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EVALUATION OF ECOLOGICAL FUNCTION IN RESTORED URBAN STREAMS

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ABSTRACT

Stream restoration projects often result from compensatory mitigation and are designed to reestablish ecological function to degraded systems. However, these projects are rarely subjected to post-restoration evaluation. The overall goal of this study was to evaluate the effect of stream restoration on ecological function in urban low and mid-order streams in Greensboro, NC. We conducted monitoring at two restoration sites and three urban reference sites from November 2002 to December 2003. We also conducted ^{15}N tracer enrichment experiments in a forested reference site, a restored urban site, and an unrestored urban site in August – Sepetember 2004. Surface and hyporheic water for dissolved oxygen and surface water were sampled at monitoring sites for nutrients (NO_3^-, PO_4^{-3}, and NH_4^+). Benthic macroinvertebrates were collected both quantitatively and qualitatively and were used in metric analyses. Dissolved oxygen was significantly higher in surface and hyporheic water in restored compared to reference monitoring sites, and was significantly lower upstream and higher downstream of restored sites. NO_3^- was significantly lower within restored sites. PO_4^{-3} and NH_4^+ levels were not different between sites. NCBI water quality scores in all sites were Fair. Taxa richness was similar at restored and unrestored sites. These monitoring results indicate that restoration may be improving urban stream water quality and macroinvertebrate communities through improved habitat. Reduced NO_3^- is especially important to basin-wide efforts to reduce N-loadings to downstream receiving waters. ^{15}N tracer experiments also indicated higher N-retention in a 2nd order restored reach compared to an unrestored reach. ^{15}N enrichment experiments showed that NH_4^+ uptake/m^2 to periphyton was greatest in Talbot's Branch, a relatively pristine forested stream, likely reflecting that the urban stream reaches have high concentrations of DIN, thus demand for NH_4^+ should be lower in those streams. However, restored Starmount GC had similar NH_4^+ vertical mass transfer velocity and higher rate of NH_4^+ uptake per m^2 compared to unrestored O'Henry despite greater stream depth and higher ambient NH_4^+ concentrations. This higher demand for NH_4^+ also suggests that restoration is having a positive effect on retention of N.

(KEY WORDS: urbanization; restoration; surface water quality; nutrients; riparian zones; periphyton; macroinvertebrates; ^{15}N uptake; N-retention; mass transfer velocity.)

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SUMMARY AND CONCLUSIONS

Monitoring of restored and unrestored reaches of urban streams in Greensboro, NC, showed that dissolved oxygen was significantly higher in surface and hyporheic water within restored than in unrestored sites, and was significantly lower upstream and higher downstream of restored sites. NO_3^- was significantly lower within restored sites. PO_4^{-3} and NH_4^+ levels were not different between sites. NCBI water quality scores in all sites were Fair. Taxa richness was similar at restored and unrestored sites. ^{15}N enrichment experiments showed that NH_4^+ uptake/m^2 to periphyton was greatest in Talbot's Branch, a relatively pristine forested stream, likely reflecting that the urban stream reaches have high concentrations of DIN, thus demand for NH_4^+ should be lower in those streams. However, restored Starmount GC had similar NH_4^+ vertical mass transfer velocity and higher rate of NH_4^+ uptake per m^2 compared to unrestored O'Henry despite greater stream depth and higher ambient NH_4^+ concentrations. This higher demand for NH_4^+ in the restored compared to the unrestored experimental reach is consistent with the monitoring observation that NO_3^- concentrations were lower in restored compared to unrestored reaches, and suggests that restoration is having a positive effect on retention of DIN.

The restoration of urban stream reaches along North and South Buffalo Creeks appears to be improving water quality in terms of dissolved oxygen availability, reduced NO_3^- levels and increased uptake of NH_4^+. Nitrate reduction, in particular, is a major water quality goal in the Cape Fear River Basin. However, the limited evidence of restoration in the macroinvertebrate community shows that the streams are still degraded. The fact that stormwater pipes bypass the filtering capability of the restored riparian zones means that nutrients, sediments, and contaminants carried within stormwater continue to reach the stream. In addition, transport distance of seston was relatively high in the restored stream where it was measured, suggesting that although restoration improved retention of DIN, restoration efforts need to also include more attention to stream geometry in order to also retain particulate N.

RECOMMENDATIONS

In light of the high cost of restoration projects, consideration needs to be given to repairing biological functions in addition to physical improvements of the restored system. Restoration of Greensboro urban streams appears to have been successful in improving N-processing, a major water quality management objective. However, in order to restore biological communities beyond tolerant organisms in urban streams, we suggest that restoration projects must include stormwater retrofitting in addition to in-stream habitat and channel improvements. To enhance retention of particulate matter, restoration projects should include channel meanders.

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INTRODUCTION

High order North Carolina coastal plain rivers are severely impacted by excess nutrients, especially nitrogen, which causes fish kills and human health problems, damages coastal shellfish fisheries, and results in deterioration of recreational uses of these waters (see Glasgow and Buckholder 2000, Malin et al. 1997). Recent research has shown that efforts to improve water quality in receiving waters need to focus on low order streams where nutrient processing and retention capacity is highest (see Peterson et al. 2001, Mulholland et al. 2000). All anthropogenic land uses can strongly influence the supply of nutrients and fine sediment to streams, which impact stream morphological features, algal productivity and fish and invertebrate populations (e. g., Lenat 1984, Everest et al. 1987, Munn et al. 1989, Osborne and Kovacic 1993, Lenat and Crawford 1994, Richards et al. 1996). Recent research has focused especially on buffer programs in agricultural lands because these areas characteristically have large areas of exposed soil and high rates of fertilizer application, both of which enter streams in high concentrations. However, less research has focused on urban streams (see Wang et al. 2003), which are strongly influenced by nutrients from lawn fertilizer, sediments, failed municipal waste (sewage) systems, pet waste, and other pollutants. In such streams, the nitrogen processing capacity of the streambed is exceeded and excess nutrients are delivered to downstream reaches. Our research using natural abundance of ^{15}N, a stable isotope of nitrogen, has shown that anthropogenic nitrogen is processed through the food web of North Buffalo Creek in urban Greensboro in a manner that is fairly site specific (Ulseth and Hershey 2005).

Stream restoration is becoming a national agenda (e. g., Charbonneau and Resh 1992, NRC 1996, Kondolf 1996, Roni et al. 2002), with participation by management agencies as well as citizen groups. Stream restoration projects are often the result of compensatory mitigation plans and are designed to reestablish ecological function to degraded systems. However, these systems are rarely subjected to post-project evaluation to determine effectiveness of restoration. Restoring in-stream habitat in degraded streams will improve habitat complexity and may promote recovery of aquatic species (Roni et al 2002). Stream bottoms composed of cobble and gravel as opposed to sand and silt are most able to support a diverse and productive benthic population. A diverse substrate influences surface and interstitial water flow, organic material accumulation and availability of refugia (Fischenich and Allen 2000). Therefore, one manner of restoring stream habitat includes the creation of artificial pool-riffle sequences. A meandering pool-riffle sequence provides differing flow conditions and a range of habitats that promote the production of benthic organisms (N.C. DWQ(A) 2003, Ferguson 1991, Fischenich and Allen 2000). Reconnecting an incised stream channel to its floodplain (at bankfull discharge) allows for retention of excess sediments and associated nutrients on the floodplain (Buck Engineering training course 2003). Restoring fragmented riparian zones is important for reestablishing connectivity between the stream and floodplain areas and for increasing spatial heterogeneity of the riparian vegetation at the local scale. Macroinvertebrates move extensively between the stream and riparian zone through dispersal, feeding, and developmental activities (Smock et al 1992, Fischenich and Allen 2000). Thus, an energy link exists between these areas that promotes connectivity through restoration.

In monitoring biological stream health, benthic macroinvertebrates are most frequently. This reflects that they have high taxonomic and functional diversity they occur in nearly all stream systems (Purcell et al 2002, Resh et al 1996). Benthic macroinvertebrates also serve as good indicators of stream water quality because they have well-characterized tolerances to pollution (Resh et al 1996, Courtemanch 1996, Benke et al 1981, Lenat 1988, 1993). Because point and non-point sources contribute a wide range of pollutants to urban streams, highly tolerant macroinvertebrate groups like Chironomidae and Oligochaeta typically dominate the benthic communities of urban streams. As macroinvertebrate community diversity declines, the lotic food web becomes disrupted, resulting in loss of ecological function. In general, urbanization affects benthic macroinvertebrates in terms of decreased diversity and abundance in response to toxins, temperature change, siltation, and organic nutrients (Mulholland and Lenat 1992, Paul and Meyer 2001, Lenat and Crawford 1994).

