INFLUENCE OF NATURE PRESERVES ON MACROINVERTEBRATE ASSEMBLAGES, WATER QUALITY, AND HABITAT QUALITY ALONG TWO URBAN STREAMS

BY

PATRICK M. WILKINS

THESIS

Submitted in partial fulfillment of the requirements
for the degree of Master of Science in Natural Resources and Environmental Sciences
in the Graduate College of the
University of Illinois at Urbana-Champaign, 2013

Urbana, Illinois

Master's Committee:

Yong Cao, Ph.D.
Jeffrey Levengood, Ph.D.
Edward Heske, Ph.D.
Assistant Professor Cory Suski, Ph.D.

Abstract

The increase in urbanization as a threat to aquatic ecosystems, particularly stream ecosystems, is of growing concern due to the reliance of the condition of surrounding watersheds. Mitigation techniques, such as riparian re-planting and habitat enhancement, are often met with little improvement in stream biodiversity. The understanding of the ecological effect of nature preserves in urban areas on stream assemblages can lead to more effective stream restoration. Benthic macroinvertebrates and water quality in two streams, Poplar Creek and Spring Creek in the Fox River watershed of northeastern Illinois, were monitored along an urban stream continuum upstream, within, and downstream of two nature preserves. Urban land cover within the riparian zone was considerably low at sites within (0.5%) the preserves compared to those outside (59.3%). Reductions in amount of silt substrates and increases in gravel-dominated substrate were evident within preserve sites in both streams. Taxa richness and a benthic macroinvertebrate index of biological integrity (MIBI) increased within the preserve sites for Poplar Creek, however there was no consistent improvement in biodiversity observed in Spring Creek. Additionally, percentage of Ephemeroptera, Trichoptera, and Plecoptera (%EPT) showed no improvements at sites located further downstream within the preserves compared to those outside or upstream in either stream. Water quality monitoring showed no consistent trends and seemed to be confounded by precipitation events between sampling occasions. Nature preserves in urbanized areas could increase macroinvertebrate biodiversity through local reductions in impervious surfaces and improvements in stream substrate based on responses in MIBI scores at Poplar Creek, despite lack of evidence that they could significantly improve water quality.

Acknowledgement

I would especially like to thank Drs. Yong Cao and Jeff Levengood for acting as my advisers, their committed guidance and assistance during the research and preparation of my thesis was greatly appreciated. I would also like to thank Dr. Ed Heske for providing valuable suggestions and comments through the project, especially during the early drafts of my writing. Thanks to other graduate students and technicians who committed their time and effort in the field to help collection of valuable data for the project.

TABLE OF CONTENTS

I. INTRODUCTION

The effects of watershed land covers and local habitats on stream ecosystem function and biodiversity has long been a focus of stream ecology (Hynes 1971, Vannote et al. 1980, Ward 1998, Weins 1989). Transition from natural to human-dominated landscapes typically degrades stream ecosystems at the microhabitat, reach, and network scales (Allan 2004, Paul and Meyer 2001). Streams draining urban areas show a flashy hydrograph, elevated nutrient and contaminant levels, homogenized channel morphology, and reduced biodiversity with increased dominance of pollutant-tolerant taxa (Booth and Jackson 1997, Meyer et al. 2005, Wang et al. 1997), a pattern of degradation referred to as the "urban stream syndrome" (Walsh et al. 2005). For example, changes in the hydrological regime occur as a result of more efficient transport of runoff due to increases in impervious surfaces and runoff from piped stormwater drainage systems (Dunne and Leopold 1978). Stormwater management, bank stabilization, channel reconfiguration, and riparian buffer replanting are common strategies for improving the physical and ecological conditions of degraded urban streams (Bernhardt and Palmer 2007). However, dense human infrastructure can limit the spatial extent of restoration options.

Macroinvertebrate assemblages reflect a multitude of physical, chemical, and biological stream features allowing them to be excellent indicators of stream health (Allen 2004). Sensitive species are absent or less abundant in nearly all studies of streams draining urban areas. Recent studies have shown that macroinvertebrates respond similarly to urbanization across regions, yet the specific environmental variables impacting invertebrates varied widely among metropolitan areas (Cuffney et al. 2010).

King et al. (2011) found sharp declines of numerous taxa in watersheds with impervious surface cover as low as 2%, indicating biological responses to urbanization may occur at levels lower than previously expected. Moore and Palmer (2005) also found that invertebrate diversity of headwater streams in suburban Maryland decreased with greater impervious surface cover in the watershed, but macroinvertebrate diversity was positively correlated with the amount of intact riparian vegetation around urban streams. Sudduth and Meyer (2006) reported that macroinvertebrate richness was strongly correlated with the percent of stream banks covered with roots or wood in urban and urban-restored streams. However, meta-analysis of 78 independent stream or river restoration projects found that only two showed statistically significant increases in biodiversity rendering them more similar to reference sites after restoring habitat heterogeneity (Palmer et al. 2010). Walsh (2004) reasoned that restoration projects completed at the reach scale do not consistently improve aquatic biodiversity probably because their beneficial impacts are not maintained unless effective measures are taken at larger spatial scales impacting stream systems.

As urbanization and associated landscape fragmentation continue globally, the use of forest fragments to help preserve stream biodiversity could be highly beneficial. However, the value of forested patches in aiding recovery of stream communities has been demonstrated in only a few studies. Studies in New Zealand evaluated shifts in macroinvertebrate composition and increases in diversity in streams in forest fragments within agriculturally dominated landscapes (Harding et al. 2006, Scarsbrook and Halliday 1999, Storey and Cowley 1997). They found that water quality was variable, and did not recover quickly to levels of control forest streams even as far as 350 m into forest fragments. Contrarily, benthic communities recovered to forested stream diversity and density levels without corresponding improvements in water quality. Houghton et al. (2011) monitored benthic macroinvertebrates and adult caddisflies along an agricultural stream upstream, within, and downstream of a small forested preserve in Michigan. They found that the diversity of adult caddisflies was significantly higher within the preserve, with a three-fold increase in species diversity despite no clear improvements in water quality.

Understanding the effectiveness of forested and grassland areas confined within urban regions as a biodiversity conservation tool is important in both freshwater and terrestrial fields alike. Few studies, if any, have examined the effects of terrestrial preserves on streams in an urbanized landscape. Evaluating the role of small, isolated natural habitats such as nature preserves in urban stream systems is particularly useful because such preserves offer the most common options for conservation in urbanized watersheds. A terrestrial preserve can provide allochthonous inputs, buffer water chemistries, reduce flow variation, and increase morphological stability, thus retaining greater aquatic biodiversity (Allan 2004). Evaluation of these potential benefits can be quantified through biological monitoring. Furthermore, managers of urban nature preserves need to know the spatial scales at which management is most effective. If stream biodiversity is most strongly affected by factors at the watershed scale, management options may be limited. If stream habitat quality and local riparian vegetation strongly affect stream biodiversity, then local management can be an effective conservation tool. Finally, the effect of water quality on stream biodiversity, relative to effects of habitat and landscape-level factors, should be evaluated in an urban stream. A longitudinal study of changes in stream biodiversity as a stream moves through a preserve can indicate distances over which positive changes occur, and thus thresholds in preserve size or structure for effective conservation.

This study had two objectives. First, I examined changes in macroinvertebrate assemblages, water quality, and habitat quality along two urban streams that travel through nature preserves in an otherwise urban landscape. I predicted that macroinvertebrate diversity would be greater, and include more species of conservation value, at sites downstream within preserves compared to sites within the preserves, but immediately adjacent to surrounding urbanized areas and sites outside the preserves altogether. I also predicted that improvements in water chemistry (e.g., total phosphorous concentration) and local habitat quality (e.g., substrate composition) would occur as streams travelled through the existing habitat remnants. Second, I modeled relationships among three macroinvertebrate metrics and environmental variables in the study streams to evaluate the relative effects of water quality, habitat quality, and land use on macroinvertebrate assemblages.

II. STUDY AREA, METHODS, AND DATA ANALYSIS

2.1 STUDY AREA

The Chicago Wilderness is a regional nature preserve system with more than 910 km^2 of protected natural areas that includes state parks, federal reserves, and county preserves in 7 counties in and around Chicago, Illinois. Many preserves are along or around rivers, creeks, and wetlands, and two of these (Poplar Creek Forest Preserve and Spring Creek Forest Preserve, both in Cook County) were selected for the present study.

These two second-order streams are located in west Chicago, Illinois (USA 42°4'N, 88°11'W) and are tributaries of the Fox River, which flows from southern Wisconsin through northeastern Illinois, draining 2429 km^2 in Wisconsin and 4454 km^2 in Illinois before joining the Illinois River (Fig. 1). Both are in primarily urban watersheds, and travel for 9.4 km and 10.6 km, respectively, through established terrestrial preserves (Poplar Creek Forest Preserve and Spring Creek Forest Preserve). Although they are named "forest preserves", significant portions of both preserves are grassland. Dominant non-forest vegetation types within the preserves were prairie, wet prairie, and marsh, with white oak (*Quercus alba*) and bur oak (*Quercus macrocarpa*) being the dominant mature woody vegetation.

2.2 STUDY DESIGN

An ideal design for capturing the potential recovery gradient of streams would be to select multiple sampling sites upstream, within, and downstream of an established preserve. However, constraints on access to sites in urban areas, particularly difficulties with access to private property and stream modifications that made sampling difficult, restricted our choices. As a result, seven sampling sites were selected on Poplar Creek, with four sites within the preserve, and three downstream of the preserve (Fig. 2). Six sites were selected on Spring Creek, with five sites located within the preserve boundaries and one site immediately upstream of the preserve boundary (Fig. 3). The sampling sites in a stream are more or less equally spaced to quantify the changes of stream conditions throughout and downstream of the preserve.

2.3 LAND COVER DATA

Land cover variables were extracted from the GIS database of the Illinois Department of Natural Resources (Brenden et al. 2004). The data set contains a variety of land cover categories (e.g., agriculture, wetland, inland lakes); urban, grassland, and forest cover were used in further analyses. Urban influence was quantified as the percent of land area covered by the urban category (%Urban), which included all impervious surfaces. Grassland and forest cover were pooled to create a single vegetative cover variable (%Veg.). Land cover at each site is summarized at two spatial scales: riparian buffer and watershed. Each stream reach was selected as a site, defined as 20 times the bankfull width of the stream. A riparian buffer was defined as the terrestrial landscape within 30 m of a stream reach, and this scale was intended to evaluate the effects of land cover type immediately adjacent to streams. The watershed level was defined as the total upstream land area draining into the most downstream point of the reach, and was intended to evaluate effects of inputs from the entire drainage area on a site. In addition, two levels of riparian buffer and watershed variables were used: reach and watershed. The boundaries for a reach riparian buffer (R) and reach watershed (W) extended as far as the boundaries of neighboring reaches (e.g., confluence to confluence, Fig. 4). Watershed riparian buffer (RT) and watershed variables (WT) were an aggregation of all riparian buffers and watershed level variables located upstream from the site of interest (Fig. 4). Thus, R_%Veg. represents the percent of vegetation at the reach level.

Each of the 4 upper-stream sites in Poplar Creek falls into a separate local watershed, but the last three sites fall into the same one. Land-cover data at a finer resolution would better differentiate the last three sites. All sites at Spring Creek fall into separate local watersheds.

2.4 SAMPLING METHODS

2.4.1 Water quality

Dissolved oxygen (DO.; mg L^{-1}), specific conductivity (SpC.; mS cm^{-1}), pH, turbidity (Turb.; NTUs), and salinity (Sal.; PSS) were measured *in situ* at all sampling sites using a portable Hydrolab Quanta Water Quality Monitoring System (Hach Environmental Inc., Loveland, CO). Water chemistry samples were taken following a standard protocol for biological water quality assessments (Hawkins et al. 2003). Two samples were collected on each sampling date. One 60 ml polyethylene bottle was filtered using a 47 mm, 0.45 micron nitrocellulose filter, and stored at 4° Celsius and analyzed for PO_4. A second 250 ml polyethylene bottle was left unfiltered, preserved with 0.02% sulfuric acid, stored at 4° Celsius and analyzed for total Kjeldahl nitrogen (TKN) and total phosphorus (TP). All water quality parameters were measured monthly June through September in 2010 and 2011 under base flow conditions. All analyses were performed using methods approved by the U.S. EPA (APHA 1998) at an analytical lab of the IL State Water Survey.

2.4.2 Physical Habitat Quality

Analysis of stream habitat conditions was conducted using a Qualitative Habitat Evaluation Index (QHEI; Rankin 1989). The QHEI is intended to provide a standard evaluation of the qualitative physical characteristics of a given stream reach, and is composed of six metrics that take into account six major habitat variables. At each site, I defined the evaluation area as 20 times the bankfull width of the stream. The six metrics included substrate, in-stream cover, channel morphology, riparian zone and bank erosion quality, pool and rifle quality, and stream gradient. The criteria of each metric were classified and recorded using the QHEI data sheet (Appendix A). Gradient was measured as elevation drop through the sampling area obtained from the GIS data of the sites. Qualitative assessments are made for a series of characteristics for each habitat variable to create a score for that variable. Scores are weighted based on the importance of each variable to stream health, then summed for an overall index of habitat quality. Maximum score for the QHEI is 100 (Appendix A).

2.4.3 Substrate Composition

Because substrate can have a strong effect on macroinvertebrate assemblages, I conducted an additional, more quantitative assessment of substrate composition using the Illinois Environmental Protection Agency wadeable streams transect approach (IEPA 1994). A sampling area was defined as ten times the channel width of the stream. Eleven transects were then spaced evenly throughout the sampling area, and the increments at which the substrate was sampled was based on the stream width at the given transect. Stream width in the study streams ranged from 1.31 – 10.97 m, requiring increment widths of 0.30 – 0.60 m. In each increment, a physical grab of the stream substrate and the depth at the given grab was recorded. Dominant-substrate type was recorded as silt/mud (< 0.062 mm), sand (0.062-2 mm), fine gravel (2.032-7.62 mm), medium gravel (7.63-15.24 mm), coarse gravel (15.25- 63.5 mm), small cobble (6.36-12.7 cm), large cobble (12.8-25.4 cm), or boulder (> 25.4 cm). Utilization of the habitat transect approach allows calculation of mean depth and width and determination of the dominant substrate type for each sample site. These physical measurements allow for actual values to be collected, as opposed to evaluation scores provided by the qualitative assessment of stream depth in the QHEI.

