used to determine significant differences between unrestored and restored sites (RES P-Value). When a two-way ANOVA indicated significant differences, Tukey's HSD or Tamhane multiple comparison tests (based on equality of variances) were used to determine significant differences between unrestored streams and restoration structures. Struc P-value indicates significant differences between unrestored sites and the restoration structures. Matching asterisks in the table correlates to the asterisk before the p-value to indicate differences between these sites only. NS denotes no significant differences between any of the sites.

Parameter	Res P-Value	RES	UN	Riffle	Cross Vane	Step Pool	Struc P-Value
Week 1							
pH	**0.025**	6.9 ±.07	6.7 ± .06	6.9 ± .11	6.9 ± .13	7.0 ± .08	**≤0.009**
DO (mg/l)	**0.000**	12.4 ±.27	9.8 ± .52	12.4 ± .63	12.5 ± .38	12.4 ± .23	**0.036**
Temperature (°C)	NS	22.2 ±.16	22.2 ± .46	22.1 ± .32	22.5 ± .23	22.0 ± .18	NS
Week 2							
pH	0.091	7.5 ±.06	7.2 ± .15	7.5 ± .06	7.5 ± .11	7.3 ± .09	NS
DO (mg/l)	**0.002**	17.7 ±.34	10.9 ± 1.53	17.6 ± .55	17.7 ± .75	18.0 ± .26	**≤0.013**
Temperature (°C)	**0.005**	28.4 ±.53	*25.4 ± .74	27.8 ± 1.03	28.2 ± 1.06	*29.3 ± .17	*** 0.004**
Week 3							
pH	NS	7.1 ±.07	7.2 ± .06	7.3 ± .08	7.1 ± .16	7.1 ± .09	NS
DO (mg/l)	NS	10.1 ±.23	9.7 ± .51	9.9 ± .38	10.5 ± .40	10.3 ± .38	NS
Temperature (°C)	NS	21.1 ±.23	21.9 ± .78	21.4 ± .29	21.1 ± .48	21.0 ± .42	NS
Week 4							
pH	NS	6.9 ±.07	6.8 ± .05	6.9 ± .09	6.9 ± .16	7.0 ±.09	NS
DO (mg/l)	NS	9.5 ±.29	8.7 ± .54	9.6 ± .29	9.2 ± .34	9.7 ±.76	NS
Temperature (°C)	0.054	27.1 ±.35	*/**25.4 ± 1.10	* 28.0 ± .46	*** 25.5 ±.35	**/***27.9 ±.60	* 0.001 ** 0.002 ***0.001

Table 4. Mean (±1SE) NH_4^+-N concentration of 2 determinations for each sample location within a site (total of six per site), were averaged for all 6 sample locations to calculate the initial NH_4^+-N concentration per stream. Restored streams are Benbow, Brown Bark, and Spring Valley. Unrestored streams are O'Henry, Rolling Roads, and Shannon Woods .

Site	**Initial NH_4^+-N Concentration (µg/L)**
Benbow	217.2 ± 19.7
Brown Bark	81.7 ± 2.5
Spring Valley	78.2 ± 3.4
O'Henry	63.7 ± 3.1
Rolling Roads	112.2 ± 1.2
Shannon Woods	87.9 ± 0.05

Table 5. Mean uptake length (S_w), uptake velocity (v_f), uptake rate (U), and uptake rate per chlorophyll *a* (U/chl *a*) (± 1 SE) for unrestored (UN) and restored (RES) streams, and for unrestored sites, riffles, cross vanes, and step pools. A one-way ANOVA was used to determine significant differences between UN and RES streams. A significant difference (p-value≤ 0.05) and trends (0.05<p-value<0.10) indicates differences between unrestored and restored streams. When the two-way ANOVA indicated a significant difference, Tukey's HSD multiple comparison test was used to find significant differences between unrestored sites and restoration structures. A significant difference (p-value≤ 0.05) indicates the unrestored sites are different from the restoration structures. NS denotes no significant difference between any of the sites.