Physical and chemical assessment of water quality are important for interpreting biological data as well as for quantifying loading to receiving systems. Stream temperatures affect a wide array of factors including chemical processing, saturation of dissolved gases, and the metabolic rates and emergence/migration activities of aquatic organisms (Hauer and Hill 1996, LeBlanc et al 1997, Fischenich and Allen 2000). Average stream temperatures increase in response to urbanization events such as the removal of riparian vegetation and increased impervious surfaces. Stream water dissolved oxygen is a key component in maintaining reproduction, development and survival of aquatic organisms (Wang et al 2003). Most undisturbed streams have abundant supplies of oxygen, and, consequently, many stream macroinvertebrates have a low tolerance to fluctuations in O_2 concentrations (Fischenich and Allen 2000). The stress imposed by point source municipal sewage treatment discharge and industrial waste, and by non-point source pollutants in urban waterways increases with rising temperatures. Microbial processing of these inputs reduces the available oxygen within the stream (Hauer and Hill 1996, Mulholland and Lenat 1992). The exchange of oxygen between stream and subsurface habitats is critical to the various organisms that utilize the hyporheic zone. Upwelling of organic matter and downwelling of oxygen have been studied extensively and have been shown to influence the distribution of the benthos in a variety of aquatic systems (Boulton et al. 1998. Boulton 1993, Triska et al 1993, Findlay 1995, Duff and Triska 1990, Triska et al. 1993). Excess loading of nutrients in urban streams can result in algal blooms that deplete dissolved oxygen concentrations during nonphotosynthetic periods (Mulholland and Lenat 1992). In most urban areas, nutrients come from point and nonpoint sources (Watson et al 1999, Lenat and Crawford 1994, Mulholland and Lenat 1992). Stormwater runoff also carries with it a large number of pollutants and is often the most difficult non-point source to assess and treat (Benke et al 1981).

Despite the interest in restoration, little attention has been paid to effectiveness of these projects at restoring ecological function. The EPA (Brady 1996) and others (e. g., Roni et al. 2002, Kondolf 1996) have emphasized the need to focus on restoration of ecological processes in order for restoration efforts to be meaningful. However, management agencies rarely have funds to conduct such studies and doing so is not necessarily in their mandate.

The objectives of the current project were to (1) to evaluate the effectiveness of stream restoration on biotic communities by comparing water quality and macroinvertebrate metrics in restored and non-restored sites; and (2) to evaluate the effectiveness of stream restoration on nitrogen processing in an urban stream through use of ^{15}N tracer experiments.

METHODS

Study Region

The research was conducted in low and mid-order reaches of North and South Buffalo Creeks, two urban streams in Greensboro, Guilford County, North Carolina, and part of the Haw River catchment of the Cape Fear River Basin. Greensboro is North Carolina's third largest city, with a growing population of over 220,000 distributed over a 296 square km area within the headwaters of the Cape Fear River Basin. The history of this landscape includes anthropogenic disturbance in the form of industry beginning in the 19th century. Recent urban expansion has resulted in a rapid spread of residential and commercial development throughout the catchment. North and South Buffalo Creeks have poor water quality, degraded stream habitat and impaired biological communities from urban runoff, point-source effluent, nutrient enrichment, and fecal coliform bacteria (N.C. DWQ, 2000).

Macroinvertebrate metrics and water quality parameters were compared in five sampling sites, chosen based on restoration status. Starmount Park, located just downstream of the Starmount Golf Course, and Lindley Park located downstream of the Starmount Park site were both restored 3-years prior to the study. Restoration included streambank stabilization and construction of artificial riffle-pool sequences. A 10 m riparian buffer is maintained at both sites. The riparian zone creates a moderate amount of shading for the streams. Lindley Park, situated within the Greensboro Arboretum, includes more established vegetation, resulting in more shading for the stream. Both the upstream and downstream sites at Lindley and Starmount sites had a riffle-pool sequence similar to the restored riffle-pools. Brown Bark Park is an unrestored tributary of North Buffalo Creek, located north of the Starmount Park site. It is situated within a residential park and is subject to continual mowing, resulting in no shrubs or trees and little shade for the stream. Latham Park is a few km downstream of the restored sites on North Buffalo Creek, located along Greensboro's greenway park system, which runs through the Duke Powerline Corridor. The park is subject to continual mowing, resulting in no shrubs or trees and little shade for the stream. The stream channel is highly eroded and deeply incised. Amber Park is an unrestored tributary of South Buffalo Creek located within a residential park. This site is subject to mowing, resulting in little shade for the stream.

Three additional sites, a reference stream, and a restored and unrestored reach of urban North Buffalo Creek, each approximately 500 m in length, were selected for ^{15}N enrichment experiments to evaluate N uptake into periphyton and incoroporation into the macroinvertebrate community. Talbots Branch, in the Uwharrie National Forest was selected as the reference site. This reach was within a fully forested watershed, had a closed canopy, and had no apparent anthropogenic inputs of N from surface water, although inputs from groundwater and the airshed are likely everywhere in the NC piedmont. O'Henry Creek is a tributary to North Buffalo Creek that drains a residential neighborhood and receives storm water and potentially sanitary exfiltration inputs. O'Henry has an unmowed riparian zone up to 5 m wide, but has not been a site of any channel restoration projects. Starmount Golf Course (Starmount GC), restored as described above for Starmount Park, was upstream of Starmount Park. The canopy of Starmount GC and O'Henry was partially open throughout most of the reaches, and more open in Starmount than O'Henry.

Temperature and dissolved oxygen

Surface water temperature was measured seasonally in the thalweg of each of the five 100 m stream reaches sampled for monitoring studies from November 2002 to December 2003, using a Vernier Lab Pro dissolved oxygen probe (Lynam 2004).

Surface and hyporheic water dissolved oxygen was measured seasonally in each 100 m monitoring reach from November 2002 to October 2003 (Lynam 2004). DO concentrations were measured with a Vernier LabPro dissolved oxygen probe. At each site, DO was measured in 3 riffle, 3 pool and 3 run habitats, using a stratified sampling design. One area from each habitat type was randomly selected along the 100 m reach within each site. In August 2003, DO samples were taken to compare restored reaches with adjacent reaches. Sampling included measurements from upstream of Starmount Park and downstream of Lindley Park. No upstream site was selected for Lindley Park because the downstream reach of Starmount Park (i.e., what would be otherwise considered the upstream reach of Lindley park) transitions into the restored reach of Lindley Park.

Surface water was withdrawn into a 600 cc plastic syringe at each site. The plunger was removed from the syringe, the calibrated DO probe was inserted, and triplicate readings were taken. A PVC T-bar was used to dig wells at least 10cm into the stream bottom in a riffle, pool, and run. The T-bar was removed from the PVC pipe and a thin plastic tube was inserted down the length of the pipe until it touched the bottom of the well. The syringe was attached to the tube at the top of the PVC pipe and hyporheic water was withdrawn and rinsed through the syringe. Hyporheic water was withdrawn again and triplicate readings were taken in the same manner as with the surface water samples.

Nutrient data collection

Surface water samples were seasonally collected for nutrient analyses at the five monitoring sites on seven dates from November 2002 to December 2003, during stream baseflow conditions (Lynam 2004). Two random samples were collected per site from surface water upstream of any turbidity due to sampling activity. Water was rinsed three times through a 600 cc plastic syringe and then withdrawn into the syringe a fourth time for collection. A filter was attached to the end of the syringe and the water was ejected into an acid-washed 120 ml bottle. The bottles were kept on ice in a dark cooler until returned to laboratory, then frozen until analyzed with a Brann+Luebee TRAACS 2000 Autoanalyzer and a Shimadzu UV-VIS Spectrophotometer for NO_3^-, NH_4^+, and PO_4^{-3}.