2.4.4 Macroinvertebrates

Macroinvertebrate samples were collected from all sites using standard D-nets following a standard multi-habitat sampling protocol (IEPA 2000). Two samples were collected at each site under baseflow conditions in June 2010 and 2011 in Poplar Creek, and September 2010 and June 2011 in Spring Creek. A total of 20 jabs with a D-net were allotted to existing habitat types (e.g., riffle, pool, and edge) at each site. The multi-habitat sampling approach minimizes the potential for among-site differences in habitat to be the cause of among-site differences in macroinvertebrate information (Chessman 1995). The materials collected were washed and put into sample jars, labeled, and preserved with 95% ethanol. In the lab, the samples were washed and evenly spread over a gridded plate. Macroinvertebrates were picked from each sample and identified to the lowest possible taxonomic level. Due to inconsistencies in taxonomic resolution as a result of organisms identified at different taxonomic levels (Cuffney et al. 2007), macroinvertebrates were converted to Operational Taxonomic Units (OTUs) (Appendix B). OTUs can vary in level of resolution, but are distinct from one another. This results in all taxa being classified to a consistent taxonomic level across samples.

I used three macroinvertebrate biotic metrics to quantify characteristics of stream assemblages and assess the effects of anthropogenic stressors. The relative responses of multiple metrics can provide a clearer representation of stream condition than one metric only, as responses across stressors may differ in severity (Yuan and Norton 2003). Taxa richness is a measure of the overall diversity of the benthic assemblage sampled and has been used to demonstrate relationships between diversity and stream condition (Berkman and Rabeni 1987, Richard et al. 1996). The macroinvertebrate index of biological integrity (MIBI, IL-EPA 2003) is a multi-metric index that has long been used in evaluating changes in macroinvertebrate assemblages due to human disturbance (Barbour et al. 1999, USEPA 2006, Wang et al. 2001). The MIBI is composed of seven metrics that are sensitive to human disturbances. Scores are compared to what is expected against a baseline condition that reflects little human impact. Percent Ephemeroptera, Plecoptera, and Trichoptera (EPT) is the total number of EPT individuals divided by the total number of individuals in a sample, and is a common metric that assesses the presence of sensitive taxa at a given site (Lenat 1993, Barbour et al. 1999).

2.5 DATA ANALYSIS

2.5.1 Longitudinal Changes in Macroinvertebrates and Habitat

I first examined the changes in the composition of macroinvertebrate assemblages among sampling sites using non-metric multidimensional scaling (NMDS) based on a Bray-Curtis similarity index and log(x+1) transformed data (Clarke and Gorley 2001). NMDS is an indirect gradient analysis that shows relationships of samples based on the rank-ordered similarity, i.e., more similar samples are closer in the ordination plot.

Because water quality variables often co-vary, the data on these variables (Table 1) were summarized with principal components analysis (PCA). Because the variables used in the PCA measured in various units and ranges, the PCA was conducted on the correlation matrix. PCA scores in the first two dimensions, together with QHEI, were then related to the location of the site to examine whether water quality and habitat quality changed in a predictable manner as the stream travelled through each forest preserve.

2.5.2 Modeling Changes in Macroinvertebrate Metrics

I used generalized linear models (GLMs) to evaluate the relative importance of different types of environmental variables for taxa richness, MIBI score, and %EPT taxa. First, sets of competing models were developed to identify the best models representing each of three competing but not mutually exclusive hypotheses: macroinvertebrate assemblages are affected by 1) water quality, 2) habitat quality, and 3) land cover. For purposes of evaluating the collective effects of water quality, habitat quality, and land cover, the next highest model was selected if the null model, which contained only the intercept, was the highest ranked model for a given environmental variable type. Second, the top-ranked models representing each hypothesis and all additive combinations of those models with the addition of a model representing distance downstream were then compared to evaluate which types of environmental variables (i.e., water quality, habitat quality, land cover, distance downstream) were the best predictors of each macroinvertebrate metric. Distance downstream was defined in kilometers from most upstream sampling site in each stream.

For water-quality-based modeling, I used the first two axes of the PCA on the collected water quality variables, singly and in an additive model, as competing water quality models. Before constructing models for habitat quality and land cover, collinearity among variables was assessed by constructing a correlation matrix; variables that were correlated at $r > 0.6$ were not included in the same model. Competing models for habitat quality included the QHEI, plus the quantitative measurements of mean wetted width, mean channel width, depth, %Sand (representing finer substrates), and %Coarse (sum of %Gravel and %Cobble, representing coarser substrates) alone and in all possible additive combinations. Competing models for land use included only %Veg variables (i.e., R_%Veg., W_%Veg., RT_%Veg., WT_%Veg.) because %Veg and %Urb variables were inversely related at all scales. Again, models included each variable alone and in all possible additive combinations. A Poisson family distribution was used for the taxa richness response variable (Guisan et al. 2002). %EPT taxa were transformed by arcsine ($\sqrt{[y/100]}$) prior to analysis. Model performance of individual models was evaluated using the coefficient of variation (R^2), the parameter estimate (β) of the variable of interest, and the 95% confidence interval of the parameter estimates. A model was proved to be insignificant if the confidence interval of the associated parameter estimate included zero.

I used an information criterion approach to model selection (Burnham and Anderson 2002) to rank the most important variables affecting each macroinvertebrate metric for each hypothesis (i.e., water quality, habitat quality, land cover). Models were ranked via the Akaike Information Criterion for small sample size (AIC_c). The top-ranked models for each type of environmental variable were then compared in a balanced design (i.e., each model separately and in all possible additive combinations) and Akaike model weights (w_i) were summed across all possible models to assess the relative importance of the three environmental variable categories (e.g. water quality, habitat quality, and land cover, distance downstream) included in the models.

All data analyses were conducted within the R statistical software (R Development Core Team 2011). The package 'vegan' was used to perform the multivariate analyses (Oksanen et al. 2012). The package 'MuMIn' was used to conduct model selection procedures for competing generalized linear models (Barton 2012).

III. RESULTS

3.1 LAND COVER AND LOCAL HABITATS

3.1.1 Poplar Creek

Percent vegetative and urban land cover varied from upstream to downstream of the preserve (Table 1). At the network watershed level, WT_%Veg. increased and WT_%Urban decreased as the stream travelled through the preserve, with the trends reversing slightly as the stream emerged from the preserve. Riparian cover at the network level varied little among sites, with a slight increase in RT_%Veg. from site P1 to site P4 and the inverse for RT_%Urban. Values of RT_%Veg. and RT_%Urban for sites downstream of the preserve were intermediate rather than higher or lower than sites within the preserve.

At the reach watershed level, sites P1 through P4 had high W_%Veg. due to their locations within the preserve, whereas sites NP5 through NP7 downstream of the preserve had low W_% Veg. W_%Urban varied considerably among sites within the preserve, but without regard to position downstream; as expected, values of W_%Urban were highest outside the preserve. R_%Veg. was much higher and R_%Urban lower at sites P1 through P4 than at sites outside the preserve, although there was no consistent longitudinal gradient for either metric within the preserve.

Wetted width and average depth increased slightly downstream (Table 2). Substrate composition was dominated by silt at site P1 but shifted to a gravel-dominated composition at sites P2 through NP6. Cobble was greater in the 3 sites downstream of the preserve, and site NP7 was dominated by cobble. QHEI scores increased greatly from site P1 to site P2, then remained high at all other sites except NP6. Although QHEI increased with distance downstream within Poplar Creek Forest Preserve and continued to increase through the first site downstream of the preserve, the overall relationship was not statistically significant (parameter estimate, β: 1.15; 95% confidence interval, CI: -0.56-2.87).

3.1.2 Spring Creek

WT_%Veg. increased along the gradient into the preserve, whereas WT_%Urban decreased (Table 1).). RT_%Veg. was high at all sites except S2, and RT_%Urban was low at all sites except S2 (Table 1).

At the reach watershed level, W_%Veg. was lower at upstream site NS1 and the first site within the preserve, S2, than at sites S3 through S6. W_%Urban was greatest at sites NS1 and S2 (Table 1). R_%Veg. was high at all sites except S2 and, surprisingly, S6 (farthest site into the preserve).

Wetted width increased downstream into the preserve (Table 2). Substrate composition varied considerably among sites, with the greatest amounts of silt at sites NS1 and S4 through S6, the greatest amounts of cobble at S2 and S3, and the greatest amount of gravel at S3. QHEI scores were lower at NS1 than at sites within the preserve. QHEI increased with distance downstream (β: 1.11), but the relationship was not significant (CI: -0.62-2.86).

3.2 WATER CHEMISTRY

3.2.1 Poplar Creek

All sites had pH levels slightly above 7 (Table 3). Salinity, turbidity, dissolved oxygen, and conductivity levels varied little among sites (Table 3). Nutrient concentrations varied among sampling occasions, but tended to decrease downstream (Table 3).

Principle components analysis yielded 2 axes (PCA1 and PCA2) that cumulatively explained 70% of the total variance in the selected water quality parameters (Table 4). PCA1 explained 42% of the total variance and had positive loadings from total phosphorus and TKN, whereas Sal and DO were negatively loaded. PCA2 explained 28% of the total variance, with pH loading positively and turbidity and specific conductivity loading negatively. PCA1 (β: -0.082, CI: -0.21-0.049) and PCA2 (β: -0.075, CI: -0.21-0.058) were not related to distance downstream.

3.2.2 Spring Creek

Dissolved oxygen and pH varied little among sites; salinity and specific conductivity tended to decrease downstream (Table 3). Variation in nutrient concentrations and turbidity did not correspond to position downstream through the preserve (Table 3). Principle components analysis yielded 2 axes that cumulatively explained 66% of the total variance in the selected water quality parameters (Table 4). PCA1 explained 35% of the total variance with salinity and specific conductivity positively loaded. PCA2 explained 31% of the variance; TP, TKN, DO, and pH all loaded positively, whereas PO_4 loaded negatively. PCA1 was weakly negatively related to distance downstream (β: -0.170, CI: -0.33-0.01). PCA2 (β: -0.046, CI: -0.24-0.15) was not related to distance downstream.

High inter-annual variation was exhibited among sites. Trends did not appear to be consistent between streams, indicating that hydrological variation and a complex urban setting confounded the ability to capture water quality trends between sampling occasions.

3.3 MACROINVERTEBRATE ASSEMBLAGES

Over 9500 benthic macroinvertebrates were identified in each stream during this study. A total of 31 taxa were collected in Poplar Creek, and 33 taxa were collected in Spring Creek (Appendix B). Of the 31 taxa identified from Poplar Creek, 9 were members of Ephemeroptera, Trichoptera, or Plecoptera (EPT). In Spring Creek, 11 of 33 taxa were members of EPT.

3.3.1 Poplar Creek.

Gradients in taxonomic composition among samples are shown via NMDS (Fig. 5).The best representation of the original data at Poplar Creek was found in two dimensions a stress value of 0.14 with a good relationship between original distance matrix and two ordination axes (Axis 1 R^2: 0.55, Axis 2 R^2: 0.56)). Axis 1 uncovered temporal differences in samples, indicating variability in taxonomic composition at sites from 2010 to 2011. Axis 2 appears to reveal a change in composition along the longitudinal gradient of selected sites in 2010; the same gradient was present but less pronounced in 2011.

3.3.2 Spring Creek

A good representation of the original data from Spring Creek was found in two dimensions with a stress value of 0.11 with a relationship between the original distance matrix and distance on the two ordination axes (Axis 1 R^2: 0.49, Axis 2 R^2: 0.49) . No longitudinal gradient in taxonomic composition among sites was captured by either axis. Most sites mapped in similar regions of the biplot in 2010 and 2011, but S6 differed strongly between years along NMDS1 and S3 differed strongly between years along NMDS2. The upstream site, NS1, was intermediate along both axes and mapped in the middle of the other sites.

3.4 MODELING THE VARIATION IN MACROINVERTEBRATE METRICS

3.4.1 Macroinvertebrate metrics

Taxa richness increased downstream in Poplar Creek, with the highest richness occurring at site NP5 in both years (Table 5). In comparison, taxa richness was similar at most Spring Creek sites although the highest richness did occur at the most downstream site (e.g. farthest into the Preserve) at Spring Creek in both 2010 and 2011 (Table 5).

MIBI scores ranged from 20.5 to 54.4 in Poplar Creek. Site P1 had the lowest scores in both years. MIBI scores tended to increase with position downstream in both 2010 and 2011; site NP7 had the highest score (Table 5). MIBI scores did not vary longitudinally at Spring Creek in either year (Table 5).

Percent EPT taxa ranged from 3.4 to 36.8 in Poplar Creek and 3.7 to 23.7 in Spring Creek. Site P4 had the highest percentage of EPT taxa in 2010 and site NP7 had the highest percentage in 2011 (Table 5). In Spring Creek, site NS1 had the lowest percentage and site S3 had the highest in 2010. In 2011, the lowest percentage occurred at site S2 and the highest occurred in the farthest downstream site, S6 (Table 5).).

