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PILOT PROJECT FOR EVALUATION OF STANDARD BIOTIC METRICS IN URBANIZED WATERSHEDS

submitted by

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EXECUTIVE SUMMARY

It was the objective of this pilot project to compare the performance of biological metrics and indices as predictors of the effects of urbanization on ecosystem integrity. Biological metrics and indices were compiled for three urban/suburban streams. Additionally, a suite of land use metrics for the watersheds were compiled from a GIS database. The intent was to evaluate the covariance among the biological and land use metrics and to assess the most significant variables affecting the sensitivity of the biological metrics to variation in landscape characteristics. We consider this a pilot of a larger project for developing predictive models of the effects of the urban landscape on biological metric scores.

Several measures of biological integrity were examined in this study including: the Index of Biological Integrity (IBI), based on fish; the Invertebrate Community Index (ICI), the Hilsenhof Biotic Index (HBI), and the percent Ephemeroptera, Plecoptera, Trichoptera taxa (%EPT) all based on macro invertebrates; and the Langa-Bertalot Index (LBI), the Trophic Diatom Index (TDI), the Tolerant Species Index (TSI), the Generic Diatom Index (GDI), and the Sensitive species Index (SSI), all based on diatoms.

These metrics were evaluated based on samples from three second to third order urban/suburban streams in greater Cleveland: Doan Brook which drains directly into Lake Erie, and Salt Run and Mill Creek, both of which are tributaries of the Cuyahoga River. Mill Creek and Doan Brook were selected as representing typical urban/suburban impacted streams and for which remediation efforts are being attempted and water quality sampling has been conducted (NEORSD, 1997). Salt Run was selected to reflect a less impacted suburban stream based on water chemistry data, ground surveys, and available knowledge.

Land use metrics against which the biological measures are analyzed in this study include GIS derived measures of perviousness , undeveloped area, area urban soil classification, tree canopy cover, grass cover, residential density, and commercial/industrial density within the watersheds. These metrics were calculated for aggregate watershed regions, defined as the aggregate watershed area above a site, and sub-watershed regions, defined as the area of the watershed that drains into the stream from site to site. Metrics were also calculated for 58 and 116 meter buffer zones on either side of the steam. Other metrics used as covariables in this analysis include: the Qualitative Habitat Evaluation Index (QHEI) which measures stream habitat quality; and measures of water chemistry.

Findings Summary

(1) Six of the metrics which comprise the IBI were unable to discern differences between the sites of this study at a 95% confidence level. This means that in degraded conditions some metrics continue to serve as discriminant indicators, while others, though still providing biological information, do not.

(2) Most of the variance in IBI scores was accounted for by metric 1 (total number of indigenous fish species) and to a lesser extent by metric 12 (numbers of individuals with deformities, eroded fins, lesions, and tumors). This means that in these urban systems the IBI functions mainly as a richness measure weighted by the condition of individual fish.

(3) Four Invertebrate Community Index (ICI) metrics provided all the variance in the ICI scores for the study sites. Two of the metrics, metric 6 (percent Caddisflies) and metric 4 (number of Dipteran taxa), accounted for most of this variance. This means the ICI functions mainly as a measure of Caddisflies and Dipterans in these sites.

(4) The IBI scores appeared to correlate best with soils in the both the aggregate and sub-watersheds along with residential density, undeveloped area, and perviousness. The sub-watershed associations were similar to those of the aggregate watershed. The IBI scores were correlated most strongly with perviousness in the 116 and 58 meter buffer strips. Some individual IBI metrics also appeared to correlate with land use variables examined in this study.

(5) The correlations between the IBI scores and land use variables were similar to the correlations between soluble reactive phosphorous (SRP) and land use variables. However, causal links were not examined in this study, so it was not determined if these correlations are redundant effects of urban impacts or if water chemistry impacts IBI scores in these systems.

(6) No IBI score above 35 (approximately the level considered "good" for this ecoregion) was found in a sub-watershed with less than 90% perviousness in this study. This is consistent with previous studies conducted elsewhere.