Benthic macroinvertebrate collection

Macroinvertebrate sampling occurred at Brown Bark, Lindley, and Starmount parks during baseflow from November 2002 to June 2003. Amber and Latham parks were additionally sampled from February to June 2003 (Lynam 2004).

A stovepipe sampler with a 300 cm^2 area, was used to collect eight random benthic samples per site, using a stratified sampling design. Four areas from both riffle and bank edges were randomly selected and sampled along the 100 m reach (8 samples per reach x 6 sites = 48 samples). The riffles were similar in terms of depth, but the restored sites had a cobble substrate while the unrestored sites had a finer gravel substrate. Rocks were removed from the stovepipe enclosure and examined for benthic macroinvertebrates. A 250 µm mesh hand held net was used to stir and collect organic material and macroinvertebrates within the enclosed area. This procedure was repeated until no more organic material was found. Samples from each replicate were placed in separate bottles containing 95% alcohol. To account for taxa missed by stovepipe sampling, 1 sweep-net sample of the streambank root zones at each habitat type (e.g., riffle, pool, run) was performed during each sampling event. The sampling procedure was not specific to the North Carolina Biotic Index.

In the laboratory, samples were sorted for macroinvertebrates, which were stored in 70% alcohol and identified to the lowest possible taxonomic levels using keys of Merritt and Cummins (1996), Brigham et al (1982), and Epler (2001).

Data analyses for water quality and macroinvertebrate monitoring studies

Taxa richness and densities were determined and the North Carolina Biotic Index (Lenat, 1993) was used to apply tolerance values to the taxa found within the streams for each sampling date. Because the sampling method was not restricted to the NCBI method, water quality scores were compared between site conditions and were not applicable to other published NCBI scores. Richness was the total number of taxa per site (from riffle and edge habitats). Taxa densities were separately calculated by habitat type for each sampling date and were combined for between site comparisons. Metrics, including numbers of Ephemeroptera, Trichoptera, and Corbiculidae, percent Chironomidae and Oligochaeta, percent dominant taxon, and the Shannon-Weiner diversity index, were applied to data from each season. Number of Plecoptera (the third sensitive order in the EPT metric) was not applied to our data due to the low occurrence of this order in piedmont urban streams (Lenat 1988). Percent dominant taxon is a measure of redundancy in which high levels are correlated with the dominance of a tolerant organism and low community diversity. The Shannon diversity index calculates a value based on the proportional abundance of taxa richness and evenness (Fischenich and Allen 2000). For this study, diversity values are determined at the species level.

Repeated measures MANOVA was used to account for seasonal variations in each physical variable (e.g., surface water dissolved oxygen concentration), with sampling date as the within subject factor and site condition as the between subject factor. Temperature data were log transformed prior to analyses. Pairwise differences for data were examined with Tukey's test. In order to eliminate the effect of temperature on differences between DO concentrations, percent saturation of DO of surface water was calculated and compared through repeated measures MANOVA. A 1-way Welch's ANOVA was conducted to determine if DO varied between sites upstream and downstream of the restored sites for the August sampling date. Analyses were performed using the JMP V 5.0.1.2 for Windows (SAS Institute Inc. 2003).

Multi-Response Permutation Procedures (MRPP) were used to test for significant differences in the macroinvertebrate communities between the site conditions. MRPP is a permutation test based on distance measures between all pairs of members of each group that then calculates an average distance for each group. Tight group clustering results in small intra-group distances (A<0), thus indicating heterogeneity that exceeds that determined by chance (A=0). Loose clustering results in large intra-group distances (A>0) thus indicating no difference between determined communities (McCune and Mefford 1999; McCune and Grace 2002; Muotka et al 2002). The Euclidean metric was used as the distance measure for MRPP. The indicator value method (IndVal) was used to detect species differences between the site conditions. The IndVal method compares information about the species abundance in a group and its appearance only in that group. Indicator values for each species in a group are based on the standards for a perfect indicator, in which a species is always present and exclusive to that particular group (IndVal = 100). The indicator values are tested for statistical significance using Monte Carlo tests with 1000 permutations (McCune and Grace 2002; Dufrene and Legendre 1997; Muotka et al 2002). Rare taxa were included in all analyses. All multivariate analyses were performed using PC-ORD software (Version 4.0 McCune and Mefford 1999).

^{15}N enrichment experiments

^{15}N tracer addition experiments were conducted in 500 m reaches of a forested reference (Talbot's Branch), and in restored (Starmount GC) and unrestored (O'Henry Creek) low-order reaches of North Buffalo Creek, to study uptake of NH_4^+, and movement of algal-derived N into the dominant primary consumers in the food web. At each site, a ^{15}N addition site (the "dripper") was selected for its mixing potential, potential for disguising the dripper bottle to prevent vandalism, and for the presence of multiple riffles located downstream. Nine sampling stations at riffles were selected downstream of the dripper in the ~500 m reach, and one upstream site was also selected to serve as a correction for background $\delta^{15}N$ of food web components.

Using 98% enriched ^{15}N ammonium chloride ($^{15}NH_4Cl$), a $^{15}NH_4Cl$/L solution was prepared to achieve a target ^{15}N enrichment in the periphyton of 50-100 per mil based on ambient NH_4^+ concentrations in each stream (calculations given in Hershey and Peterson 1996). The solution for each stream was added to a 2-L nalgene bottle, prepared as a Marriott bottle (see Peterson et al. 1983) and outfitted with Tygon capillary tubing to withdraw the $^{15}NH_4Cl$ solution out of the bottle and deliver it to the stream at a constant rate. This technique successfully enriched the periphyton with ^{15}N (see below) but had negligible effect on stream water NH_4^+ concentration.

^{15}N addition experiments were conducted approximately simultaneously during August-September of 2004 for approximately 4 weeks each. During this period there was minimal rainfall such that base flow conditions existed in all streams most of the time. Periphyton, leaf detritus, suspended fine particulate organic matter (FPOM), fine benthic organic matter (FBOM) and dominant consumers were collected weekly during the 4 wk experiments (see Hershey and Peterson 1996, Peterson et al. 2001). Macroinvertebrates were placed in scintillation vials filled with filtered stream water for 24 h to permit clearing guts of unassimilated foods before storage in a freezer. Samples were dried at 60^0C for 24 h, and macerated. Samples were analyzed for $\delta^{15}N$ at Colorado Plateau Stable Isotope Laboratory. Data analysis focused on samples from the last sampling date, but other dates were also examined to evaluate whether components were at isotopic equilibrium. Epilithic material should reach equilibrium levels of $\delta^{15}N$ within 2 weeks, and most primary consumers this has been shown to be about 4 weeks (see Peterson et al. 2001, Tank et al. 2001, Rezanka and Hershey 2003).

All ^{15}N values were expressed as δ (del) units, which are parts per thousand (per mil) relative to a standard:

$$\delta^{15}N = [(R_{sample}/R_{standard}) - 1] * 1000 \qquad [1]$$

where R = $^{15}N : {}^{14}N$ ratio, and the N isotope standard is air (Peterson and Fry 1987).

$\delta^{15}N$ of upstream control samples was subtracted from the $\delta^{15}N$ of the downstream samples for each food web component in order to correct for the background values. Our previous work shows that these background values range from 0-9 per mil for the various food web components (Ulseth and Hershey 2005). Exponential decay models were fit to these data to determine the decay rate of ^{15}N labeling with downstream distance:

$$\delta^{15}N_D = \delta^{15}N_O e^{-kD} \qquad [2]$$

where D = distance from dripper (at 0 m) and k = slope of regression line, or decay rate of the ^{15}N signal with distance downstream. We used k as an approximation of the uptake of ^{15}N by the stream bed. This is an approximation of k because we actually measured uptake into periphyton, and some uptake could occur through other mechanisms. However, the ^{15}N signal in the periphyton should provide a very good approximation of ^{15}N available in the water column. Additional uptake parameters, including uptake length (S_w), uptake per unit area (U), and the transfer rate coefficient (v_f) were calculated from k (Stream Solute Workshop 1990, Webster and Ehrman 1996). S_w measures the average distance traveled by an NH_4^+ molecule before immobilization on the streambed, whereas U and v_f permit more direct comparisons among streams that differ in discharge and background concentrations of NH_4^+.