3.4.2 Modeling macroinvertebrate metrics

Poplar Creek

Models based on PCA1 of water-quality variables performed better for all three macroinvertebrate metrics than PCA2-based models (β_{rich}: -0.10, CI: -0.20 – 0.0211, R^2: 0.18; β_{MIBI} -3.7, CI: -9.07 – 1.69, R^2: 0.14; $\beta_{\%EPT}$ -0.03, CI: -0.11 – 0.04, R^2: 0.07), although only slightly (ΔAICc < 2) for MIBI and %EPT. The additive models were not competitive for any metric (Table 6). The best model for habitat quality differed for each metric. QHEI (β: 0.01, CI: -0.001 – 0.016, R^2: 0.20) was the top-ranked habitat quality predictor for taxa richness (Table 7), although models based on average channel width or %Coarse substrate were competitive. The top-ranked habitat model for MIBI included additive effects of average channel width (β: 1.41, CI: 0.88 – 1.93) and QHEI (β:0.38, CI: 0.18 – 0.58) with an R^2 of 0.82 (Table 7). There was no top-ranked habitat quality model for %EPT, as the null model had the lowest AIC_c score. The model based on QHEI only (β: 0.01, CI: -0.001 – 0.010, R^2: 0.20) for %EPT taxa competed highly with the null (ΔAIC_c: 0.5; Table 7). For both taxa richness and MIBI scores, WT_%Veg. (β_{rich} : 0.03, CI: 0.006 – 0.059, R^2: 0.36; β_{MIBI} 2.07, CI: 1.45 – 2.67, R^2: 0.80, respectively) was selected as the top-ranked land-cover model, while the model including RT_%Veg. (β:0.04, CI: -0.009 – 0.0. R^2: 0.18) was the top-ranked land-cover model for %EPT taxa (Table 8). Distance downstream competed highly with the top-ranked models for both taxa richness and MIBI.

Land cover (WT_%Veg.) was the best approximating model for taxa richness and MIBI when comparing the top-ranked water quality and habitat quality (Table 9). There was no best approximating model for %EPT taxa as the null model had the lowest AIC_c score. No consistent longitudinal gradient was apparent in either stream in either year as distance downstream did not correlated well with %EPT in the models discussed below (Table 9). While top habitat-based models and land-cover-based models were competitive for %EPT (both with $\Delta AICc < 2$), water-quality-based models were ranked lowest for each metric (Table 9).

Cumulative weights ($\sum w_i$) for models based on water quality, habitat quality, land cover, and distance downstream confirmed that land cover was the factor that best predicted both taxa richness and MIBI whereas habitat quality was the factor that best predicted %EPT (Table 10).

Spring Creek

The null model was ranked the highest for taxa richness and MIBI score models and second highest for %EPT taxa (Table 9). In the %EPT taxa models, the land cover model depicted by W_%Veg. (β: 0.003, CI: 0.0002 – 0.006, R^2: 0.31) was ranked the highest, however ΔAIC_c of the null model was 0.73 (Table 9). The data collected for Spring Creek did not predict the variation exhibited by any of the three biological metrics well. The cumulative weights for models in each environmental variable category were spread more evenly among the three groups, however land cover was the factor that best predicted taxa richness and %EPT whereas habitat quality was the best predictor for MIBI (Table 10).

IV. DISCUSSION

4.1 PRESERVE IMPACTS

Physical and chemical stream conditions for biota were predicted to recover as the streams flowed from urbanized landscapes through forested nature preserves. However, the improvement varied between streams and among different measures of biological and physical conditions. Although two of three biological metrics used in this study improved as predicted at Poplar Creek, none of the three showed a consistent relationship with longitudinal position along Spring Creek. An index of habitat quality (QHEI) improved longitudinally within the preserve boundaries at Poplar Creek, but not at Spring Creek. Other variables affecting habitat quality, such as substrate type, did not vary consistently with location within either preserve. Water quality did not improve as streams travelled longitudinally within either forest preserve. These results provided some new insights into the effects of natural reserves on streams.

Vegetation in the watershed and in this case, the preserves, assimilates nutrients and blocks sediment from entering the stream (Paul and Meyer 2001). Although I found some decreases in nutrients in Poplar Creek as it flowed through a preserve, overall the two preserves studied did not measurably improve the water quality of the streams passing through them. A better reference site upstream of the preserve at Poplar Creek may provide more evidence that improvements occur at downstream locations within the preserve. These responses were consistent with previous studies on the impact of forest fragments on stream water quality. In these instances, water quality variables such as nitrogen and turbidity, reduced slowly with only small changes downstream of sites where shifts in the benthic communities were seen (Storey and Growley 1997, Scarsbrook and Halliday 1999). Temporal variation in water quality variables made it difficult to detect any longitudinal trends in water quality, and could overshadow any positive effects of the preserves. Additionally, water runoff from surrounding urbanized areas may be at such a level at which the benefits of a riparian forest to water quality are overshadowed. In previous studies, the effectiveness of riparian forests has been reported to be limited due to surrounding impervious surfaces and piped drainage systems, reducing the interaction between the riparian zone and pollutants moving in shallow groundwater from upland areas (Groffman et al. 2002). A continuous water quality monitoring protocol would be crucial in revealing improvements in water quality.

Despite lack of measurable improvement in water quality of the study streams, the preserves did appear to positively affect macroinvertebrates. Taxa richness and MIBI scores both increased in Poplar Creek as it travelled through the preserve. Improved values of these metrics were maintained downstream of the preserve in spite of the stream's return to more urbanized surroundings. Just as improvements in water quality resulting from changes in land use practices may require time and distance to emerge (Anbumozhi et al. 2004, Meals et al. 2010), benefits of the preserve to biological communities may persist downstream of its boundaries.

The lack of a relationship between %EPT and longitudinal position of a sampling site in our study suggests that these taxa may be affected more by specific habitat variables, such as substrate type and debris, than more tolerant macroinvertebrate taxa. Although certain taxa in the orders Ephemeroptera, Plecoptera, and Tricoptera are sensitive to stream conditions (Barbour et al. 1992) and many species in these orders are of conservation interest, %EPT may not be as good an index of overall stream quality in urban settings. %EPT abundance was low and varied annually at the sites sampled in both streams. As a result, a broader metric such as MIBI is more representative of changes in stream condition.

In contrast to Poplar Creek, Spring Creek exhibited no longitudinal trends in any of the macroinvertebrate metrics. Recovery of the stream appeared to be interrupted by several major roads that crossed the preserve. Previous studies have indicated that roads can be a source of perturbation and significantly alter macroinvertebrate assemblages compared to control locations away from road crossings (King et al. 2000). A detailed analysis of point influences on recovery of stream quality and macroinvertebrate communities would be enlightening.

Understanding the relative importance of land cover, water quality, and habitat quality in this study was critical in proposing what mechanism provided by the preserves was most beneficial to stream biodiversity. Modeling the relationships between the three macroinvertebrate metrics and water quality, habitat quality, and land cover suggested that land cover was the most important environmental influence on taxa richness and MIBI at Poplar Creek, whereas habitat quality had a slightly greater influence on %EPT. At Spring Creek, results were mixed, with land cover and habitat quality having similar, moderate influences on taxa richness, MIBI, and %EPT. It was clear, however, that water quality had the least influence on all three macroinvertebrate metrics in both streams. Change in the amount of impervious surface connected to the stream was the best single predictor of fish density, diversity, and biotic integrity across a gradient from predominantly agriculture to predominantly urban land in southeastern Wisconsin (Wang et al. 2001). While the importance of land cover was shown by the modeling approach in my study, the hierarchical nature of stream systems (Allan 2004) suggests that macroinvertebrate diversity also benefited from the habitat conditions, at least in Poplar Creek, which were expressed by the longitudinal changes in QHEI score.

4.2 LIMITATIONS ON INFERENCE

A main assumption in biomonitoring for the last several decades is 'the valley rules the stream' (Hynes 1975). Conditions at a given site reflect catchment conditions and land use upstream of that site. Changes in conditions are represented by changes in water quality and invertebrate composition. Studies have supported this assumption for quite some time (Allan 2004, Townsend et al. 2003, Vannote et al. 1980). The foundation of this assumption is the response of natural streams to a disturbance, not the response of an already disturbed stream to the presence of a remaining habitat fragment. Macroinvertebrates may not be able to respond readily due to lack of source populations and fragmentation in riparian zones (Allan 2004). Changes in water quality may not be evident due to excessive non-point source pollution (Palmer et al. 2010). Therefore, responses of both macroinvertebrate composition and water quality in an urban stream to a preserve may not occur as expected. These issues may warrant a more cautious approach when evaluating the changes in stream condition observed in this study. Small changes in stream condition or MIBI score, may be more critical than in situations in which stream condition is better compared to an appropriate reference site rather than observing changes in stream condition along a longitudinal gradient.

The dynamic nature of stream ecosystems makes them difficult to monitor, especially in urban environments (Walsh et al. 2005). Temporal differences in the hydrological regime of stream systems have contributed to significant reductions in species richness and changes in macroinvertebrate density in previous studies (McElravv et al. 1989). Limited inferences can be made about the changes in the hydrological regime in the study streams in each of the years sampled. As a result, the inter-annual variability in macroinvertebrate composition should be accounted for. The days since a peak flood event, prior to macroinvertebrate sampling, increased from 8 days in 2010 to 31 days in 2011. Large increases in stream discharge can displace many types of macroinvertebrates (Resh et al. 1988). The proximity of a large increase in flow to invertebrate sampling in 2010 may have attributed to the large amount of variability in macroinvertebrate composition between years.

Differences between preserves make it difficult to characterize consistent trends in both water quality and changes in macroinvertebrate assemblages. The size and shape of the preserves, in addition to the general differences between watersheds, are among the many factors that influence differences in how streams will respond to the presence of a preserve. The Poplar Creek preserve has less road crossings and only flows through one pond whereas multiple road crossings and multiple ponds and wetlands disrupt the continuum of Spring Creek.

Additionally, the sampling sites in each stream are located differently along the recovery gradient. Poplar Creek had sampling sites located within and downstream of the preserve whereas Spring Creek had sites both upstream and within the preserve. The low values of biological and habitat quality metrics at the single site upstream of the preserve at Spring Creek suggests that most sites within the preserve showed improved quality, but more comparisons to sites outside preserves, even if they can't be located immediately upstream of the study areas, would provide a better contrast of conditions within preserves to outside preserves. The differences in site location likely represent different positions along the potential recovery gradient in each stream, making it difficult to compare the changes in stream condition between the two streams.

The use of macroinvertebrates to assess biological diversity and conditions of a stream must be done carefully. Li et al. (2001) found that 70% of the variance in macroinvertebrate samples was associated with random spatial variation, and not associated with habitat type at the stream reach level. In my study the sampling was done at a local scale to detect changes within the preserve and the variability expressed in the macroinvertebrate data is a result of random spatial variation. It is possible the preserves had an influence on all sites and species dispersal from nearby source populations could be available to all reaches in the study.

4.3 MANAGEMENT IMPLICATIONS

Habitat heterogeneity can promote biotic recovery and biodiversity in stream systems and is often a goal of stream restoration (Harper et al. 1998; Palmer et al. 1997). The gradual improvement in many habitat quality variables (e.g., those combined and indexed in the QHEI) as Poplar Creek travelled through the preserve, and the greater values of QHEI within Spring Creek than at the site upstream of the preserve, suggest that even relatively small and isolated natural areas can have a valuable role in the conservation of aquatic communities. Improvement in the biodiversity metrics among sites in both streams were mixed, making it difficult to demonstrate the value of local stream habitat. However, in a study specifically examining the role of a small terrestrial preserve on macroinvertebrates, the diversity of adult caddisflies was three-fold higher within the preserve (Houghton et al. 2011).

My results suggest that increases in vegetation at the network watershed scale and presence of important habitat (e.g., coarse substrates) within the preserves are likely to promote macroinvertebrate diversity, without a corresponding improvement in water quality. The valley may rule the stream but my results indicate that local habitat quality still contributes substantially to stream communities in some situations. This should be a source of optimism for managers in highly urbanized landscapes with few options for "valley-scale" conservation. Management actions that focus on sustaining habitat quality in the preserves, including enhancing habitat heterogeneity, maintaining coarse substrates and debris, and reducing channelization (e.g., factors weighted positively in the QHEI), can make practical contributions to conservation of stream communities. Although managing land cover at watershed scales can be daunting, approaches that can be taken at a larger, watershed-wide scale should not be ignored when possible (Palmer et al. 2010).

Urbanization and fragmentation of natural vegetation will continue to impact the ecology of stream systems. Mitigating the effects of urbanization, specifically non-point source pollution, should continue to be an important focus of stream ecology research. The presence of remaining fragments and their influence on urban streams warrants further study as yet another management tool to help conserve aquatic biodiversity in urban areas.

SUMMARY

This study monitored two urban streams that travel through established nature preserves. The objective was to examine macroinvertebrate assemblages, water quality, and habitat quality with the prediction of observing improvements in stream condition in reaches of the streams further embedded within the preserve boundaries as opposed to reaches near the surrounding urban areas or outside the preserves entirely. Poplar Creek exhibited increases in gravel-dominated substrates at sites within the preserves, and its highest values of taxa richness and MIBI within the preserve boundaries. However, at Spring Creek, results were variable with no evidence in higher levels of macroinvertebrate biodiversity within preserve boundaries. Additionally, water quality was variable in both streams, making it difficult to infer preserve benefits on stream water quality. As growing urban areas continue to use forest and nature preserves as conservation measures, understanding their impact and effectiveness on the biological communities within them will become more and more important. Future studies should evaluate more robust study designs and improvements in monitoring intensity to capture the changes in streams within such dynamic environments.