Site	S_w (m)	p-value	v_f (m hr^{-1})	p-value	U (mg m^{-2} min^{-1})	p-value	U/chl *a* (mg m^{-2} min^{-1})	p-value
UN	197 ± 68		0.4 ± .26		3.5 ± 2.1		2.5 ± 1.4	
RES	70 ± 10	**0.021**	1.1 ± .18	0.069	25.1 ± 8.2	0.055	6.1 ± 3.8	NS
UN	197 ± 68 a		0.4 ± .26		3.5 ± 2.1		2.5 ± 1.4	
Riffle	69 ± 15 b		1.3 ± .01		5.3 ± 1.2		2.0 ± .1	
Cross Vane	80 ± 13 b		1.0 ± .51		22.3 ± 10.2		1.8 ± 1.3	
Step Pool	61 ± 24 b	**≤ 0.009**	1.1 ± .15	NS	41.2 ± 16.0	NS	13.1 ± 9.8	NS

Table 6. Biotic index (BI), North Carolina Biotic Index (NCBI), and water quality rating for each unrestored and restored site. Unrestored sites are O'Henry (OH), Rolling Roads (RR), and Shannon Woods (SW). Restored sites are Benbow (BB), Brown Bark (BK), and Spring Valley (SV). The mean BI for all unrestored (UN) and restored (RES) sites (± 1 SE) and correlating NCBI and water quality rating are listed at the bottom of the table. There were no significant differences between unrestored sites and restoration structures ($F_{7,4}$=1.614, p-value=0.337).

Site	BI	NCBI	Rating
OH	7.31	2	Fair
RR	8.17	1	Poor
SW	7.02	2	Fair
BB Riffle	7.11	2	Fair
BK Riffle	5.84	3	Good-Fair
SV Riffle	6.75	2	Fair
BB Cross Vane	8.23	1	Poor
BK Cross Vane	6.94	2	Fair
SV Cross Vane	7.40	2	Fair
BB Step Pool	7.86	1	Poor
BK Step Pool	7.79	1	Poor
SV Step Pool	6.95	2	Fair
UN	7.50 ± .4	1-1.6	Poor/Fair
RES	7.21 ± .2	2	Fair