RESULTS

Surface water temperature

Average temperatures ranged from 3.2°C (February 2003) to 26.7°C (August 2003) in restored sites and from 6.7°C (December 2003) to 24.2°C (August 2003) in unrestored sites (Figure 1). Because there appeared to be a seasonal variability in temperature, a repeated measures MANOVA was conducted with sampling date as the repeated measure. No significant differences existed between site conditions (P=0.36, MANOVA) or for the date x condition interaction term (P=0.31, MANOVA), but did occur between sampling dates (P=0.01).

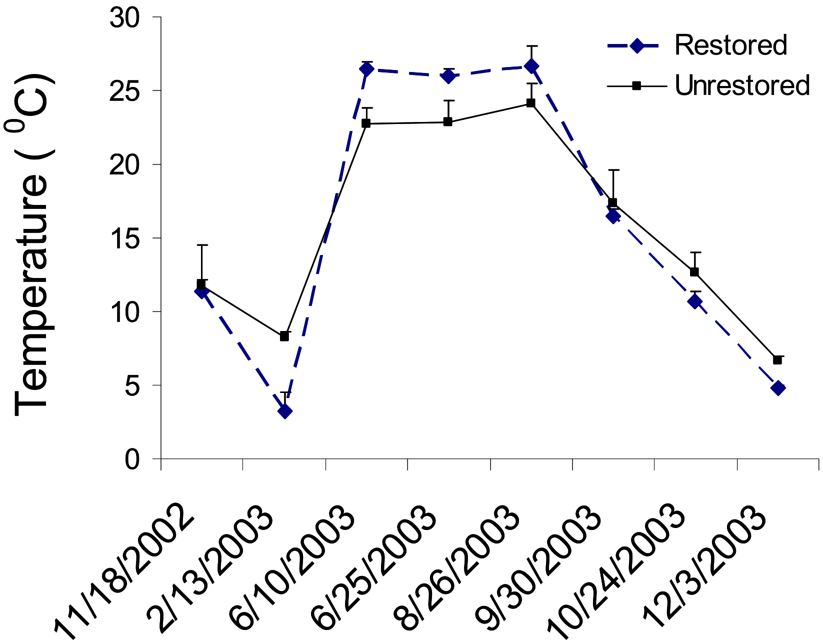

Figure 1. Surface water temperature readings across study period. Measurements were taken the thalweg of the stream reach in each site. Error bars represent 1 SE. Temperatures are not significantly different between sites across the study period .(Repeated measures MANOVA, P=0.36). From Lynam 2004.

Dissolved oxygen

Average surface water DO concentrations ranged from 8.3 mg l^{-1} (August 2003) to 12.3 mg l^{-1} (October 2003) in restored sites and from 6.6 mg l^{-1} (August 2003) to 9.0 mg l^{-1} (October 2003) in unrestored sites (Figure 2A). Repeated measures MANOVA showed that DO concentrations were higher in the restored sites than in the unrestored reference sites across the study period (P=0.005). Differences between sites were more noticeable in the colder months (February, October) than in the warmer months (June, August) (Figure 2A). A significant difference existed between sampling dates (P=0.010) (within subjects factor) but not for the interaction term (P=0.17) across the study period.

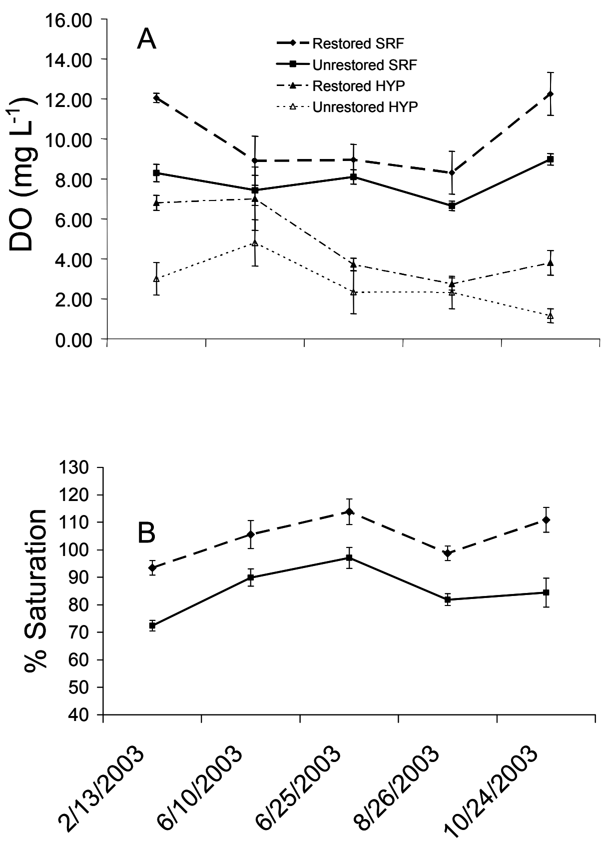

Figure 2. Surface and hyporheic water dissolved oxygen concentrations (mg l^{-1}) (A) and surface water % saturation (B) across study period. Data are mean values from a stratified sampling design (riffle, pool, run). Error bars represent 1 SE. From Lynam 2004.

Restoration of Starmount and Lindley Parks resulted in an increase of surface water DO by 1.3 to 3.6 mg l $^{-1}$ compared to reaches upstream and downstream of those sites. DO concentrations upstream of restored Starmount Park were significantly lower than within that site (Table 1, P=0.02, 1-way Welch's ANOVA). DO concentrations downstream of restored Lindley Park were significantly higher than within that site (P=0.0003).

DO percent saturation ranged from 72-114% between site conditions and was significantly higher in the restored sites than in the unrestored sites across the study period (Figure 2B, P=0.04). On the August sampling date, DO percent saturation ranged from 76-127% upstream to downstream of the restored sites (Table 1). DO upstream of restored Starmount Park was significantly lower than within that site. DO downstream of restored Lindley Park was significantly higher than within that site.

Average hyporheic DO concentrations ranged from 2.7 mg l $^{-1}$ (August 2003) to 7.0 mg l $^{-1}$ (June 2003) in restored sites and from 1.2 mg l $^{-1}$ (October 2003) to 4.8 mg l $^{-1}$ (June 10, 2003) in unrestored sites. A repeated measures MANOVA showed that hyporheic DO concentrations were significantly different between sites (Figure 2A). Hyporheic DO concentrations were higher within the restored sites than within the unrestored sites across the study period (p=0.009). A significant difference occurred for the sampling dates (P=0.0003), but not for the interaction term across the study period (P=0.36).

Restoration of Starmount and Lindley Parks resulted in no discernable difference in hyporehic DO concentrations compared to upstream and downstream reaches for the August sampling date. Hyporheic DO concentrations ranged from 1.7-3.5 mg l $^{-1}$ upstream to downstream of the restored sites (Table 1). DO concentrations for upstream of Starmount Park were not significantly different than within that site (P=0.41). DO concentrations downstream of restored Lindley Park were not significantly different than within that site (P=0.87).

Table 1. Surface and hyporheic water DO concentration and surface water % saturation for sites along North and South Buffalo Creek upstream and downstream of restored sites on 8/26/3003. Concentration in mg/l (±SE). Concentration, % saturation are mean values based on stratified sampling design (riffle, pool and run). Data were log transformed and compared using 1-way Welch's ANOVA. * denotes means that are significantly different between site pairs.

	Surface Water Concentration	% Saturation	Hyporheic Water Concentration
Upstream Starmount	7.9 (0.16)*	102.5 (1.84)*	3.5 (0.78)
Restored Starmount	9.2 (0.16)*	126.7 (1.84)*	2.4 (0.78)
Restored Lindley	9.4 (0.20)*	123.3 (2.94)*	3.4 (0.62)
Downstream Lindley	12.9 (0.23)*	172.5 (2.94)*	3.1 (0.62)
Upstream Hillsdale	6.7 (0.40)	82.3 (4.90)	1.9 (0.40)
Restored Hillsdale	6.2 (0.40)	76.1 (4.90)	2.4 (0.40
Downstream Hillsdale	6.9 (0.40)	85.2 (4.90)	1.7 (0.40)

Nutrients

Phosphate concentrations varied seasonally from 0.03 mg l^{-1} (February 2003) to 0.14 mg l^{-1} (June 2003). Concentrations ranged from 0.03-0.11 mg l^{-1} in restored sites and 0.07-0.14 mg l^{-1} in unrestored sites (Figure 3A). Ammonium concentrations varied seasonally from 0 mg l^{-1} (December 2003) to 0.10 mg l^{-1} (June 2003). Concentrations ranged from 0.003-0.03 mg l^{-1}in restored sites and 0-0.10 mg l^{-1} in unrestored sites (Figure 3B). Nitrate concentrations varied seasonally from 0.03 mg l^{-1} (April 2003) to 0.40 mg l^{-1} (June 2003). Concentrations ranged from 0.03-0.30 mg l^{-1} in restored sites and 0.17-0.39 mg l^{-1} in unrestored sites (Figure 3C).