REFERENCES

APHA (1998) Standard methods for the examination of water and wastewater, 20th ed. American Public Health Association, Washington, DC, USA

Akaike H (1973) Information theory as an extension of the maximum likelihood principle. In: Petrov BN, Csaki F (eds) Second international symposium on information theory. Budapest, Akaemiai Kiado, pp 267-281

Allan JD (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu Rev Ecol Syst 35: 257-284

Anbumozhi V, Radhakrishnan J, Yamaji E (2004) Impact of riparian buffer zones on water quality and associated management considerations. Ecol Eng 24: 517-523

Barbour MT, Graves CG, Plafkin JL, Wisseman RW, Bradley BP (1992) Evaluation of EPA's rapid bioassessment benthic metrics: metric redundancy and variability among reference stream sites. Env Toxicol Chem 11: 437-449

Barton K (2012) MuMIn: Model selection and model averaging based on information criteria. R package version 1.7.7. R Project for Statistical Computing, Vienna, Austria

Booth DB, Jackson CR (1997) Urbanization and aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. J Am Water Resour As 33: 311-323

Bekele EG, Knapp HV (2009) Hydrologic modeling of the Fox River watershed using SWAT2000: model development, calibration, and validation. Contract Report 2009-09. Center for Watershed Science, Illinois State Water Survey, Champaign, IL, USA

Bernhardt ES, Palmer MA (2007) Restoring streams in an urbanizing world. Freshwater Biol 52: 738-751

Berkman HE, Rabeni CF (1987) Effect of siltation on stream fish communities. Environ Biol Fish 18: 285-294

Brenden TO, Clark RD, Cooper AR, Seelbach PW, Wang L, Aichele SS (2006) Analyzing multi-scale landscape variables across large regions for river conservation and management. The American Fisheries Society. Herndon, VA, USA, Chapter 3, pp. 49-74

Burnham KP, Anderson DR (2002) Model selection and multimodal inference: a practical information-theoretic approach. Second edition. Springer, New York, New York, USA

Clarke KR, Gorely RN (2006) PRIMER v6: user manual/tutorial. PRIMER-E, Plymouth, UK

Chessman BC (1995) Rapid assessment of rivers using macroinvertebrates: a procedure based on habitat-specific sampling, family level identification and a biotic index. Aust J Ecol 20: 122-129

Cuffney TE, Bilger MD, Haigler AM (2007) Ambiguous taxa: effects on the characterization and interpretation of invertebrate assemblages. J N Am Benthol Soc 26: 286-307

Dunne T, Leopold LB (1978) Water in environmental planning. Freeman, NY, USA

Ellis JJ, Schneider DC (1997) Evaluation of a gradient sampling design for environmental impact assessment. Environ Monit Assess 48: 157-172

Fausch, DO, Karr JR, Yant PR (1984) Regional application of an index of biotic integrity based on stream fish communities. T Am Fish Soc 113: 39-55

Groffman PM, Bain DJ, Boulware NJ, Zipperer WC, Pouyat RV, Band LET, Colosimo MF (2002) Soil nitrogen cycle processes in urban riparian zones. Environ Sci Technol 36:4547-4552

Guisan A, Edwards TC, Hastie T (2002) Generalized linear and generalized additive models in studies of species distributions: setting the scene. Ecol Model 157: 89-100

Harding JS, Claassen K, Evers N (2006) Can forest fragments reset physical and water quality conditions in act as refugia for forest stream invertebrates? Hydrobiologia 568: 391-402

Harper D, Ebrahimnexhad M, Climent I, Cot F (1998) Artificial rifles in river rehabilitation: setting the goals and measuring the successes. Aquat Conserv 8: 5-16

Hawkins C, Ostermiller J, Vinson M, Stevenson RJ, Olson J (2003) Stream algae, invertebrate, and environmental sampling associated with biological water quality assessments. Department of Aquatic, Watershed, and Earth Resources, Western Center for Monitoring and Assessment of Freshwater Ecosystems, Utah State University, Logan, UT, USA

Houghton DC, Berry EA, Gilchrist A, Thompson A, Nussbaum MA (2011) Biological changes along the continuum of an agricultural stream: influence of a small terrestrial preserve and use of adult caddisflies in biomonitoring. J Freshwater Ecol 26: 381-397

Hynes HBN (1975) The stream and its valley. Edgardo Baldi Memorial Lecture. Verh Internat Verin Limnol 19: 1-15

Illinois Environmental Protection Agency (2000) Quality assurance and field methods manual. IEPA, Division of Water Pollution Control, Springfield, IL, USA

King RS, Baker ME, Kazyak PF, Weller DE (2011) How novel is too novel? Stream community thresholds at exceptionally low levels of catchment urbanization. Ecol Appl 21: 1659-1678

King RS, Nunnery KT, Richardson CJ (2000) Macroinvertebrate assemblage response to highway crossings in forested wetlands: implications for biological assessment. Wetl Ecol Manag 8: 243-256

Lepš J, Šmilauer P (2003) Multivariate analysis of ecological data using CANOCO. Cambridge University Press, Cambridge, UK

Li J, Herlihy A, Gerth W, Kaufmann P, Gregory S, Urquhart S, Larsen DP (2001) Variability in stream macroinvertebrates at multiple spatial scales. Freshwater Biol 46: 87-97

Mancini L, Formichetti P, Anselmo A, Tancionia L, Marchini S, Sorace A (2005) Bioloigcal quality of running waters in protected areas: the influence of size and land use. Biodivers Conserv 14: 351-364

McElravy EP, Lamberti GA, Resh VH (1989) Year-to-year variation in the aquatic macroinvertebrate fauna of a northern California USA stream. J N Am Benthol Soc 8: 51-63

Meals DW, Dressing SA, Davenport TE (2010) Lag time in water quality response to best management practices: a review. J Environ Qual 39: 85-96

Meyer JL, Paul MJ, Taulbee WK (2005) Stream ecosystem function in urbanizing landscape. J N Am Benthol Soc 24: 602-612

Montgomery DR, MacDonald LH (2002) Diagnostic approach to stream channel assessment and monitoring. J Am Water Resour As 38: 1-16

Moore AA, Palmer MA (2005) Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. Ecol Appl 15: 1169-177

Oksanen L (2001) Logic of experiments in ecology: is pseudoreplication a pseudoissue? Oikos 94: 27-38

Oksanen J, Blanchet G, Kindt R, Legendre P, Minch PR, O'Hara RB, Simpson GL, Solymos P, Stevens HH, Wagner H (2012) vegan: Community Ecology Package. R package version 2.0.3. R Project for Statistical Computing, Vienna, Austria

Palmer MA, Hakenkamp CC, Nelson-Baker K (1997) Ecological heterogeneity in streams: why variance matters. J N Amer Benthol Soc 16:189-202

Palmer MA, Menninger HL, Bernhardt E (2010) River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biol 55: 205-222

Paul MJ, Meyer JL (2001) Streams in the urban landscape. Annu Rev Ecol Syst 32: 333-365

R Development Core Team (2011) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria

Rankin ET (1989) The Qualitative Habitat Evaluation Index (QHEI): rationale, methods, and application. Ohio Environmental Protection Agency, Ecological Assessment Section, Division of Water Quality and Assessments, Columbus, Ohio, USA

Retzlaff R (2010) The Illinois Forest Preserve District Act of 1913 and the emergence of metropolitan park system planning in the USA. Plan Perspect 25: 433-455

Resh VH, Brown AV, Covich AP, Gurtz ME, Li HW, Minshall GW, Reice SR, Sheldon AL, Wallace JB, Wissmar RC (1998) The role of disturbance in stream ecology. J N Am Benthol Soc 7: 433-455

Richards C, Johnson LB, Host GE (1996) Landscape-scale influences on stream habitats and biota. Can J Fish Aquat Sci 53: 295-311

Scarsbrook MR, Halliday J (1999) Transition from pasture to native forest land=use along stream continua: effects on stream ecosystems and implications for restoration. New Zeal J Mar Fresh 33: 293-310

Sedell JR, Reeves GH, Hauer FR, Stanford JA, Hawkins CP (1990) Role of refugia in recovery from disturbances: modern fragmented and disconnected river systems. Environ Manage 14: 711-724

Storey RG, Cowley DR (1997) Recovery of three New Zealand rural streams as they pass through native forest remnants. Hydrobiologia 353: 63-67

Sudduth EB, Meyer JL (2006) Effects of bioengineering on bank habitat and macroinvertebrates in urban streams. Environ Manage 38: 218-226

Ter Braak CJF, Verdonschot PFM (1995) Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquat Sci 57: 255-289

Ter Braak CJF, Prenctice IC (2004) A Theory of Gradient Analysis. Adv Ecol Res 34: 235-282

Townsend CR, Doledec S, Norris R, Peacock K, Arbuckle C (2003) The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. Freshwater Biol 48: 768-85

Vannote RL, Minshall GW, Cummins KW, Sedell JR, Cushing CE (1980) The river continuum concept. Can J Fish Aquat Sci 37: 130-137

Ward JV (1998) Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Freshwater Biol 47: 517-539

Wang L, Lyons J, Kanehl P, Gatti R (1997) Influence of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries 22: 6-12

Wang L, Lyons J, Kanehl P (2001) Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environ Manage 28: 255-266

Walsh CJ, Leonard AW, Lardson AR, Fletcher TD (2004) Urban stormwater and the ecology of streams. CRC for Freshwater Ecology and CRC for Catchment Hydrology, Canberra, Australia

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP II (2005) The urban stream syndrome: current knowledge and the search for a cure. J N Am Benthol Soc 24: 706-723

Weins JA (1989) Spatial scaling in ecology. Funct Ecol 3: 385-397

Yuan LL, Norton SB (2003) Comparing responses of macroinvertebrate metrics to increasing stress. J N Am Benthol Soc 22: 308-322

FIGURES AND TABLES

Fig. 1 Fox River watershed with Poplar Creek and Spring Creek watersheds highlighted. Note that Spring Creek flows north and Poplar Creek flows southwest on this map. Inset of Fox River watershed in northeastern Illinois.

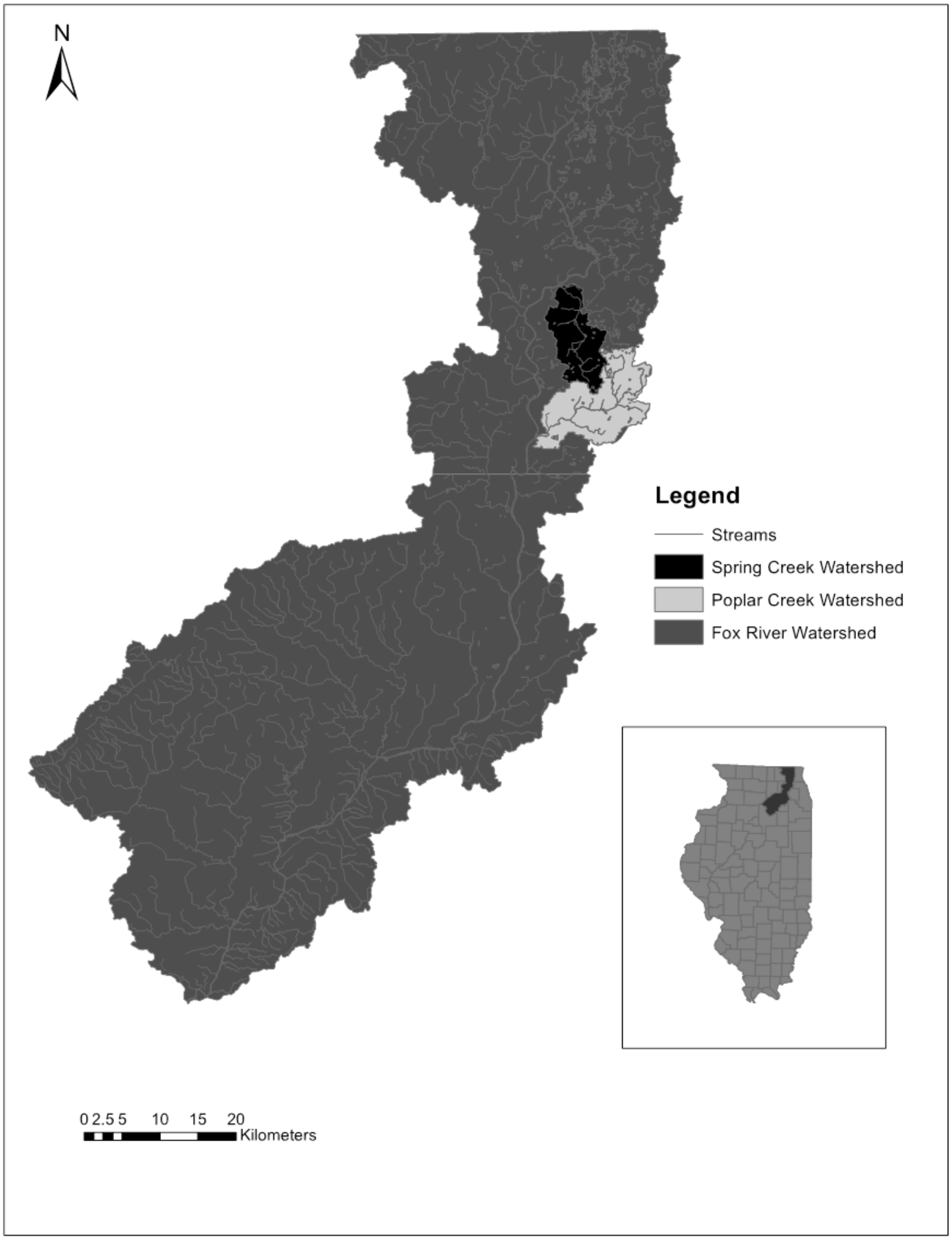


Fig. 2 Poplar Creek watershed and locations of 7 sampling sites.