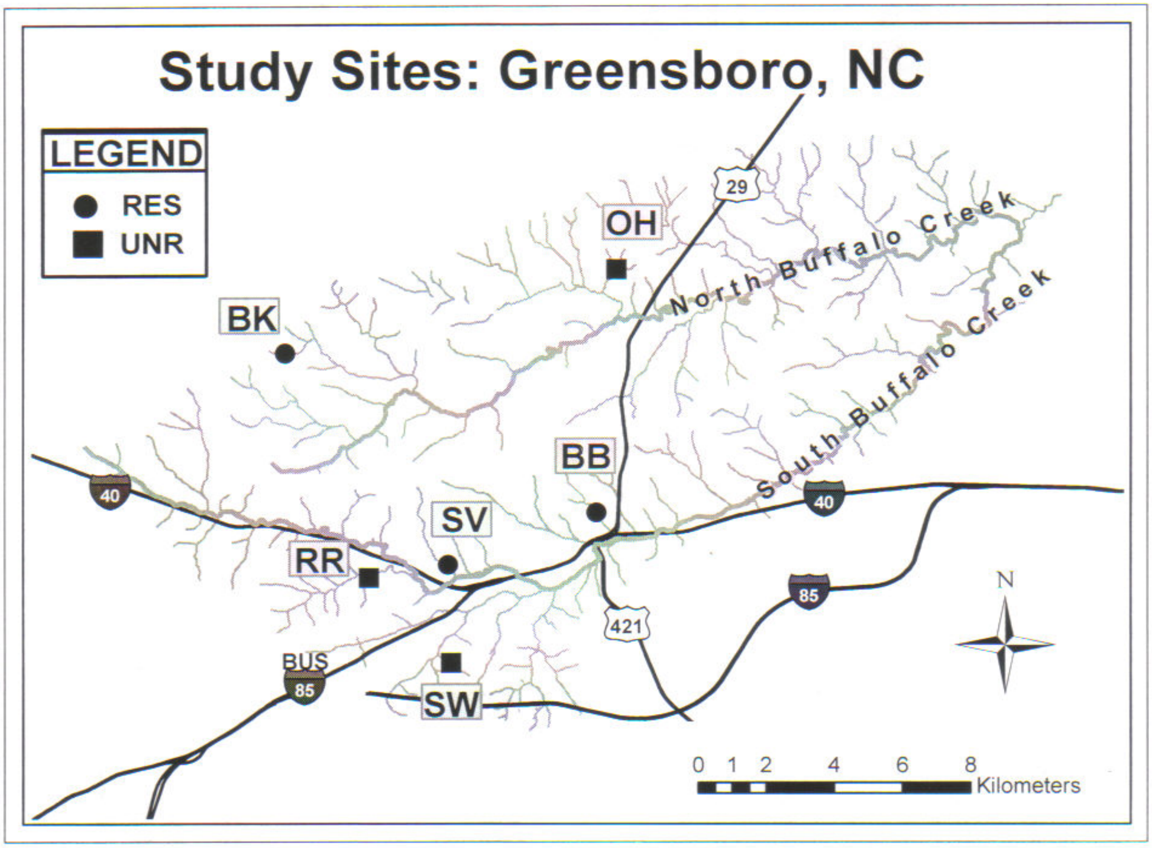


Figure 1. Map of all study sites. Restored (RES) streams are Benbow (BB), Brown Bark (BK), and Spring Valley (SV). Unrestored (UNR) streams are O'Henry (OH), Rolling Roads (RR), and Shannon Woods (SW).