A repeated measures MANOVA showed that PO_4^{-3} and NH_4^+ concentrations for surface water samples were not significantly different between sites across the study period (Figure 3A, 3B). There was a significant difference between sampling dates for phosphate and ammonium (P=0.02, 0.002). Nitrate concentrations were significantly lower in restored sites than in unrestored sites across the study period (Figure 3C, P=0.02). There was a significant difference between the sampling dates for NO_3^- (P=0.004). No difference existed for the interaction term for any nutrient across the study period (P>0.05).

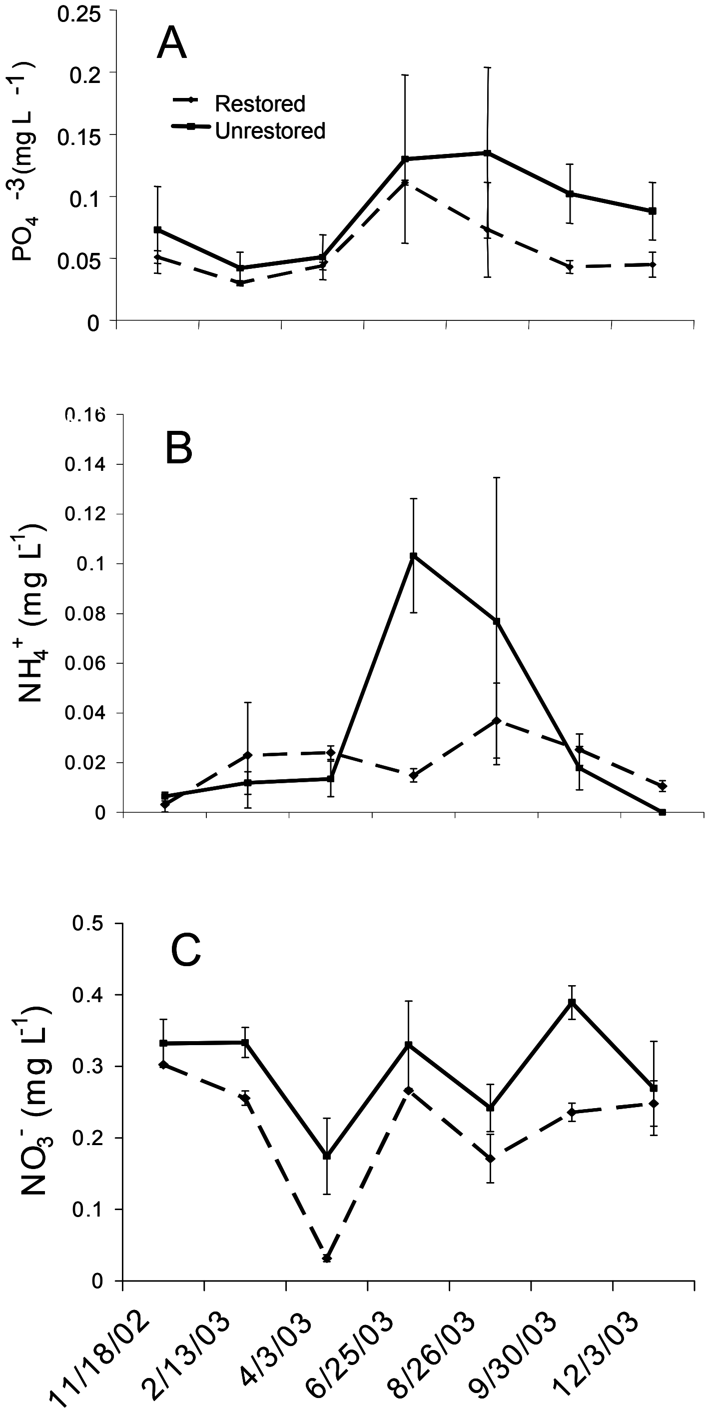

Figure 3. Surface water phosphate (A), ammonium (B) and nitrate (C) concentrations across study period. Concentrations are mean values from repeated measures MANOVA. Error bars represent 1 SE. Mean concentrations between site conditions are not significantly different for phosphate (P=0.32) or ammonium (P=0.17). Nitrate was significantly lower in restored than unrestored sites (P=0.02). From Lynam (2004).

Benthic macroinvertebrates

Eighty-nine total taxa were collected through stove-pipe sampling across the sampling period. Fifty-three taxa (30 in November, 34 in February, 53 in June) were collected in restored sites and 46 taxa (37 in November, 28 in February, 46 in June) were collected in unrestored sites. Seventeen total taxa were collected through stream-bank sweep-netting during June. Thirteen taxa were collected in restored sites and nine taxa were collected in unrestored sites. The percent abundance of the dominant taxa varies between site conditions but Oligochaeta is the most common dominant taxon overall. NCBI scores for restored sites were Fair, and for unrestored sites varied from Good-Fair to Fair.

Repeated measures MANOVA results for stove-pipe samples were highly significant only for site condition for the log mean density of taxa ($P=0.0030$). Marginally significant differences were seen for site condition for the log % Chironomidae ($P=0.08$), and for time and the interaction term for mean # *Corbicula* ($P=0.06$). Pairwise means comparisons (Tukey's test, $P<0.05$) indicated that restored sites supported a higher taxa richness than unrestored sites in June (Figure 4A). Restored and unrestored sites were similar in H′ diversity (Figure 4B) and the mean # *Corbicula* was lower in restored sites than unrestored sites in February (Figure 4C). For sweep-net samples, no difference existed for taxa richness between site conditions (Figure 5).

The restored sites included five intolerant species, *Hydropsyche scalaris, Symphtiopsyche sparna, Eukiefferiella brehmi, Orthocladius nigritis* and *Cricotopus cylindraceus* while the unrestored sites included only one intolerant species, *Paracladopelma nereis*.

MRPP test results indicate among-group differences close to that expected by chance for both the winter data (Table 2, $A=0.09$) and for the summer data ($A=0.03$). MRPP analysis could not be run on the fall data due to too few study sites. Results from the indicator value test indicate that no particular species are functioning as indicators of either restored or unrestored site conditions for the fall, winter and summer data ($P>0.05$).