(7) The ICI scores did not appear to significantly correlate with land use metrics of the watersheds or buffer zones. However, an examination of some scatter plots suggests that the ICI could separate sites in this study with low urban impact. For example, the plot of ICI scores V.S. pervious area showed no ICI scores less than 28 for areas above 75% perviousness. ICI metrics 1, 2, and 9 did appear to correlate with some variables. ICI metric 9 (tolerant species) correlated with a standardized, summed nutrient score (conductivity + SRP + NO_{2+3} + NH_3) in the streams and metric 5 (percent Mayflies) correlated with SRP in the streams.

(8) The %EPT and HBI scores appeared to correlate best with commercial/industrial land use and also correlated with tree canopy, perviousness, and undeveloped area in the aggregate watersheds. However, the relationships were counterintuitive (e.g. the more commercial\industrial land use, the better the %EPT and HBI scores) and may have been an artifact of a small data set, but may have suggested a geomorphological relationship such that macro invertebrates may have been less resilient in smaller urban streams or in the smaller upstream sections of urban streams (where there was less

commercial\industrial area in these systems).

(9) The LBI and TSI were the only diatom indices that correlated with land use variables. Both were correlated best in the aggregate and sub-watersheds with soils. They also correlated with residence density and the LBI also correlated with undeveloped area in the aggregate and sub-watersheds. Both the LBI and TSI correlated with residence density, undeveloped area, and perviousness in the buffer zones. Neither were correlated with water chemistry parameters in this study.

(10) The TSI correlated with the IBI index scores in these systems. Except for this correlation, the fish index , the macro invertebrate indices, and the diatom indices did not appear to strongly correlate between organismal groups. This means that for the degraded sites, the relative level of stream system health at a site indicated by one organismal group may not serve as a proxy for the relative level of stream system health at that site indicated by another organismal group. For example, Doan Brook site 5 had the best ICI score and the worst LBI score.

(11) The Qualitative Habitat Evaluation Index (QHEI) did not correlate with any biological measures or water chemistry parameters in these samples.

(12) This study could not clearly detect covariables such as higher QHEI scores or greater tree canopy in the buffer zones which might affect the relationships between the land use variables in the watershed and the biological measures.

(13) The method of assessing the status of the landscape by classifying imagery from the Landsat Thematic Mapper and from Digital Aerial Photography generated a flexible and quantitative data set well suited to the heterogeneous nature of these small urban watersheds.

Conclusions and recommendations for further research

(1) The traditional biological metrics are limited in their ability to discern urban impacts in these highly degraded systems. Even biological metrics which seem to be associated with land use impacts in this study do not clearly distinguish between the highly degraded sites. The traditional metrics function well in assessing use attainment, but would be ambiguous measures of effectiveness of incremental remediation efforts.

(2) Water chemistry change is a factor in urban stream degradation that is not currently measured in any way that has meaning for biological integrity of urban streams. The IBI and ICI scores do not clearly reflect water chemistry changes. The best measures of water chemistry impacts in these systems might be the IBI index which correlates with SRP and Cl in these systems and IBI metric 12 (DELT anomalies) which correlates with several water chemistry and land use variables, but the causal nature of these associations is not clear. A diatom index such as the TSI which has been shown to respond to pollution (Kelly and Whitton 1995) should be further tested as a

pollution monitor in these systems.

(3) The associations between the biological metrics and land use examined in this study are intriguing, but correlative. Hence, it is difficult to interpret differences in biological metric scores. Rather, several questions are raised by these correlations, for example: what is the causal relationship between land use and biological measures; why do some biological metrics correlate with land use while others do not; what is the cause of counterintuitive correlations; what is the nature of redundancy in land use correlations with biological metrics; how are the shifting correlations between biological metrics and land use at different watershed levels to be interpreted; why is there a lack of correlation between QHEI scores and biological metrics in these sites; and to what extent are water chemistry associations with biological metrics redundant?