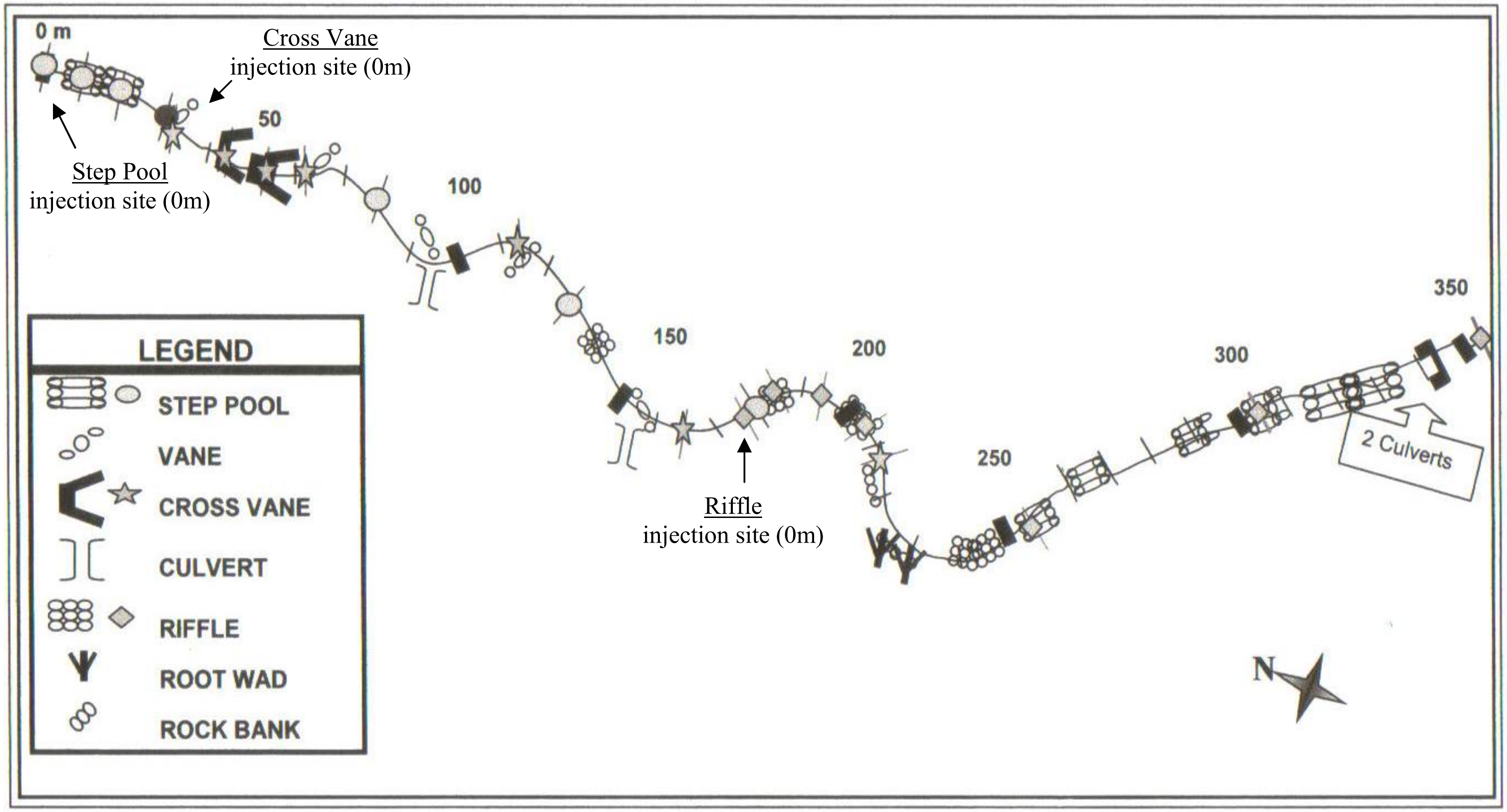


Figure 2. Map of Benbow, restored study site. The long thin lines with corresponding shapes (see map legend) represents the sampling sites for $^{15}N\text{-}NH_4Cl$ injection (0m- injection site: 10, 20, 30, 80, 130, and 180 meters were sampling sites). The bold lines are 50 meter increments. 3 in-stream structures- riffle, cross vane, and step pool- were sampled during this experiment.