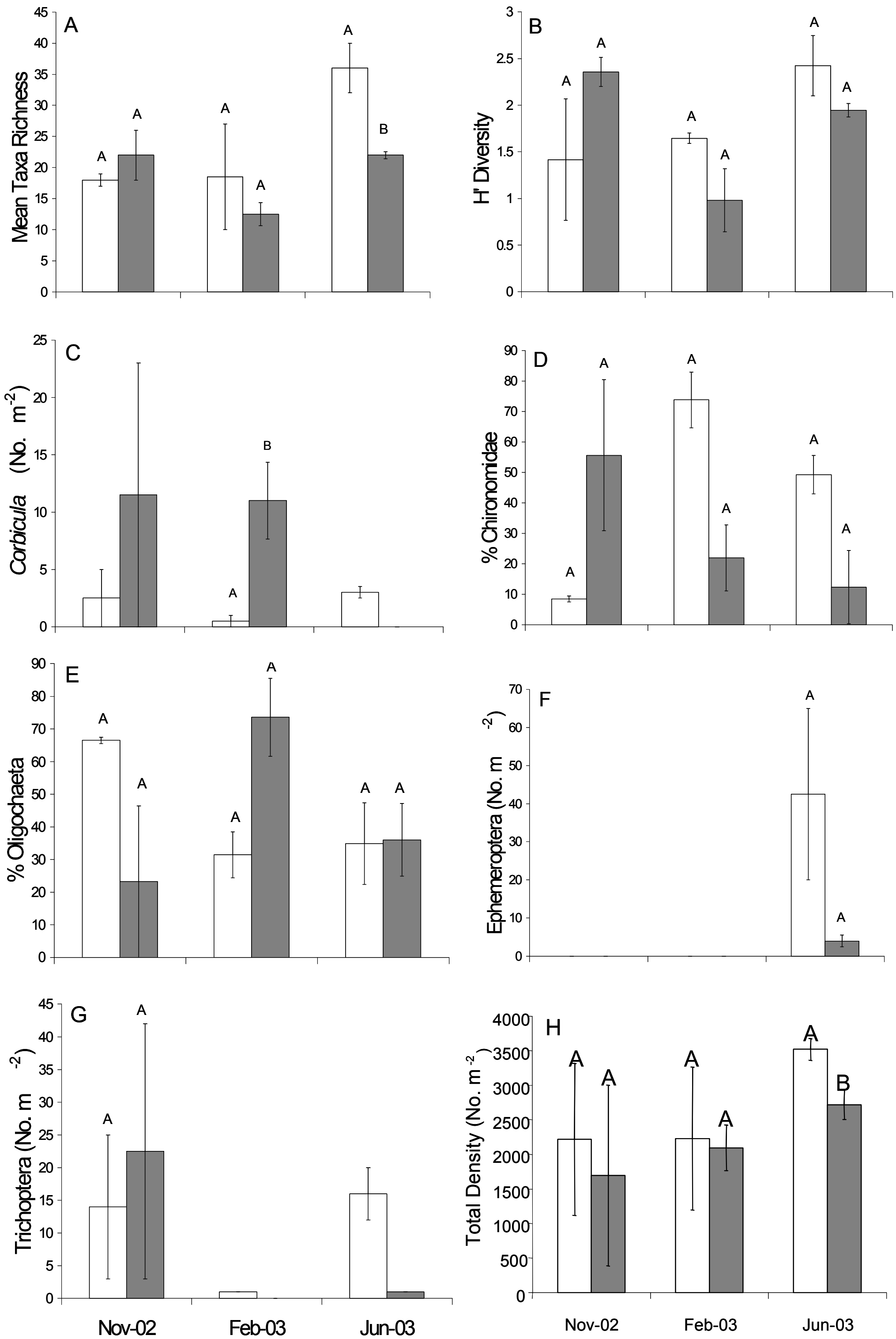

Figure 4. Comparison of metrics used to evaluate differences in stove-pipe samples. Values are means + standard errors. For each sampling date, bars with the same letter designation were not significantly different between sites (Tukey's test, P>0.05). Bars with no letter designation did not include enough taxa for pairwise comparisons (Lynam 2004).

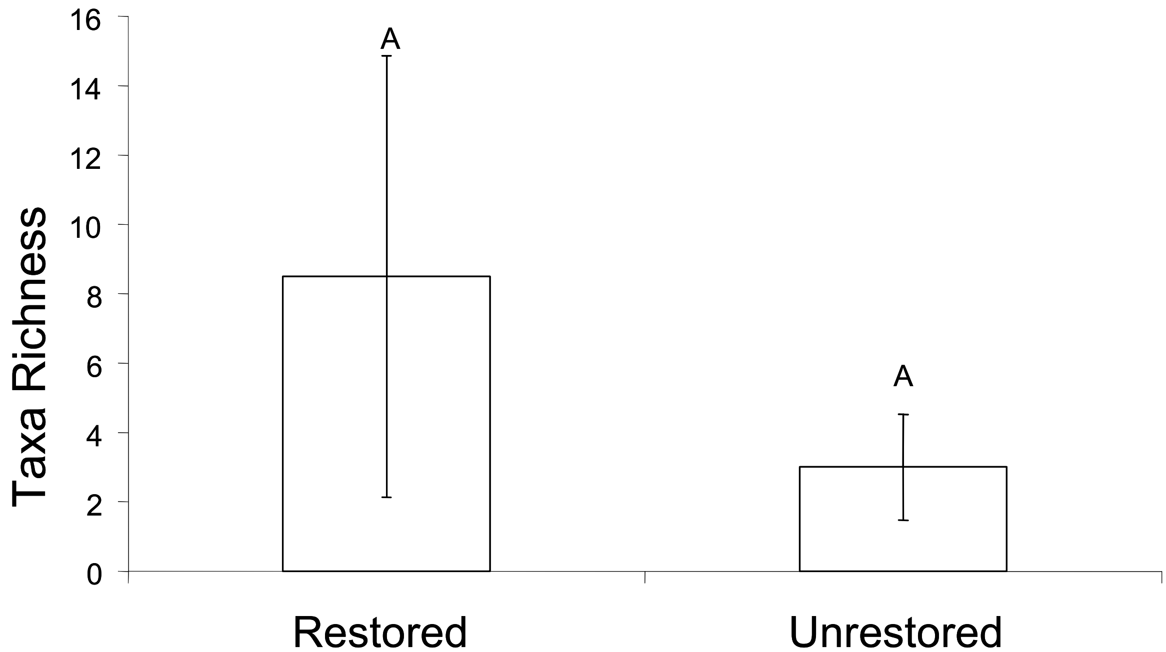

Figure 5. Comparison of taxa richness from streambank sweep-net samples between site conditions. Samples were collected in June 2003. Richness was determined to be the total number of taxa per sweep-net samples (3 sweeps per streambank). From Lynam 2004.

^{15}N enrichment experiments

^{15}N enrichment experiments showed that ^{15}N was incorporated into periphyton, seston, and consumer components in all streams, although the level of enrichment was lower than anticipated in Talbot's Branch (Figure 6 A-C). It was unclear why ^{15}N enrichment was low in Talbot's Branch. However, enrichment was still sufficient to evaluate uptake length (S_w) into periphyton and retention by seston and dominant consumers. S_w was shortest in Talbot's Branch (96 m), longest in Starmount GC (245 m), and intermediate in O'Henry (162 m) (Table 3). When differences in stream discharge and background concentrations of NH_4^+ were considered, however, the vertical mass transfer of NH_4^+ (v_f) was similar in restored Starmount and unrestored O'Henry, but still a little less than ½ that of forested Talbot's Branch (Table 3). In addition, background concentration of NH_4^+ was higher in Starmount GC than the other streams, likely reflecting fertilizer input from the golf course that it drained. Consequently, uptake per m^2 (U) was was somewhat higher in Starmount than Talbot's Branch and nearly 3-fold higher than in O'Henry.

Distribution of ^{15}N in seston with distance did not follow the same pattern as did periphyton. Seston was most highly retained at the O'Henry site, intermediate at Talbot's Branch, and was least retained at Starmount (Figure 7). This likely reflects that Starmount GC was a relatively straight run with higher discharge than the other streams, thus had higher transport of seston, whereas O'Henry had well-developed pools that would be expected to retain seston. Pools at

Talbot's Branch were less well developed, which likely accounted for greater transport of seston in that stream compared to unrestored O'Henry.