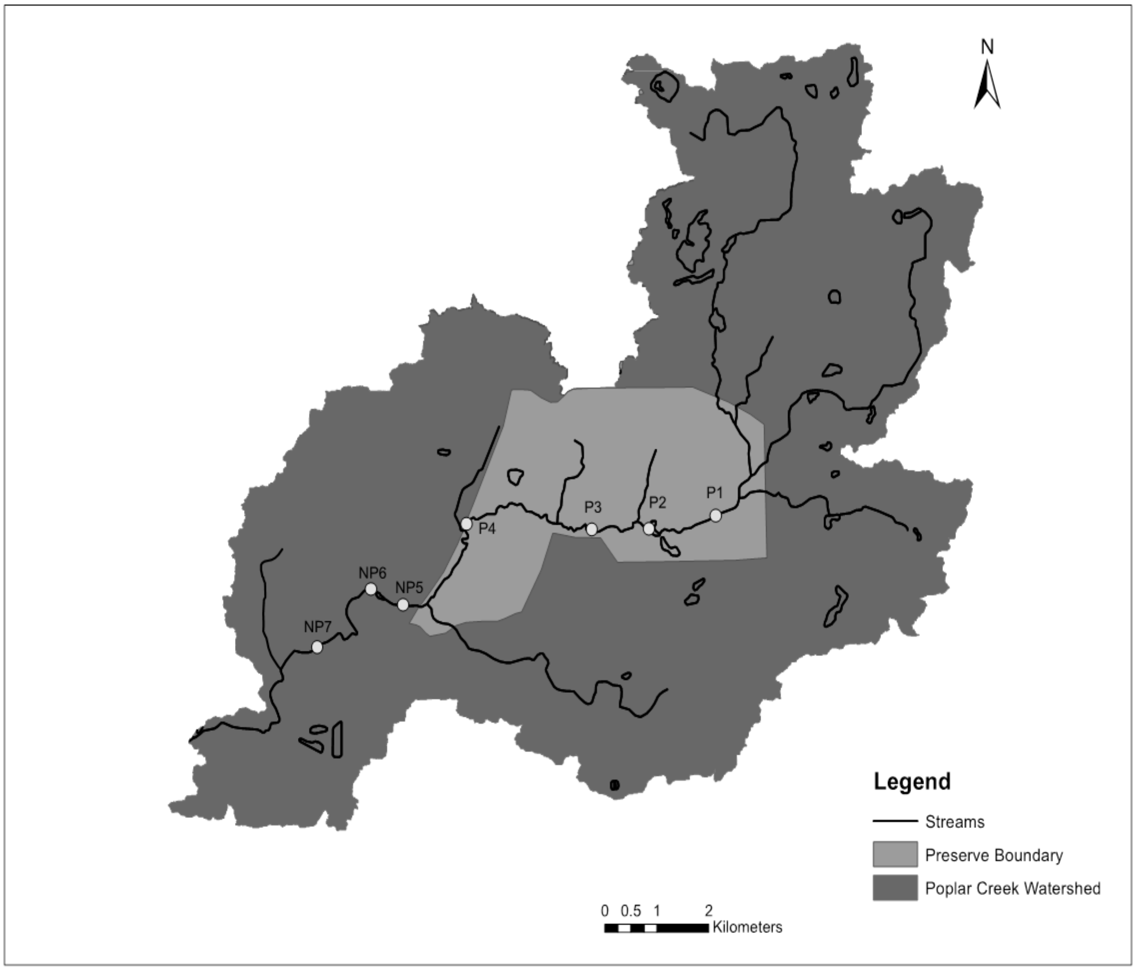


Fig. 3 Spring Creek watershed and locations of 6 sampling sites.

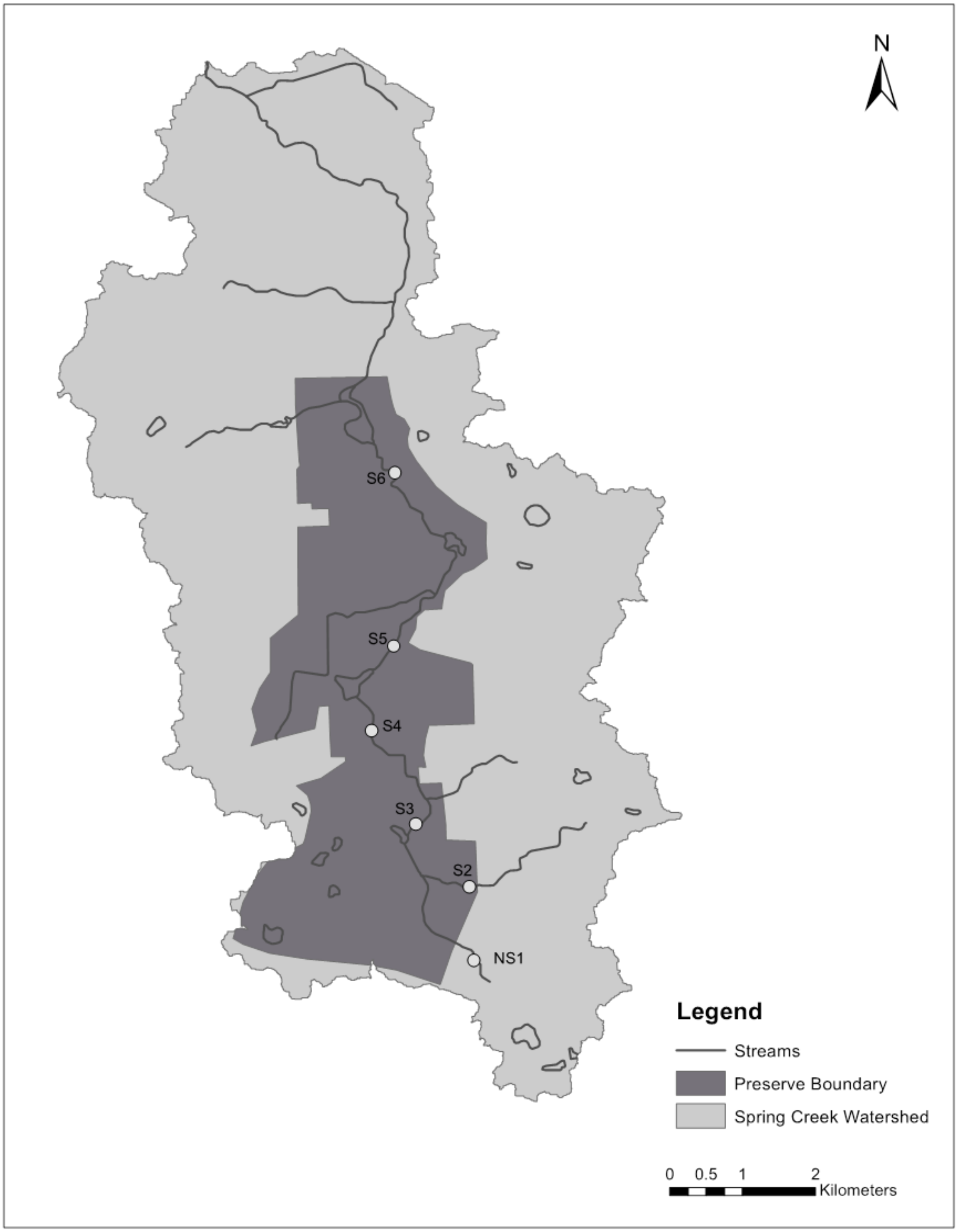


Fig. 4 Representation of the 2 land cover types used at the reach and network scales . (A) R: reach/local riparian (B) RT: total watershed riparian (C) W: reach/local watershed (D) WT: total watershed (after Brenden et al. 2006).

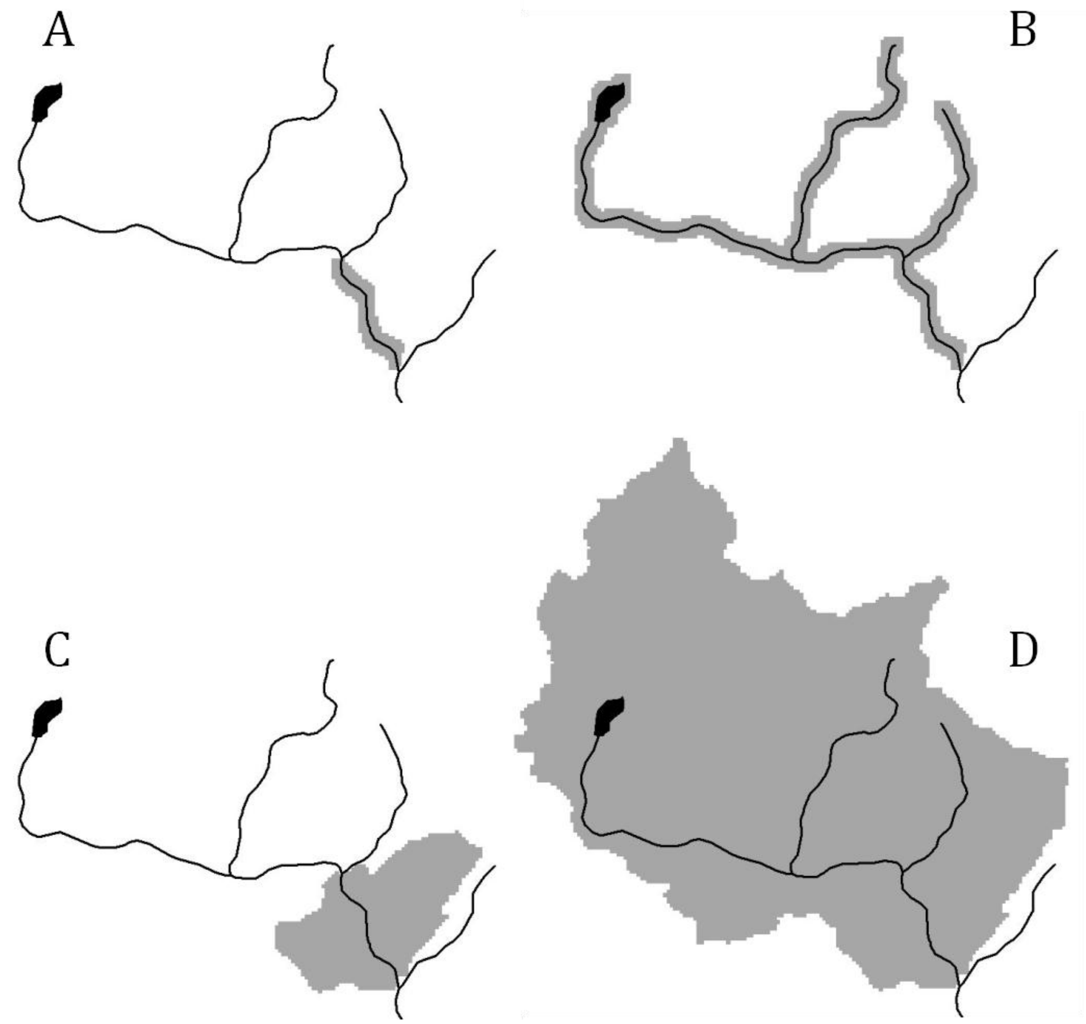
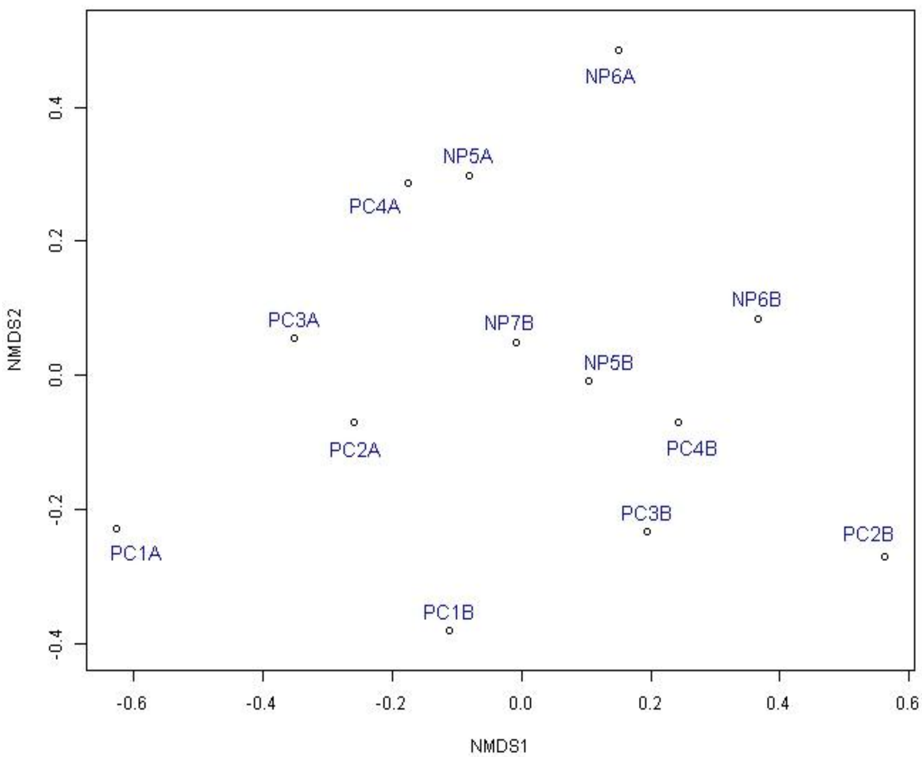


Fig. 5 Ordination plots of non-metric multidimensional scaling (NMDS) results. Poplar Creek- *top*; Spring Creek- *bottom*. A – *2010 data*; B – *2011 data* (PC & S – preserve sites; NP & NS – non-preserve sites).



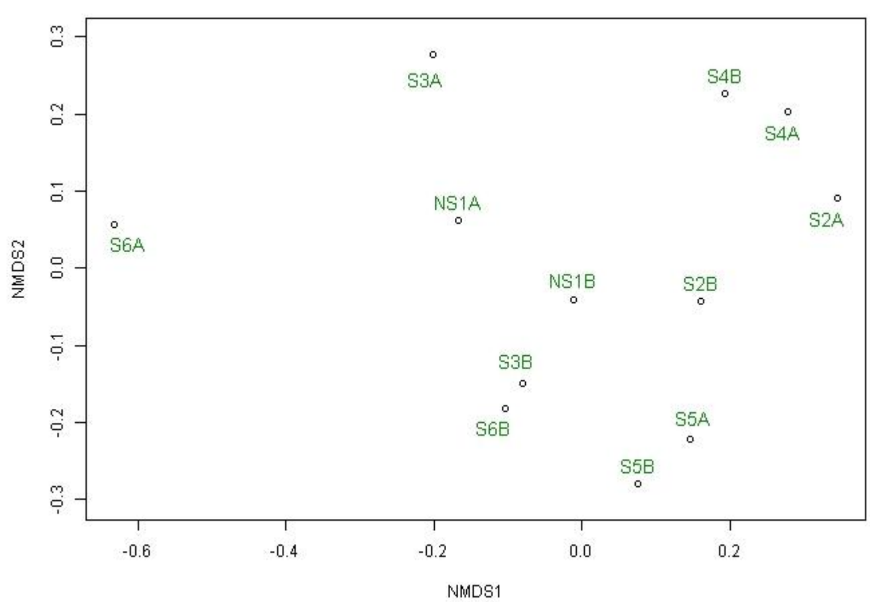


Table 1 Summary of distance downstream from most upstream site and percent land use for study sites in Poplar Creek and Spring Creek. Dist. downstream = distance from most upstream site in kilometers (km); %Veg = percent vegetative cover at each site; %Urban = urban land cover at each site. WT = total upstream watershed; W = local watershed; RT = total upstream riparian; R = local riparian (See Fig. 2 and Fig. 3 for site codes).