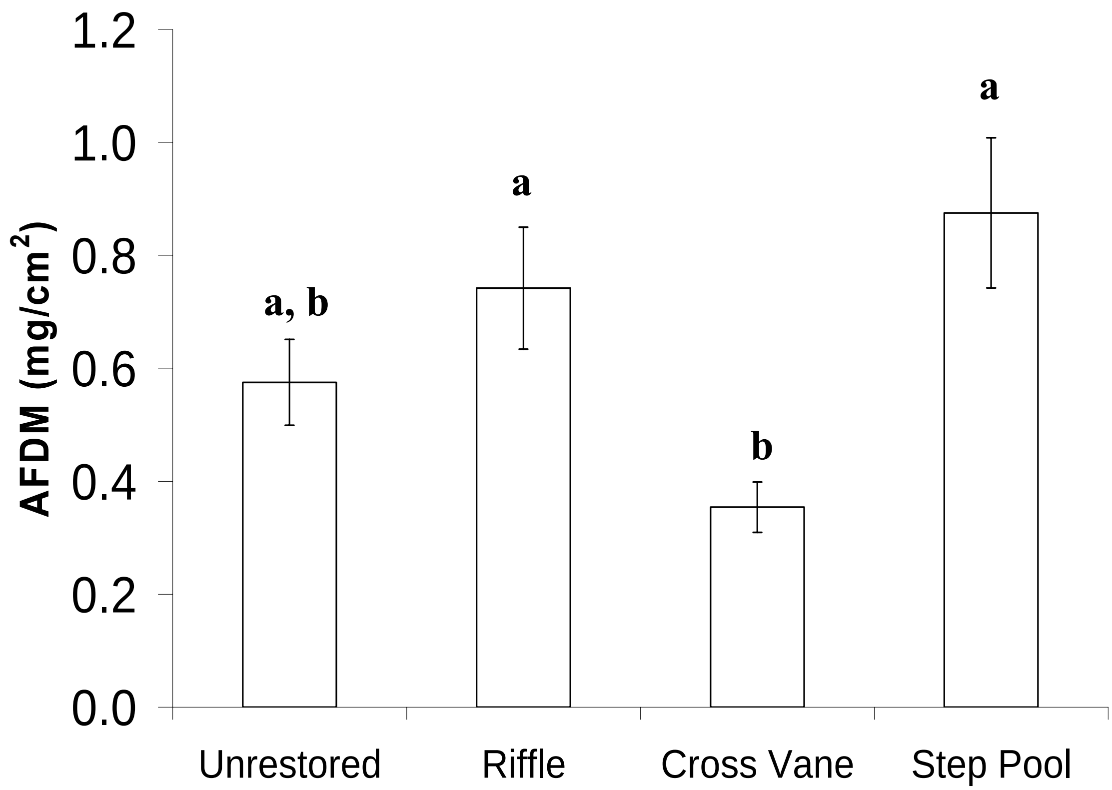


Figure 3. Mean (± 1 SE) accumulation of epilithic biomass on the bricks during the 3-4 week incubation period as determined by ash-free dry mass (AFDM) for unrestored sites, riffles, cross vanes, and step pools. Matching letters above bars indicates they were not significantly different at p= 0.05 (Tukey's HSD multiple comparison test).

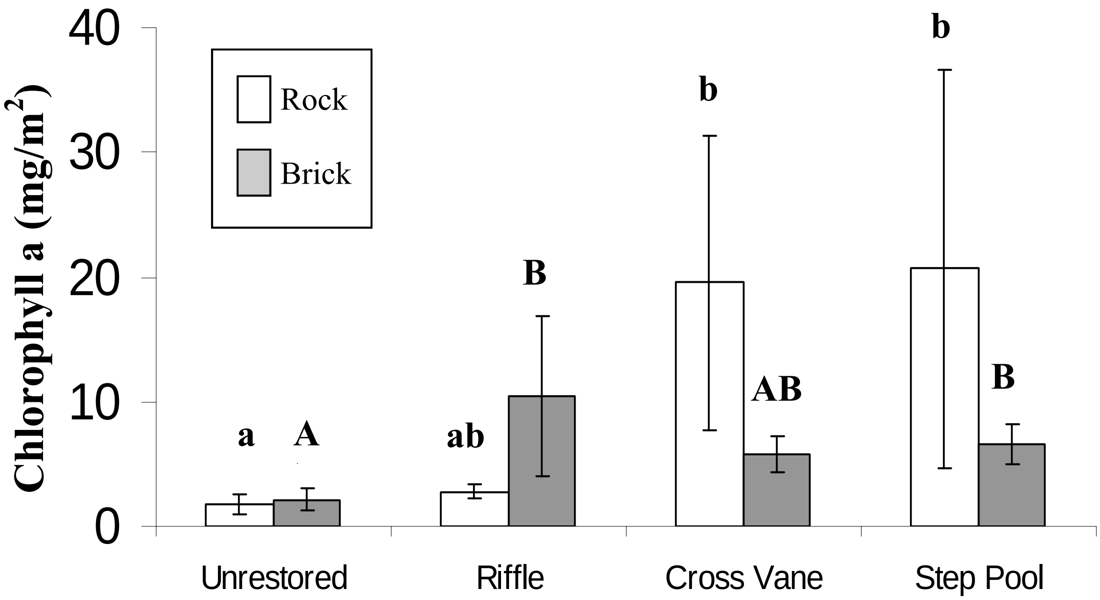


Figure 4. Mean chlorophyll *a* (± 1 SE) on rock substrates and mean chlorophyll *a* (± 1 SE) in the epilithic biomass that accumulated on the bricks during the 3-4 week incubation period for unrestored sites, riffles, cross vanes, and step pools. Matching letters above the bars indicates no significant differences at p= 0.05 (Tukey's HSD multiple comparison test).