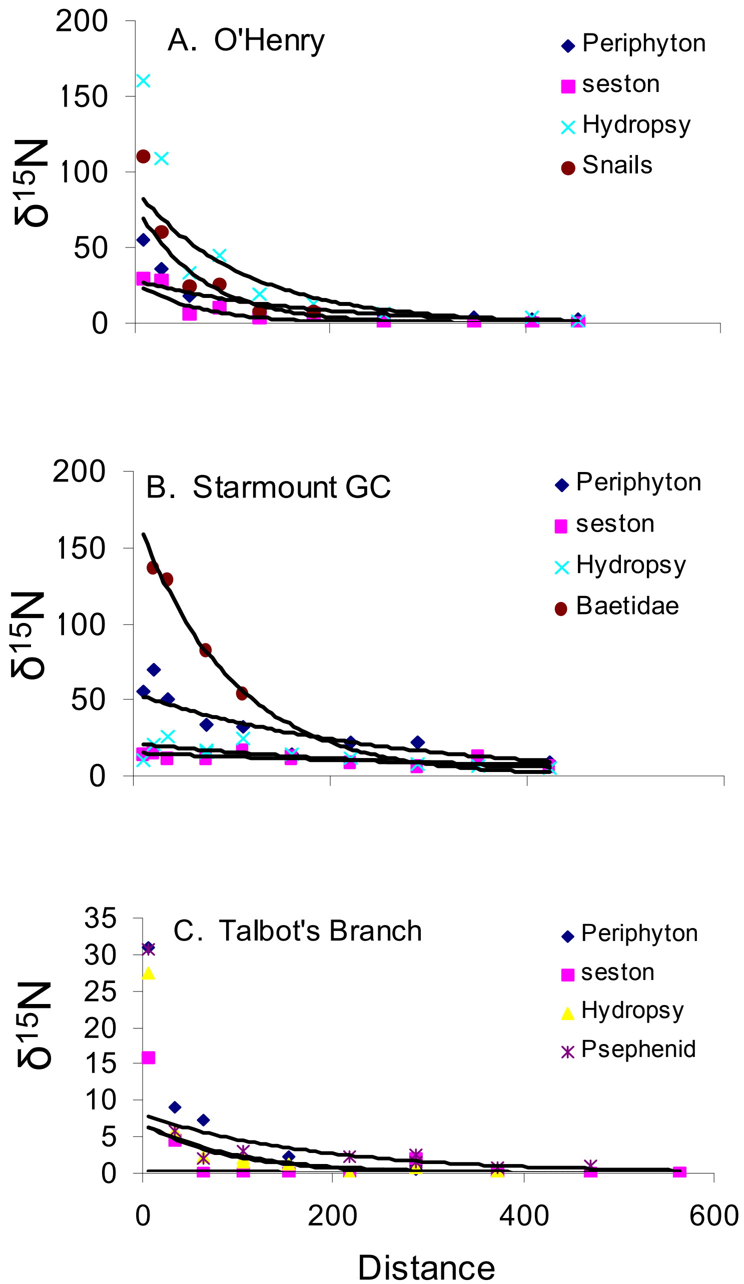

Figure 6. Exponential decay of δ ^{15}N in food web components of an urban unrestored, an urban restored, and a pristine forested stream following 4-wk of experimental ^{15}N addition.

Table 3. Uptake parameters for 15N experiments.

	Q (m^3/sec)	h (m)	k	S_w (m)	v_f (m/sec)	NH_4^+(mg/m^3)	U(mg/m^2)
O'Henry	0.00322	0.051	0.006154	162	0.000314	38	0.0118
Starmount	0.00635	0.075	0.004077	245	0.000306	98	0.0300
Talbot's Br	0.00235	0.07	0.010368	96	0.000726	35	0.02501

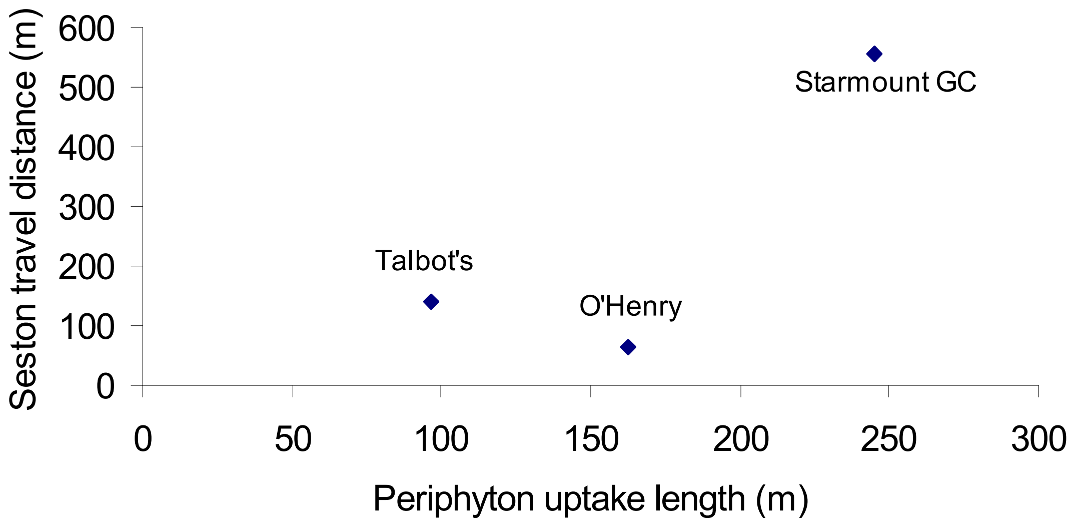

Figure 7. Travel distance of seston as a function of uptake length (S_w) to periphyton in ^{15}N tracer enrichment streams.

Hydropsychid caddisflies, which are net-spinning caddisflies generally considered to be collectors of suspended FPOM (seston), were the only consumer found at all sites. At all sites, hydropsychid became more enriched than seston (Fig. 6 A-C), their presumed food resource, suggesting some degree of selective assimilation of the primary producer components of the seston. At the O'Henry site, hydropsychids were also more highly labeled than was periphyton (Figure 6 A), suggesting an even greater degree of selective feeding or assimilation on periphyton.

Different taxa of grazers were dominant at each of the three sites. Physid snails at O'Henry and baetid mayflies at Starmount were the only grazers that could be collected from enough sites to measure ^{15}N retention relative to periphyton. Both of these grazers became more ^{15}N enriched than periphyton at the most upstream sites (Fig. 6 A-B), suggesting selective grazing or selective assimilation. Psephenid water pennies were similarly enriched as periphyton at Talbot's Branch, suggesting that they were assimilating bulk periphyton (Fig. 6C).

DISCUSSION

Riparian vegetation provides streams with shade that should promote cooler water temperatures, potentially resulting in higher DO concentrations. However, our restored sites, which included a re-vegetated buffer, did not differ in temperature from the unrestored sites which included a sparse or non-existent buffer; the DO trend line for surface water actually follows an inverse relationship with the temperature trend line. It is well established that, in the absence of other factors, DO solubility decreases with increasing water temperature. Therefore it is unlikely that water temperature accounts for observed differences in DO. It is also possible that a recovery period of 2-3 years is too short a time for riparian vegetation to become established enough to shade the stream. However, visual differences in vegetation development were apparent

Surface water DO concentrations were higher in restored than unrestored sites throughout the study period (Figure 2A). Streams that include heterogeneous streambed and in-stream structures provide differing flow regimes (Ferguson 1991, Fischenich and Allen 2000). The restored sites had riffle-pool sequences that the unrestored sites did not. Riffles promote aeration of the surface water while pools function in creating downwelling of the oxygen-rich surface water into the hyporheic zone (Eriksen et al. 1996, Boulton et al 1998, Jones et al. 1995). Thus, the addition of riffle-pools to the restored sites likely increased areas of surface-hyporheic water exchange, resulting in the observed higher DO concentrations in the hyporheic water of restored sites. Because surface-hyporheic exchanges are multidimensional, there are likely to be factors such as horizontal flows from streambanks, flooding and riparian transpiration also influencing hyporheic zone DO (Boulton et al. 1998, Jones et al. 1995, Duff and Triska 1990).

Surface water DO concentrations were lower upstream of the restored Starmount Park site and higher downstream of the restored Lindley Park site (Table 1). Because riffle-pool sequences were similar between reaches, differences in DO concentrations between the sites would indicate that factors other than the creation of riffle-pool sequences are also affecting DO. The restored sites included stabilized, re-vegetated streambanks that may have been preventing erosion and subsequent turbidity that otherwise would decrease DO in sites. However, if so, the effect was not sufficient to result in a difference in hyporheic DO concentrations between upstream and downstream of the restored sites.

Restored riparian vegetation should also reduce nutrient delivery from the watershed and provide detrital resources and habitat for macroinvertebrates. Riparian zones are important sites of NO_3^- retention, effectively trapping from 40% to over 90% of NO_3^- from runoff and subsurface flow (Lowrance et al. 1984 Peterjohn and Correll 1984, Cooper 1990, Osborne and Kovacic 1993, Hubbard and Lowrance 1994, Hill 1996, Daniels and Gilliam 1996, Lowrance et al. 1997). The restored sites in our study had an average of 8% lower surface water NO_3^- than did the unrestored sites (Figure 3C). However, because stormwater is routed directly into streams, bypassing riparian vegetation, it is unlikely that the re-vegetated riparian zone was responsible for intercepting much of the NO_3^- in surface runoff. It is likely that the re-vegetated riparian zones at the restored sites are retaining excess NO_3^- from below-ground transport, which may be contributing to the lower in-stream nitrate concentrations. Improved substrate conditions could also contribute to enhanced NO_3^- uptake within the reach. This reduced NO_3^- concentration is important because it shows that even in urban systems, mitigation can significantly reduce NO_3^- delivery to receiving waters, which is a management priority in the Cape Fear River Basin (NCDWQ 2000).