Site	Dist. Downstream	WT_%Veg.	W_%Veg.	RT_%Veg.	R_%Veg.	WT_%Urban	W_%Urban	RT_%Urban	R_%Urban
Poplar Creek									
P1	0	35.8	26	37	71.6	56	54.5	36.4	3.2
P2	1.6	35.9	71.4	37.8	85.7	56.1	4.3	35.6	1.8
P3	2.9	38.8	53.7	39.4	60.4	52.7	31.1	32.6	6.5
P4	6.6	44.9	83.2	42	69.8	48.7	9.8	29.6	3.2
NP5	9.4	45.2	12.8	39.6	28.7	51.7	80.9	34.7	59.3
NP6	10.2	45.2	12.8	39.6	28.7	51.7	80.9	34.7	59.3
NP7	12.5	45.2	12.8	39.6	28.7	51.7	80.9	34.7	59.3
Spring Creek									
NS1	0	46.8	32.2	76.7	76.7	29	29	3.8	3.8
S2	1.7	36.3	23	39.7	39.7	43.1	43.1	20.2	20.2
S3	2.5	70	80.1	62.8	91.5	25	10.4	10.5	6.2
S4	4.6	73.5	62	62.4	79.5	28.6	18.8	14.4	2.5
S5	6.2	77.8	62.7	63.7	81.9	26	12.3	11.8	3.4
S6	9.8	95.5	80.1	60.5	36.7	21	5.9	8.2	0.5

Table 2 Summary of selected habitat variables used for analyses. Values for selected channel measurements are in meters followed by minimum and maximum values.

Site	Mean Channel Width (m)	Mean Wetted Width (m)	Mean Depth (m)	Mean Thalweg Depth (m)	%Silt	%Gravel	%Cobble	QHEI
Poplar Creek								
P1	6.51 (4.27-7.62)	7.04 (6.10-7.62)	0.17 (0.02-0.55)	0.3	60.33	13.22	0	40
P2	6.95 (4.57-9.45)	6.95 (4.57-9.45)	0.14 (0-0.37)	0.21	15.13	75.63	5.04	68.5
P3	6.34 (4.57- 9.14)	6.13 (4.57-7.92)	0.13 (0-0.73)	0.24	15.93	61.95	8.85	71.5
P4	8.17 (6.10-10.10)	7.77 (6.10-9.45)	0.19 (0-0.40)	0.27	15.49	69.72	5.63	72.5
NP5	8.70 (5.79-11.28)	8.70 (5.79-11.28)	0.16 (0-0.40)	0.24	6.62	45.03	43.71	79
NP6	10.34 (10.06-10.97)	10.34 (10.06-10.97)	0.26 (0-0.49)	0.37	12.15	62.43	15.47	52
NP7	10.35 (5.6-12.50)	7.86 (3.32-11.19)	0.20 (0.02-0.58)	0.4	0	35.63	50	71.5
Spring Creek								
NS1	1.97 (1.31-2.38)	2.16 (1.31-2.35)	0.34 (0.23-0.69)	0.56	56.36	14.55	1.82	36.5
S2	2.74 (1.83-4.75)	2.29 (1.83-2.62)	0.17 (0-0.51)	0.51	9.09	24.68	24.68	63
S3	4.25 (3.05-4.36)	3.54 (2.44-6.40)	0.18 (0-0.43)	0.27	2.99	64.18	17.91	66.5
S4	5.39 (4.27-6.61)	4.30 (3.51-4.85)	0.18 (0-0.41)	0.3	38.2	35.96	0	57.5
S5	5.97 (3.05-7.92)	5.64 (3.05-7.92)	0.17 (0-0.41)	0.29	21.36	20.39	0	61
S6	5.71 (4.11-6.74)	5.71 (4.11-6.74)	0.34 (0.03-0.64)	0.45	40.63	16.67	0	58

Table 3 Summary of water quality parameters of sampling sites. Values for selected physiochemical parameters are averages followed by coefficient of variation (%).

Site	pH	Sal. (PSS)	Turb. (NTUs)	DO. (mg/L)	SpC. (mS/cm)	PO_4 (µg/L)	TP (µg/L)	TKN (µg/L)
Poplar Creek								
P1	7.9 (0.23)	0.48 (0.06)	14.4 (7.7)	7.7 (1.2)	0.95 (0.12)	20.4 (12.7)	95.0 (24.6)	935.6 (180.5)
P2	7.8 (0.33)	0.51 (0.04)	22.9 (7.6)	7.5 (0.9)	1.03 (0.09)	11.9 (6.9)	79.4 (35.6)	1004.4 (139.9)
P3	7.9 (0.28)	0.50 (0.08)	15.4 (2.5)	8.0 (0.8)	1.02 (0.14)	11.0 (4.9)	76.7 (25.3)	1022.8 (152.7)
P4	7.8 (0.32)	0.51 (0.07)	15.6 (9.2)	7.8 (0.8)	1.03 (0.14)	15.6 (11.0)	67.0 (31.2)	907.1 (141.9)
NP5	7.7 (0.21)	0.51 (0.06)	18.5 (7.2)	7.7 (0.7)	1.05 (0.11)	16.6 (9.7)	70.4 (31.2)	825.3 (277.9)
NP6	7.8 (0.21)	0.47 (0.19)	20.2 (7.5)	7.4 (1.0)	1.04 (0.15)	17.9 (8.7)	75.2 (33.9)	922.3 (409.7)
NP7	8.0 (0.25)	0.51 (0.10)	14.0 (7.8)	8.1 (0.8)	1.03 (0.17)	17.2 (14.9)	51.3 (15.9)	809.0 (207.0)
Spring Creek								
NS1	7.9 (0.21)	0.61 (0.13)	22.9 (31.2)	7.7 (1.3)	1.22 (0.25)	15.6 (7.6)	49.5 (16.3)	785.9 (202.4)
S2	7.8 (0.18)	0.86 (0.34)	21.3 (5.8)	7.6 (0.9)	1.70 (0.65)	13.8 (10.5)	54.6 (16.2)	900.3 (240.3)
S3	7.8 (0.32)	0.73 (0.61)	18.1 (8.1)	7.4 (1.7)	1.45 (1.13)	19.7 (7.9)	57.6 (28.4)	848.4 (187.1)
S4	8.1 (0.23)	0.59 (0.18)	26.0 (11.3)	8.6 (1.2)	1.18 (0.34)	10.9 (6.3)	80.1 (28.0)	1059.2 (139.6)
S5	7.8 (0.21)	0.57 (0.18)	19.9 (12.6)	6.9 (0.9)	1.15 (0.33)	25.1 (17.2)	68.0 (27.5)	887.9 (170.3)
S6	8.0 (0.14)	0.38 (0.19)	8.6 (3.8)	7.5 (0.5)	0.94 (0.07)	15.6 (11.2)	55.4 (33.0)	725.6 (64.7)

Table 4 Weights of individual water-quality variables on the first two axes of principal component analysis (PCA).

Poplar Creek	PCA1	PCA2	Spring Creek	PCA1	PCA2
%Variance	42	28	%Variance	35	31
Water Quality variables			*Water Quality variables*		
pH	-0.38	0.82	pH	-0.73	0.65
Sal	-0.81	-0.45	Sal	0.93	0.01
Turb	0.42	-0.75	Turb	0.47	0.49
DO	-0.83	0.38	DO	-0.24	0.64
SpC	-0.52	-0.75	SpC	0.87	-0.11
PO_4	0.40	0.33	PO_4	0.09	-0.76
TP	0.86	0.05	TP	0.25	0.52
TKN	0.75	0.02	TKN	0.56	0.75

Table 5 Summary of macroinvertebrate metrics. NP7 was not added until 2011

	Taxa Richness		MIBI Score		%EPT Taxa	
Site	2010	2011	2010	2011	2010	2011
Poplar Creek						
P1	20	21	20.5	34.6	8.9	10.2
P2	20	21	33.5	38.7	14.3	2.6
P3	17	28	35.6	43.5	23.4	8.4
P4	27	30	52.8	48.5	36.8	10.6
NP5	28	31	52.8	51.3	25.2	10.7
NP6	21	27	48.1	51.1	3.4	4.8
NP7	-	30	-	54.4	-	13.7
Spring Creek						
NS1	27	21	31.0	28.4	3.7	2.4
S2	30	25	49.9	30.7	15.5	1.9
S3	24	23	27.8	27.4	19.4	16.1
S4	30	21	53.8	35.9	13.2	4.4
S5	30	23	38.7	45.2	9.1	2.8
S6	33	30	45.4	39.4	7.4	23.7

Table 6 Competing water quality-based models for taxa richness, MIBI scores, and %EPT taxa in Poplar and Spring Creek. K = no. estimable parameters, L = likelihood of model, R^2: coefficient of determination, AIC_c = Akaike's Information Criterion corrected for small sample size, ΔAIC_c : AIC_i – minimum AIC_i. (-) = negative relationship with metric. Null model includes only the intercept as an explanatory variable.

Stream & Metric	Parameters	K	-2log(L)	R^2	ΔAIC_c	w_i
Poplar Creek						
	(-)PCA1	3	74.18	0.18	0.33	0.37
Richness	(-)PCA2	3	76.41	0.02	2.58	0.12
	(-)PCA1 +(-)PCA2	4	73.91	0.01	2.87	0.07
	Null	2	76.67	0.00	0.00	0.44
	(-)PCA1	3	94.32	0.14	0.96	0.27
MIBI	(-)PCA2	3	95.83	0.03	2.23	0.13
	(-)PCA1 +(-)PCA2	4	93.79	0.18	5.14	0.04
	Null	2	96.29	0.00	0.00	0.57
	(-)PCA1	3	-16.25	0.07	2.55	0.19
%EPT	(-)PCA2	3	-15.34	0.00	3.46	0.12
	(-)PCA1 +(-)PCA2	4	-16.25	0.07	6.88	0.02
	Null	2	-15.33	0.00	0.00	0.67
Spring Creek						
	(-)PCA1	3	67.37	0.07	2.02	0.22
Richness	(-)PCA2	3	68.04	0.02	2.67	0.16
	(-)PCA1 +(-)PCA2	4	67.11	0.09	5.41	0.04
	Null	2	68.30	0.00	0.00	0.59
	(-)PCA1	3	84.81	0.10	2.44	0.20
MIBI	(-)PCA2	3	86.02	0.00	3.65	0.11
	(-)PCA1 +(-)PCA2	4	84.80	0.10	7.14	0.02
	Null	2	86.04	0.00	0.00	0.67
	(-)PCA1	3	-17.65	0.11	2.33	0.21
%EPT	(-)PCA2	3	-16.50	0.02	3.48	0.12
	(-)PCA1 +(-)PCA2	4	-17.85	0.12	6.84	0.02
	Null	2	-16.31	0.00	0.00	0.66

Table 7 Competing habitat quality models for taxa richness, MIBI scores, and %EPT taxa in Poplar and Spring Creek. K = no. estimable parameters, L = likelihood of model, R^2: coefficient of determination, AIC_c = Akaike's Information Criterion corrected for small sample size, ΔAIC_c : AIC_i – minimum AIC_i . (-) = negative relationship with metric. Null model includes only the intercept as an explanatory variable.

Stream & Metric	Parameters	K	-2log(L)	R^2	ΔAIC_c	w_i
Poplar Creek						
	QHEI	2	73.76	0.20	0.00	0.10
	Ave. Chan. Wdth.	2	73.83	0.20	0.08	0.10
	%Coarse	2	74.17	0.18	0.41	0.08
	(-)%Detritus	3	74.42	0.16	0.66	0.07
Richness	Ave. Chan. Wdth + QHEI	4	71.24	0.34	0.95	0.06
	Ave. Thalweg Dpth + QHEI	4	71.30	0.34	1.00	0.06
	(-)%Detritus + Ave. Thalweg Dpth	4	71.95	0.31	1.66	0.04
	(-)%Detritus + Ave. Chan. Dpth	4	71.96	0.31	1.67	0.04
	Null	2	76.67	0.00	0.08	0.10
	Ave. Chan. Wdth. + QHEI	4	73.93	0.82	0.00	0.53
	Ave. Chan. Wdth.+ (-) Silt	4	78.43	0.75	4.50	0.06
MIBI	Ave. Chan. Wdth + QHEI + (-) %Coarse	5	73.12	0.83	4.77	0.05
	Null	2	96.29	0.00	14.56	0.00
	QHEI	3	-18.29	0.20	0.51	016
%EPT	QHEI + %Coarse	4	-21.68	0.39	1.45	0.10
	(-)%Detritus	3	-17.17	0.13	1.63	0.09
	Null	2	-15.33	0.00	0.00	0.21
Spring Creek						
	Detritus	3	67.13	0.09	1.76	0.10
Richness	Ave. Water Wdth.	3	67.34	0.08	1.97	0.09
	Null	2	68.30.	0.00	0.00	0.24
	Ave. Chan. Wdth.	3	83.18	0.21	0.81	0.15
MIBI	QHEI + (-)%Coarse	4	79.51	0.42	1.85	0.09
	Null	2	86.03	0.00	0.00	0.22
	QHEI	3	-19.62	0.24	0.35	0.18
%EPT	%Coarse	3	-18.79	0.19	1.18	0.12
	Null	2	-16.31	0.00	0.00	0.21

Table 8 Competing land cover models for taxa richness, MIBI scores, and %EPT taxa in Poplar and Spring Creek. K = no. estimable parameters, L = likelihood of model, R^2: coefficient of determination, AIC_c = Akaike's Information Criterion corrected for small sample size, ΔAIC_c : AIC_i – minimum AIC_i . (-) = negative relationship with metric. Null model includes only the intercept as an explanatory variable.