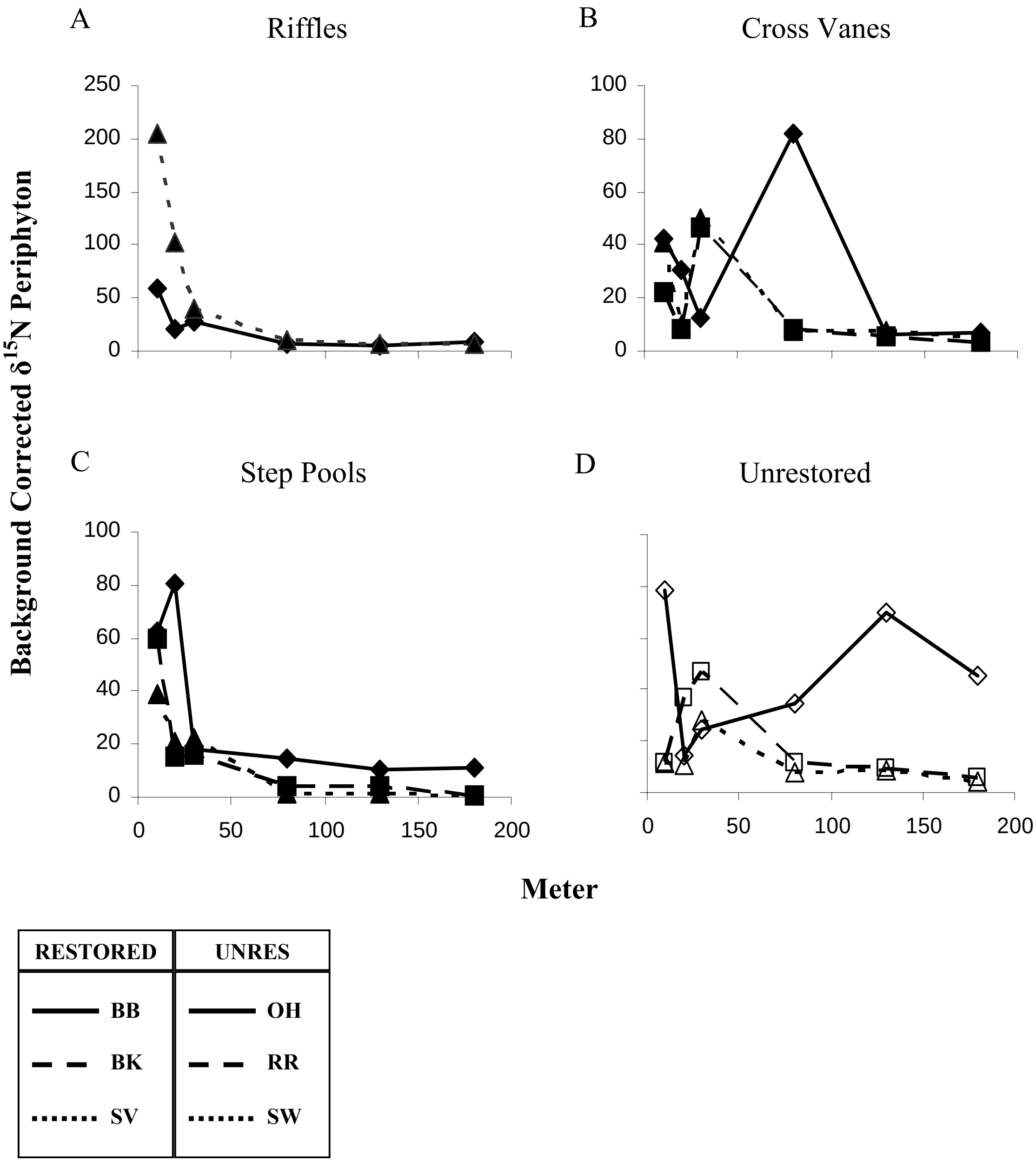


Figure 5. δ^{15}N of periphyton (background corrected) for (A) riffles, (B) cross vanes, (C) step pools and (D) unrestored sites. Due to disturbance during the $^{15}N\text{-}NH_4Cl$ pulse, Brown Bark riffle was excluded from analysis.