Stormwater routed into the streams would be expected to have very similar concentration of NH_4^+ between restored and unrestored sites. Once in the reach, NH_4^+ adsorbs onto stream channel sediments, which may function as transient storage pools (Triska et al 1994). Ammonium is released during the decomposition of organic material and nitrified in well-oxygenated waters (Duff and Triska 1990, Strauss and Lamberti 2002). Lack of an observed difference in surface water NH_4^+ concentrations (Figure 3B) indicates that nitrification and NH_4^+ sorption to channel sediments is similar between sites, or if not, the difference was not sufficient to overcome stormwater delivery.

Riparian vegetation also effectively retains much of the P in streams with forested and grassy buffers (Peterjohn and Correll 1984, Osborne and Kovacic 1993). However, a study of vegetated riparian buffers in the North Carolina Piedmont reported poor P retention capabilities for both forested and grassy buffers (Parsons et al. 1994). Sediment bound P from construction sites is the major source of PO_4^{-3} entering urban streams (Carpenter et al. 1988). Thus, the routing of stormwater directly into streams effectively bypasses any beneficial effect of buffers on sediment retention. The only potential beneficial effects of buffers on PO_4^{-3} reduction would be through plant uptake within the reach or soption onto allochthonous detritus, which clearly was not sufficient to reduce PO_4^{-3} at our sites.

In June, the unrestored sites showed spikes in NH_4^+ concentrations, and in August showed spikes in NH_4^+ and PO_4^{-3} concentrations compared to the restored sites. A sewer overflow was visually detected during sampling on both of these dates at one of the unrestored sites. The sewage water was spilling directly into the stream and would account for increased levels of NH_4^+ and PO_4^{-3} on that date.

Results from the comparison of stove-pipe samples between sites indicated that restoration had a small, positive effect on benthic macroinvertebrates. However, the majority of benthic metric comparisons were similar between sites, which suggest that the beneficial effects are being overridden by continuous urban impacts. Because there was no difference between sites in stream temperature, we believe temperature did not the drive the few community differences observed. A more heterogeneous substrate, re-vegetated riparian zone, higher DO and lower nitrate levels in the restored sites are factors that could have led to a positive benthic community response.

In March 2001, the N.C. DENR conducted qualitative sampling of Starmount Park one month post-restoration (N.C. DENR 2003). Their results indicated a tolerant community with decreased taxa richness after construction commenced. They stated that the tolerant benthic community within the restored reach was likely the result of nutrient enrichment from the adjacent golf course while the presence of a few intolerant taxa in the upstream reach was likely due to the forested riparian zone and heterogeneous bed substratum. Our results for the restored sites show increased taxa richness at 1 ½ to 2 ½ years post-restoration (Figure 4A) suggesting that the upstream reach is serving as a colonizing pool and that colonizers were positively responding to the restoration. The presence of a few intolerant taxa within the restored reach, such as *Eukieferella brehmi*, *Hydropsyche phalerata* and *Hydropsyche scalaris* indicates slightly improved water quality.

The results for the applied benthic metrics indicated that taxa richness was higher in restored than in unrestored sites in June 2003, but not on the earlier dates (Figure 4A). Case studies of urban stream restoration of Strawberry Creek, CA (Charbonneau and Resh 1992) and Baxter Creek, CA (Purcell et al 2002) showed increased taxa richness within three years post-restoration. Because stream recovery had reached 2 ½ years by June, the increased refugia, riparian vegetation, higher DO and lower nitrate levels in the restored sites may have provided a higher quality environment that promoted the recolonization and establishment of more taxa species than within the unrestored sites.

Corbicula fluminea is successful in disturbed habitats due to its short life span, early maturity, high fecundity, small juvenile size and capacity for downstream dispersal (McMahon 1991). In February, the mean # of *C. fluminea* was higher in the unrestored than in the restored sites, which may indicate that the unrestored sites were more unstable than the restored sites. Lower *C. fluminea* in restored sites might also have been due to insufficient time since construction for recolonization. Although they readily disperse to downstream habitats, the time required for such dispersal is probably longer than that for insects, which are more active in the drift and can fly.

The fact that there was no difference for each season in diversity, % Chironomidae and Oligochaeta, mean # Ephemeroptera and Trichoptera and taxa densities between the restored and unrestored sites indicates that these urban streams are composed of similar (mostly tolerant) communities, suggesting that urban impacts are overriding the local site improvements of restoration. Similarly, the MRPP analysis of benthic communities showed community heterogeneity similar to that expected by chance for all sites in February and in June, indicating no site-driven differences in community structure (Table 2). Results from the indicator value test produced no particular species functioning as indicators of any of the site conditions.

NCBI scores for water quality between sites never exceeded the Good-Fair ranking which further illustrates the fact that these streams are still degraded. The majority of water quality scores were Fair across the study season, which would explain the relative dominance of Oligocheata, a highly tolerant class, at all sites across the sampling period. The order Trichoptera is considered one of the three orders of intolerant macroinvertebrates, whose presence in a community should indicate good water quality (N.C. DENR 2003). The family, Hydropsychidae in this order is the exception, in that they are pollution tolerant (Loch et al. 1996). All Trichoptera taxa collected at all sites in this study were hydropsychids which suggests degraded water quality at all sites.

^{15}N enrichment experiments showed that the vertical mass transfer velocity of NH_4^+ to periphyton was greatest in Talbot's Branch, the relatively pristine forested stream. This result was expected because the urban stream reaches have high concentrations of DIN, thus demand for NH_4^+ should be lower in those streams. However, restored Starmount GC had similar NH_4^+ vertical mass transfer velocity and higher rate of NH_4^+ uptake per m^2 compared to unrestored O'Henry despite greater stream depth and higher ambient NH_4^+ concentrations. This result is consistent with the monitoring observation that nitrate concentration were lower in restored compared to unrestored reaches, and suggests that restoration is having a positive effect on retention of DIN. Note that we did not conduct pre-restoration studies in Starmount Golf Course, thus cannot conclude whether restoration improved N-retention at this site.

Both S_w for solutes and transport of seston is generally related to discharge in stream ecosystems (Wollheim 2001, Peterson et al. 2001) but is affected by stream morphometry. The result that seston travel distance was disproportionately long at Starmount GC suggests that although restoration improved retention of DIN, restoration efforts need to also include more attention to stream geometry to also retain particulate N.

Relatively few grazers and a very low diversity of grazers were collected from the urban ^{15}N tracer sites. Grazers were more enriched in ^{15}N than periphyton at both O'Henry and Starmount GC sites. This phenomenon also has been observed in some grazers during previous ^{15}N enrichment experiments in forested streams, and has been attributed to either selective grazing or selective assimilation of the periphyton, whereas other grazers assimilated bulk periphton (Tank et al. 2000, Rezanka and Hershey 2001), similar to what we observed for water pennies in Talbot's Branch. Selective grazing versus selective assimilation cannot be fully resolved with the data collected here or in previous studies. However, Rezanka and Hershey (2001) conducted a tile colonization experiment simultaneously with a ^{15}N enrichment and determined that tiles that had been colonized for shorter time periods were more enriched in ^{15}N than those colonized longer, suggesting that newer, surficial periphyton is more active in NH_4^+ assimilation than is more established bulk periphyton. They suggested that some grazers were grazing this more surficial material, which would lead to their higher level of enrichment relative to bulk periphyton.

Monitoring and ^{15}N enrichment experiments both suggested that restored streams were more effective at processing DIN. Restoration of this ecological function is of primary importance in protecting and restoring water quality in receiving waters in North Carolina. However, transport of particulate matter was disproportionately long in restored Starmount GC, which implies that restoration of channel morphometry was inadequate. In addition, there was minimal restoration of invertebrate communities at the restored sites. This result also implies that other aspects of water quality were not effectively restored.

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