Stream & Metric	Parameters	K	-2log(L)	R^2	ΔAIC_c	w_i
Poplar Creek						
Richness	WT_%Veg.	3	70.78	0.36	0.00	0.34
	RT_%Veg.	3	72.51	0.27	1.73	0.14
	Null	2	76.67	0.00	3.05	0.07
MIBI	WT_%Veg.	3	75.29	0.80	0.00	0.59
	WT_%Veg. + R_%Veg.	4	74.51	0.81	3.55	0.10
	Null	2	96.29	0.00	17.53	0.00
%EPT	RT_%Veg.	3	-17.99	0.18	0.81	0.23
	W_%Veg.	3	-16.25	0.07	2.55	0.10
	Null	2	-15.33	0.00	0.00	0.35
Spring Creek						
Richness	(-)R_%Veg.	3	-66.02	0.17	0.66	0.30
	WT_%Veg.	3	-67.60	0.06	2.23	0.22
	Null	2	-68.30	0.00	0.00	0.10
MIBI	(-)R_%Veg.	3	84.52	0.12	2.15	0.14
	(-)RT_%Veg.	3	84.79	0.10	2.41	0.12
	Null	2	86.04	0.00	0.00	0.40
%EPT	W_%Veg.	3	-20.71	0.31	0.73	0.23
	WT_%Veg.	3	-18.35	0.16	2.35	0.10
	Null	2	-16.31	0.00	0.00	0.34

Table 9 Competing set of top candidate models for water quality (WQ), habitat quality (HQ), and land cover (LC) for each metric in Poplar and Spring Creek. K = no. estimable parameters, L = likelihood of model, AIC_c = Akaike's Information Criterion corrected for small sample size, ΔAIC_c : AIC_i – minimum AIC_i and w_i are Akaike weights. (-) = negative relationship with metric.

Stream & Metric	Model	Parameters	k	-2logLik	ΔAICc	w_i
Poplar Creek						
Richness						
	WQ	(-)PCA1	3	74.17	3.38	0.053
	HQ	QHEI	3	73.76	2.97	0.066
	LC	WT_%Veg.	3	70.79	0.00	0.290
	DIST	DistDwnst	3	71.76	0.98	0.178
	WQ + HQ	(-)PCA1 + QHEI	4	72.43	5.11	0.023
	WQ + LC	(-)PCA1 + WT_%Veg.	4	70.09	2.77	0.073
	WQ + DIST	(-)PCA1 + DistDwnst	4	71.10	3.78	0.044
	HQ + LC	QHEI + WT_%Veg.	4	70.12	2.79	0.072
	HQ + DIST	QHEI + DistDwnst	4	70.76	3.44	0.052
	LC + DIST	WT_%Veg. + DistDwnst	4	70.77	3.44	0.052
	WQ + HQ + LC	(-)PCA1 + QHEI + WT_%Veg.	5	69.62	6.64	0.011
	WQ + HQ + DIST	(-)PCA1 + QHEI + DistDwnst	5	70.34	7.35	0.007
	WQ + LC + DIST	(-)PCA1 + WT_%Veg. + DistDwnst	5	70.00	7.01	0.009
	HQ + LC + DIST	QHEI + WT_%Veg. + DistDwnst	5	70.11	7.12	0.008
	WQ + HQ + LC + DIST	(-)PCA1 + QHEI + WT_%Veg. + DistDwnst	6	69.57	12.15	0.001
	-	Null	2	76.67	3.05	0.063
MIBI						
	WQ	(-)PCA1	3	94.32	19.03	0.000
	HQ	Ave. Chan. Wdth. + QHEI	3	73.93	2.97	0.106
	LC	WT_%Veg.	3	75.29	0.00	0.468
	DIST	DistDwnst	3	77.27	1.98	0.174
	WQ + HQ	(-)PCA1 + Ave. Chan. Wdth. + QHEI	4	73.78	8.39	0.007
	WQ + LC	(-)PCA1 + WT_%Veg.	4	74.64	3.68	0.074
	WQ + DIST	(-)PCA1 + DistDwnst	4	76.95	5.99	0.023
	HQ + LC	Ave. Chan. Wdth. + QHEI + WT_%Veg.	5	69.84	4.45	0.051
	HQ + DIST	Ave. Chan. Wdth. + QHEI + DistDwnst	5	72.62	7.24	0.013
	LC + DIST	WT_%Veg. + DistDwnst	4	74.60	3.64	0.076
	WQ + HQ + LC	(-)PCA1 + Ave. Chan. Wdth. + QHEI + WT_%Veg.	6	69.67	11.71	0.001
	WQ + HQ + DIST	(-)PCA1 + Ave. Chan. Wdth. + QHEI + DistDwnst	6	72.57	14.61	0.000
	WQ + LC + DIST	(-)PCA1 + WT_%Veg. + DistDwnst	5	74.11	8.72	0.006
	HQ + LC + DIST	Ave. Chan. Wdth. + QHEI + WT_%Veg. + DistDwnst	6	69.80	11.85	0.001
	WQ + HQ + LC + DIST	(-)PCA1 + Ave. Chan. Wdth. + QHEI + WT_%Veg. + DistDwnst	7	69.60	22.04	0.000
	-	Null	2	96.29	17.53	0.000

(Table 9 cont.)

%EPT					
WQ	(-)PCA1	3	-16.25	2.55	0.079
HQ	QHEI	3	-18.29	0.51	0.220
LC	RT_%Veg.	3	-17.99	0.81	0.190
DIST	DistDwnst	3	-15.39	3.41	0.052
WQ + HQ	(-)PCA1 + QHEI	4	-18.57	4.57	0.029
WQ + LC	(-)PCA1 + RT_%Veg.	4	-18.31	4.83	0.029
WQ + DIST	(-)PCA1 + DistDwnst	4	-16.26	6.87	0.009
HQ + LC	QHEI + RT_%Veg.	4	-19.04	4.10	0.037
HQ + DIST	QHEI +DistDwnst	4	-18.49	4.65	0.028
LC + DIST	RT_%Veg. + DistDwnst	4	-18.91	4.22	0.034
WQ + HQ + LC	(-)PCA1 + QHEI + RT_%Veg.	5	-19.20	9.50	0.002
WQ + HQ + DIST	(-)PCA1 + QHEI +DistDwnst	5	-18.94	9.77	0.002
WQ + LC + DIST	(-)PCA1 + RT_%Veg. +DistDwnst	5	-19.60	9.11	0.003
HQ + LC + DIST	QHEI + RT_%Veg. +DistDwnst	5	-20.15	8.55	0.004
WQ + HQ + LC + DIST	(-)PCA1 + QHEI + RT_%Veg. +DistDwnst	6	-20.62	15.52	0.000
-	Null	2	-15.33	0.00	0.285

Spring Creek					
Richness					
WQ	(-)PCA1	3	67.37	2.00	0.094
HQ	Detritus	3	67.13	1.76	0.106
LC	(-)R_%Veg.	3	66.03	0.66	0.184
DIST	DistDwnst	3	66.32	0.95	0.159
WQ + HQ	(-)PCA1 + Detritus	4	67.03	5.33	0.018
WQ + LC	(-)PCA1 + (-)R_%Veg.	4	65.64	3.94	0.036
WQ + DIST	(-)PCA1 + DistDwnst	4	66.28	4.58	0.026
HQ + LC	Detritus + (-)R_%Veg.	4	65.44	3.74	0.039
HQ + DIST	Detritus + DistDwnst	4	66.30	4.60	0.026
LC + DIST	(-)R_%Veg. + DistDwnst	4	65.21	3.51	0.044
WQ + HQ + LC	(-)PCA1 + Detritus + (-)R_%Veg.	5	65.42	8.44	0.004
WQ + HQ + DIST	(-)PCA1 + Detritus + DistDwnst	5	66.20	9.21	0.003
WQ + LC + DIST	(-)PCA1 + (-)R_%Veg. + DistDwnst	5	65.19	8.20	0.004
HQ + LC + DIST	Detritus + (-)R_%Veg. + DistDwnst	5	65.21	8.22	0.004
WQ + HQ + LC + DIST	(-)PCA1 + Detritus + (-)R_%Veg. + DistDwnst	6	65.18	14.48	0.000
-	Null	2	68.30	0.00	0.255

(Table 9 cont.)

Stream & Metric	Model	Parameters	k	-2logLik	ΔAICc	w_i
MIBI						
	WQ	(-)PCA1	3	84.81	2.44	0.079
	HQ	Ave. Chan. Wdth.	3	83.18	0.81	0.179
	LC	(-)R_%Veg.	3	84.52	2.15	0.092
	DIST	DistDwnst	3	82.85	0.48	0.211
	WQ + HQ	(-)PCA1 + Ave. Chan. Wdth.	4	82.77	5.12	0.021
	WQ + LC	(-)PCA1 + (-)R_%Veg.	4	83.71	6.05	0.013
	WQ + DIST	(-)PCA1 + DistDwnst	4	82.80	5.15	0.021
	HQ + LC	Ave. Chan. Wdth. + (-)R_%Veg.	4	80.74	3.09	0.057
	HQ + DIST	Ave. Chan. Wdth. + DistDwnst	4	82.74	5.09	0.021
	LC + DIST	(-)R_%Veg. + DistDwnst	4	82.32	4.66	0.026
	WQ + HQ + LC	(-)PCA1 + Ave. Chan. Wdth + (-)R_%Veg.	5	80.67	9.30	0.003
	WQ + HQ + DIST	(-)PCA1 + Ave. Chan. Wdth + DistDwnst	5	82.62	11.25	0.001
	WQ + LC + DIST	(-)PCA1 + (-)R_%Veg. + DistDwnst	5	82.27	10.90	0.001
	HQ + LC + DIST	Ave. Chan. Wdth + (-)R_%Veg. + DistDwnst	5	78.87	7.50	0.006
	WQ + HQ + LC + DIST	(-)PCA1 + Ave. Chan. Wdth + (-)R_%Veg. + DistDwnst	6	76.68	14.11	0.000
	-	Null	2	86.04	0.00	0.269
%EPT						
	WQ	(-)PCA1	3	-17.65	3.06	0.063
	HQ	QHEI	3	-19.62	1.08	0.168
	LC	W_%Veg.	3	-20.71	0.00	0.29
	DIST	DistDwnst	3	-17.64	3.07	0.062
	WQ + HQ	(-)PCA1 + QHEI	4	-22.44	2.98	0.065
	WQ + LC	(-)PCA1 + W_%Veg.	4	-20.85	4.57	0.03
	WQ + DIST	(-)PCA1 + DistDwnst	4	-18.06	7.36	0.007
	HQ + LC	QHEI + W_%Veg.	4	-22.05	3.37	0.054
	HQ + DIST	QHEI + DistDwnst	4	-20.00	5.43	0.019
	LC + DIST	W_%Veg. + DistDwnst	4	-20.83	4.59	0.029
	WQ + HQ + LC	(-)PCA1 + QHEI + W_%Veg.	5	-23.10	8.61	0.004
	WQ + HQ + DIST	(-)PCA1 + QHEI + DistDwnst	5	-22.84	8.86	0.003
	WQ + LC + DIST	(-)PCA1 + W_%Veg. + DistDwnst	5	-21.14	10.57	0.001
	HQ + LC + DIST	QHEI + W_%Veg. + DistDwnst	5	-22.28	9.43	0.003
	WQ + HQ + LC + DIST	(-)PCA1 + QHEI + W_%Veg. + DistDwnst	6	-24.37	16.13	0.000
	-	Null	2	-16.31	0.73	0.201

Table 10 Cumulative Akaike weights (w_i) of the model groups for each biological metric in the three environmental variable categories.

Stream & Metric	Water Quality w_i	Habitat Quality w_i	Land Cover w_i	Distance Downstream w_i
Poplar Creek				
Taxa Richness	0.221	0.240	0.516	0.351
MIBI	0.111	0.166	0.677	0.293
%EPT	0.149	0.322	0.295	0.132
Spring Creek				
Taxa Richness	0.185	0.200	0.315	0.266
MIBI	0.139	0.288	0.198	0.287
%EPT	0.173	0.316	0.411	0.124

APPENDIX A Qualitative Habitat Evaluation Index (QHEI) Data Form

OhioEPA **Qualitative Habitat Evaluation Index and Use Assessment Field Sheet** *QHEI Score:* ☐

Stream & Location: ______ *RM:* _ _ _._ *Date:* _ _/_ _/ 06

______ *Scorers Full Name & Affiliation:* ______

River Code: _ _-_ _ _-_ _ _ *STORET #:* _ _ _ _ _ _ *Lat./ Long.:* (NAD 83 - decimal °) _ _ . _ _ _ _ /8_ . _ _ _ _ *Office verified location* ☐

1] *SUBSTRATE* Check *ONLY* **Two** substrate *TYPE BOXES*; estimate % or note every type present

BEST TYPES	POOL	RIFFLE	OTHER TYPES	POOL	RIFFLE
☐ ☐ BLDR /SLABS [10]	___	___	☐ ☐ HARDPAN [4]	___	___
☐ ☐ BOULDER [9]	___	___	☐ ☐ DETRITUS [3]	___	___
☐ ☐ COBBLE [8]	___	___	☐ ☐ MUCK [2]	___	___
☐ ☐ GRAVEL [7]	___	___	☐ ☐ SILT [2]	___	___
☐ ☐ SAND [6]	___	___	☐ ☐ ARTIFICIAL [0]	___	___
☐ ☐ BEDROCK [5]	___	___	(Score natural substrates; ignore sludge from point-sources)		

NUMBER OF BEST TYPES: ☐ 4 or more [2] ☐ 3 or less [0]

Check ONE (*Or 2 & average*)

ORIGIN: ☐ LIMESTONE [1] ☐ TILLS [1] ☐ WETLANDS [0] ☐ HARDPAN [0] ☐ SANDSTONE [0] ☐ RIP/RAP [0] ☐ LACUSTURINE [0] ☐ SHALE [-1] ☐ COAL FINES [-2]

QUALITY:
SILT: ☐ HEAVY [-2] ☐ MODERATE [-1] ☐ NORMAL [0] ☐ FREE [1]
EMBEDDEDNESS: ☐ EXTENSIVE [-2] ☐ MODERATE [-1] ☐ NORMAL [0] ☐ NONE [1]

Comments

Substrate ☐ *Maximum 20*

2] *INSTREAM COVER* Indicate presence 0 to 3: **0**-Absent; **1**-Very small amounts or if more common of marginal quality; **2**-Moderate amounts, but not of highest quality or in small amounts of highest quality; **3**-Highest quality in moderate or greater amounts (e.g., very large boulders in deep or fast water, large diameter log that is stable, well developed rootwad in deep / fast water, or deep, well-defined, functional pools.