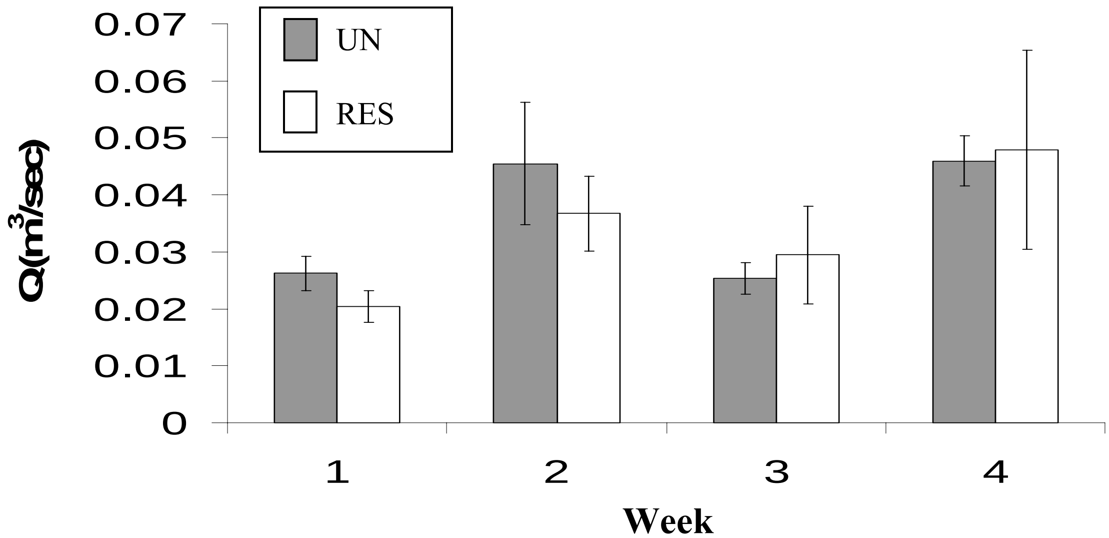
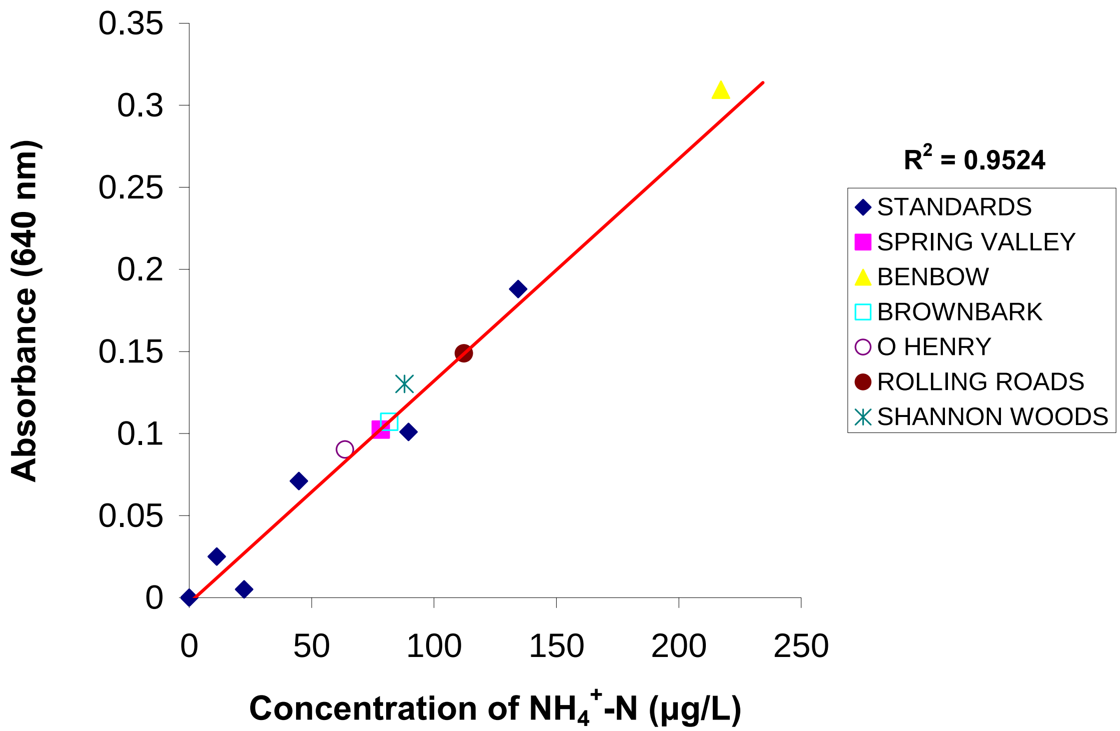


Figure 6. Weekly mean discharge rate (Q) m^3/sec (± 1 SE) for unrestored and restored streams. A two-way ANOVA indicated discharge did not vary significantly between unrestored and restored sites (p-value= 0.157).

Appendix B: Standard Curve for Initial NH_4^+-N Concentrations

B1. Standard curve for initial NH_4^+-N concentrations in the urban restored and unrestored streams.



Appendix C: List of Macroinvertebrate Taxa Collected

C1. List of all macroinvertebrate species collected at riffles (RIF), cross vanes (CV), step pools (SP), and unrestored (UNRES) sites. Abundance value abbreviations are rare (R) for 1-2 individuals collected, common (C) for 3-9 individuals collected, and abundant for 10 or more individuals collected in accordance with the NCBI (Lenat 1993). The number in front of the abundance value abbreviation indicates the number of sites (from 1-3 replicate sites per structure) the species was collected at.

Macroinvertebrate Species	RIF	CV	SP	UNRES
Coleoptera				
Helophorus spp.	1R			1R
Decapoda				
Cambarus acutus				2R, 1C
Procambarus acutus				2R, 1C
Diptera				
Dolichopodidae spp.			1R	
Simulium spp.			1C	
Simulium vittatum			1R	
Tabanus spp.			1R	1R
Tipula spp.	1R	1R	1R	1R
Diptera: Chironomidae: Chironomminae: Chironomini				
Chironomous spp.	1R	1R, 1C, 1A	2R	1R
Cryptochironomous fulvus		1R	1R	
Dicrotendipes nemodestus		1A	1A	
Dicrotendipes nervosus	1R	1A	1R, 1A	
Dicrotendipes spp.			1R	
Phaenospectra spp.		1R		
Polypedilum convictum	1R, 1A	1R, 2A	1C, 2A	1R, 1C, 1A
Polypedilum fallax		1R	1R	
Microtendipes spp.		1R, 1C	1C	
Diptera: Chironomidae: Tanypodinae				
Ablabesmia mallochi	1R	1R	2R	
Ablabesmia parajanta	1R	1R, 1C	1R, 1C	1R
Conchapelopia spp.	1R, 1A	3C	1R, 2C	1R, 1C
Larsia spp.	1R	1C	1R	
Natarsia spp.	2R	1A	1R	
Zavrelimyia spp.			1R	

C1 Continued.

Macroinvertebrate Species	RIF	CV	SP	UNRES
Diptera: Chironomidae: Chironomminae: Tanytarsini				
Rheotanytarsus spp.		1R	1C	
Tanytarsus spp.	1R	1C	2R	
Diptera: Chironomidae: Orthocladiinae				
Cricotopus bicinctus	1R		1R, 1C	
Cricotopus spp.		1R		
Hydrobaenus spp.	1R			
Orthocladius ombumbratus	1R	1R	1C, 1A	1R
Ephemeroptera				
Baetis interclaris	2C, 1A	2R	1R, 1A	2R
Mollusca				
Physella spp.	1C	2R, 1C	2C	2A
Sphaerinum spp.				1C
Odonata				
Argia spp.	2C, 1A	1R, 1C, 1A	2A	1A
Calopteryx spp.	1R, 2C			1R, 1C
Hetarina spp.				1R
Progomphus obscurus			1R	
Nasiaeschna pentacantha			1R	
Trichoptera				
Certopsyche sparna	2C, 1A	2R	1C	1C
Cheumatopsyche spp.	3A	1C, 1A	1R, 1C, 1A	1R, 1A
Hydropsyche betteni	1R			
Hydropsyche scalaris	1C			