___ UNDERCUT BANKS [1]
___ OVERHANGING VEGETATION [1]
___ SHALLOWS (IN SLOW WATER) [1]
___ ROOTMATS [1]
___ POOLS > 70cm [2]
___ ROOTWADS [1]
___ BOULDERS [1]
___ OXBOWS, BACKWATERS [1]
___ AQUATIC MACROPHYTES [1]
___ LOGS OR WOODY DEBRIS [1]

AMOUNT Check ONE (*Or 2 & average*)
☐ EXTENSIVE >75% [11]
☐ MODERATE 25-75% [7]
☐ SPARSE 5-<25% [3]
☐ NEARLY ABSENT <5% [1]

Comments

Cover ☐ *Maximum 20*

3] *CHANNEL MORPHOLOGY* Check ONE in each category (*Or 2 & average*)

SINUOSITY	DEVELOPMENT	CHANNELIZATION	STABILITY
☐ HIGH [4]	☐ EXCELLENT [7]	☐ NONE [6]	☐ HIGH [3]
☐ MODERATE [3]	☐ GOOD [5]	☐ RECOVERED [4]	☐ MODERATE [2]
☐ LOW [2]	☐ FAIR [3]	☐ RECOVERING [3]	☐ LOW [1]
☐ NONE [1]	☐ POOR [1]	☐ RECENT OR NO RECOVERY [1]	

Comments

Channel ☐ *Maximum 20*

4] *BANK EROSION AND RIPARIAN ZONE* Check **ONE** in each category for *EACH BANK* (*Or 2 per bank & average*)

River right looking downstream

EROSION (L R): ☐ ☐ NONE / LITTLE [3] ☐ ☐ MODERATE [2] ☐ ☐ HEAVY / SEVERE [1]

RIPARIAN WIDTH (L R): ☐ ☐ WIDE > 50m [4] ☐ ☐ MODERATE 10-50m [3] ☐ ☐ NARROW 5-10m [2] ☐ ☐ VERY NARROW < 5m [1] ☐ ☐ NONE [0]

FLOOD PLAIN QUALITY (L R): ☐ ☐ FOREST, SWAMP [3] ☐ ☐ SHRUB OR OLD FIELD [2] ☐ ☐ RESIDENTIAL, PARK, NEW FIELD [1] ☐ ☐ FENCED PASTURE [1] ☐ ☐ OPEN PASTURE, ROWCROP [0] ☐ ☐ CONSERVATION TILLAGE [1] ☐ ☐ URBAN OR INDUSTRIAL [0] ☐ ☐ MINING / CONSTRUCTION [0]

Indicate predominant land use(s) past 100m riparian.

Comments

Riparian ☐ *Maximum 10*

5] *POOL / GLIDE AND RIFFLE / RUN QUALITY*

MAXIMUM DEPTH Check ONE (*ONLY!*)
☐ > 1m [6]
☐ 0.7-<1m [4]
☐ 0.4-<0.7m [2]
☐ 0.2-<0.4m [1]
☐ < 0.2m [0]

CHANNEL WIDTH Check ONE (*Or 2 & average*)
☐ POOL WIDTH > RIFFLE WIDTH [2]
☐ POOL WIDTH = RIFFLE WIDTH [1]
☐ POOL WIDTH > RIFFLE WIDTH [0]

CURRENT VELOCITY Check ALL that apply
☐ TORRENTIAL [-1] ☐ SLOW [1]
☐ VERY FAST [1] ☐ INTERSTITIAL [-1]
☐ FAST [1] ☐ INTERMITTENT [-2]
☐ MODERATE [1] ☐ EDDIES [1]
Indicate for reach - pools and riffles.

Recreation Potential
Primary Contact
Secondary Contact
(circle one and comment on back)

Comments

Pool / Current ☐ *Maximum 12*

Indicate for functional riffles; Best areas must be large enough to support a population of riffle-obligate species: Check ONE (*Or 2 & average*). ☐ **NO RIFFLE [metric=0]**

RIFFLE DEPTH
☐ BEST AREAS > 10cm [2]
☐ BEST AREAS 5-10cm [1]
☐ BEST AREAS < 5cm [metric=0]

RUN DEPTH
☐ MAXIMUM > 50cm [2]
☐ MAXIMUM < 50cm [1]

RIFFLE / RUN SUBSTRATE
☐ STABLE (e.g., Cobble, Boulder) [2]
☐ MOD. STABLE (e.g., Large Gravel) [1]
☐ UNSTABLE (e.g., Fine Gravel, Sand) [0]

RIFFLE / RUN EMBEDDEDNESS
☐ NONE [2]
☐ LOW [1]
☐ MODERATE [0]
☐ EXTENSIVE [-1]

Comments

Riffle / Run ☐ *Maximum 8*

6] *GRADIENT* (ft/mi) ☐ VERY LOW - LOW [2-4] ☐ MODERATE [6-10] ☐ HIGH - VERY HIGH [10-6]

DRAINAGE AREA (mi²)

%POOL: ☐ **%GLIDE:** ☐
%RUN: ☐ **%RIFFLE:** ☐

Gradient ☐ *Maximum 10*

EPA 4520 06/16/06

APPENDIX B Presence/absence list of macroinvertebrate operational taxonomic units. A – 2010 data, B- 2011 data.

Operational Taxon. Units	PC1A	PC1B	PC2A	PC2B	PC3A	PC3B	PC4A	PC4B	PC5A	PC5B	PC6A	PC6B	PC7B
Aeshnidae	1	0	0	0	0	0	0	0	0	0	0	0	0
Amnicola	0	0	0	0	1	0	1	0	0	0	1	0	0
Ancylidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellidae	1	1	0	0	0	0	1	0	0	0	1	0	1
Baetis	1	0	1	0	1	0	1	0	1	1	1	0	1
Caenis	0	0	1	0	0	1	0	1	0	1	1	1	0
Cambaridae	1	1	0	0	0	1	1	0	1	1	0	1	1
Ceratopsyche	0	0	0	0	0	0	1	0	1	0	0	0	0
Cheumatopsyche	1	1	1	1	1	1	1	1	1	1	0	1	1
Chimarra	0	0	0	0	0	0	1	0	1	0	0	0	1
Chironomini	1	1	1	1	1	1	1	1	1	1	1	1	1
Cipangopaludina	0	1	0	0	0	0	0	0	0	0	0	0	0
Coenagrionidae	0	0	1	1	0	1	1	1	1	1	1	1	1
Corbicula	0	0	0	1	0	1	0	0	0	0	0	0	0
Corydalidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Corynoneurini	1	0	0	0	0	0	0	0	0	0	0	0	0
Crangonyx	1	0	0	0	0	0	0	0	0	0	0	0	0
Dubiraphia	0	1	1	1	1	1	1	1	1	1	1	1	1
Ectopria	0	0	0	0	0	1	1	1	1	1	0	0	1
Gyrinus	0	0	0	0	0	0	0	0	0	0	0	0	0
Helicopsyche	0	0	0	0	0	0	1	1	1	0	1	1	0
Hemerodromiinae	0	0	0	0	0	0	0	0	0	0	0	0	0
Heptagenidae	0	1	0	0	0	0	0	0	1	0	1	0	0
Hyalella	1	0	1	0	0	1	1	1	0	0	1	1	1
Hydrobiidae	0	0	0	1	0	1	0	1	0	0	0	1	1
Hydropsyche	1	0	0	0	1	0	1	1	1	1	0	1	1
Hydroptila	1	1	1	1	1	1	0	1	0	0	0	1	0
Lepidoptera	0	0	0	0	0	0	0	0	0	1	0	0	0
Leptoceridae	0	0	0	1	0	1	0	1	0	1	0	1	0
Leptohypidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Libellulidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaeidae	1	0	1	0	1	0	1	0	1	1	1	0	0
Macronychus	0	0	0	0	0	0	1	0	1	1	0	1	1
Oligochaeta	1	1	1	1	1	1	1	1	1	1	1	1	1
Optioservus	0	0	0	0	0	1	1	1	1	1	0	1	1
Orthocladiini	1	1	1	0	1	1	1	1	1	1	1	0	1
Pentaneurini	0	0	1	0	1	0	0	0	0	1	0	0	0
Perlesta	1	1	0	0	0	1	1	1	1	1	0	0	0
Physidae	0	1	0	1	0	1	0	1	0	1	0	1	1
Planorbidae	0	1	0	1	0	1	0	0	0	0	0	0	1
Pleuroceridae	0	0	0	0	0	0	1	1	1	1	1	1	1
Procladiini	1	0	0	0	0	0	0	0	0	0	1	0	0
Simulium	1	1	1	0	1	1	1	1	1	1	0	1	1
Sphaeriidae	1	1	1	1	1	1	1	1	1	1	1	1	1
Stenelmis	0	1	1	1	1	1	1	1	1	1	1	1	1
Stratiomyinae	0	0	0	0	0	0	0	1	0	0	0	0	0
Tanytarsini	0	1	1	1	1	1	1	1	1	1	1	1	1
Tipulini	0	1	1	0	1	1	0	1	1	1	1	1	1
Viviparus	0	0	0	0	0	0	0	0	0	0	0	0	1

APPENDIX B continued

Operational Taxon. Units	SC1A	SC1B	SC2A	SC2B	SC3A	SC3B	SC4A	SC4B	SC5A	SC5B	SC6A	SC6B
Ablabesmyia	0	0	1	0	0	0	1	1	0	0	0	1
Aeshnidae	0	1	0	0	0	0	0	0	0	0	0	0
Asellidae	1	1	1	1	0	1	1	1	1	1	0	1
Baetis	0	0	1	0	0	0	1	0	0	0	0	0
Belostomatidae	1	0	0	0	0	0	0	0	0	0	0	0
Caenis	1	1	0	0	1	1	1	1	0	1	1	1
Cambaridae	0	1	0	0	1	0	0	0	0	1	0	0
Ceratopogonidae	0	0	0	0	0	0	0	0	0	0	0	0
Cheumatopsyche	1	1	1	1	1	1	1	1	1	0	0	1
Chimarra	0	0	1	0	0	0	1	0	0	0	0	0
Chiromomini	1	1	1	1	1	1	1	1	1	1	1	1
Chironomini	1	1	1	1	1	1	1	1	1	1	1	1
Coenagrionidae	1	1	0	1	1	1	1	0	1	0	1	1
Coloptanypodini	0	0	0	0	1	0	0	0	0	0	1	0
Corduliidae	0	0	0	0	0	0	0	0	0	0	1	0
Corynoneurini	1	0	0	1	0	0	1	0	1	0	0	0
Crangonyx	0	0	0	1	0	0	1	0	0	0	0	0
Culicidae	0	0	0	0	0	0	0	0	0	0	0	0
Dubiraphia	1	1	1	1	1	1	1	1	1	1	1	1
Empididae	1	0	0	0	1	0	0	0	0	0	1	0
Ephydridae	1	0	0	0	0	0	0	0	0	0	1	0
Gomphidae	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus	0	0	0	0	0	0	0	0	0	0	0	0
Heptageniidae	0	0	1	0	0	0	1	0	0	0	0	0
Hetaerina	0	0	0	0	0	0	0	0	0	0	1	0
Hexagenia	0	0	0	0	0	0	0	0	0	0	0	0
Hyalella	1	1	0	0	0	1	1	0	0	0	1	1
Hydroptila	0	0	1	1	0	1	0	0	0	0	0	1
Hydroptilidae	0	0	0	0	0	0	0	0	0	0	0	0
Hyrdopsyche	1	0	1	0	1	0	0	0	1	0	0	0
Lepidoptera	0	0	0	0	0	0	0	0	0	0	1	0
Leptoceridae	0	0	1	1	1	1	1	1	1	1	1	1
Lestidae	0	1	0	0	0	0	0	0	0	0	0	0
Libellulidae	0	0	0	0	0	0	0	0	0	0	1	0
Lymnaeidae	1	0	1	1	0	0	0	0	1	1	0	0
Macryonychus	0	0	0	0	0	0	0	0	1	1	1	0
Oligochaeta	1	1	1	1	1	1	1	1	1	1	0	1
Optioservus	0	0	1	1	0	0	1	0	0	0	0	0

APPENDIX B cont.

Pentaneurini	0	0	0	0	0	0	1	0	0	0	1	1
Perlesta	0	0	0	0	0	1	0	1	0	1	0	1
Phryganeidae	0	0	0	0	0	0	0	0	0	0	0	0
Physidae	1	1	1	1	1	1	0	0	1	1	1	1
Planorbidae	1	0	1	1	0	1	0	0	0	0	1	1
Procladiini	0	0	0	0	1	0	1	1	0	1	1	0
Scirtidae	0	0	0	0	0	0	1	0	0	0	0	0
Sialidae	0	0	0	0	0	0	0	0	0	0	0	0
Simulium	0	1	1	1	0	1	1	0	1	1	0	1
Sphaeriidae	1	1	1	1	1	1	1	1	1	1	1	1
Stenelmis	0	0	1	1	1	0	1	1	1	1	0	0
Stratiomyidae	0	0	0	0	0	1	0	0	0	0	0	0
Tabanidae	0	0	0	0	0	0	0	0	1	0	1	0
Tanytarsini	1	1	1	1	1	1	1	1	1	1	1	1
Tipulini	1	1	1	1	0	1	0	1	1	1	0	1