Future study should focus on the causal links between urban impacts and the stream biota. Specific research to be undertaken might include:

(1) Statistical techniques such as path analysis using a more expansive data set which includes at least 100 sites, hydrology data, and data from additional biological organisms to examine causation between variables;

(2) Manipulative studies which might include artificial stream studies;

(3) Process based studies of aquatic life and stream and riparian habitat characteristics;

(4) Paired monitoring protocols involving sites undergoing different remediation efforts or BMPs;

(5) Efforts to identify and define features of urban stream outliers (sites which perform better or worse than expected) in terms of biological integrity;

(6) Comparative studies of stream reaches which involves detailed assessments, temporal changes, and spatial studies such as patch analysis to determine watershed scale impacts;

(7) Assessing remediation efforts by targeting specific biological goals (not necessarily the traditional metrics) appropriate for the site;

(8) Additional measures of urban ecosystem health should be explored with GIS technology, particularly those that could be incorporated into a predictive model for the effects of urban land use on the biota.

Finally, this study was conducted as a pilot project. Though it serves well as an exploratory study with many compelling findings, the results are wholly correlative, and the data set is limited in size and geographic coverage. Thus, causation underlying covariation among variables cannot be assigned with certainty at this point, and the generality of these results is unclear at present. However, the purpose of the pilot project is to indicate where future efforts along these lines may be fruitful. Exploring the relationship between the measures of biological integrity and features of urbanization as done in this study refines our ability to set goals for urban streams and leads to the development of management tools which can assist in the decision process of remediation efforts. A predictive model for urban streams with causal links identified should be the goal. Whatever indices and models are developed, however, will need to incorporate fairly complex, and sometimes counterintuitive, interactions between land use, socio-cultural variables, and stream biological health, if the current study is any indication.

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INTRODUCTION

A number of metrics and indices have been developed and employed to measure the quality and integrity of in-stream biological life (Hilsenhoff 1987, OEPA 1987, Sgro and Johansen 1998). Some of these measures, for example the Index of Biological Integrity (IBI) and the Invertebrate Community Index (ICI), are used to establish and monitor attainment of beneficial use for rivers and streams in Ohio. However, no headwater stream in any older urbanized region in Ohio in the past 18 years has ever exhibited full attainment of warmwater habitat (WWH) use designation where these measures have been applied (Yoder et al. 1998).

A possible explanation for this phenomenon is that the extent of imperviousness due to urbanization in these watersheds prevents the urban streams from achieving full attainment. Several studies have revealed that watershed imperviousness is a key variable affecting aquatic health in urbanized areas (Shueler 1994). Imperviousness in an urban watershed can affect stream habitat, pollution loads, temperature, and biological diversity. The studies have indicated that generally when watersheds reach levels of around 10% imperviousness few streams can support a diverse fish or benthic community.

However, Yoder et al. (1998) suggest that urbanization and imperviousness alone do not automatically disqualify urban streams from meeting use designation based on biological measures. They hypothesize, based on their study of Cuyahoga Basin and Columbus area watersheds, that co-occurring factors at the watershed scale such as the quality of riparian buffers and the mosaic of different types of land use can have a great influence on biological quality. There is a need, then, to refine the our understanding of the relationship between measures of biological quality and features of urbanization at the watershed level.

It was the objective of this pilot project to compare the performance of biological metrics and indices as predictors of the effects of urbanization on ecosystem integrity. Biological metrics and indices were compiled for three urban/suburban streams. Additionally, a suite of land use metrics for the watersheds were compiled from a GIS database. The intent was to evaluate the covariance among the biotic and land use metrics and to assess the most significant variables affecting the sensitivity of the biotic metrics to variation in landscape characteristics. We consider this a pilot of a larger project for developing predictive models of the effects of the urban landscape on biological metric scores.

Site Descriptions

Three second to third order urban/suburban streams in greater Cleveland were the focus of this study: Doan Brook which drains directly into Lake Erie, and Salt Run and Mill Creek, both of which are tributaries of the Cuyahoga River (Figs. 1 and 2). The Mill Creek watershed is primarily residential and industrial. The Doan Brook watershed is primarily residential. The Salt Run watershed is primarily parkland with mixed mesophytic

Figure 1. Map of Salt Run watershed. Sample sites occur where the stream crosses sub-watershed boundaries. Sub-watersheds are indicated by labels. Sub-watersheds S1 through S3 are the sample watersheds for Salt Run. SM is the sub-watershed of the stream below sample site 3.

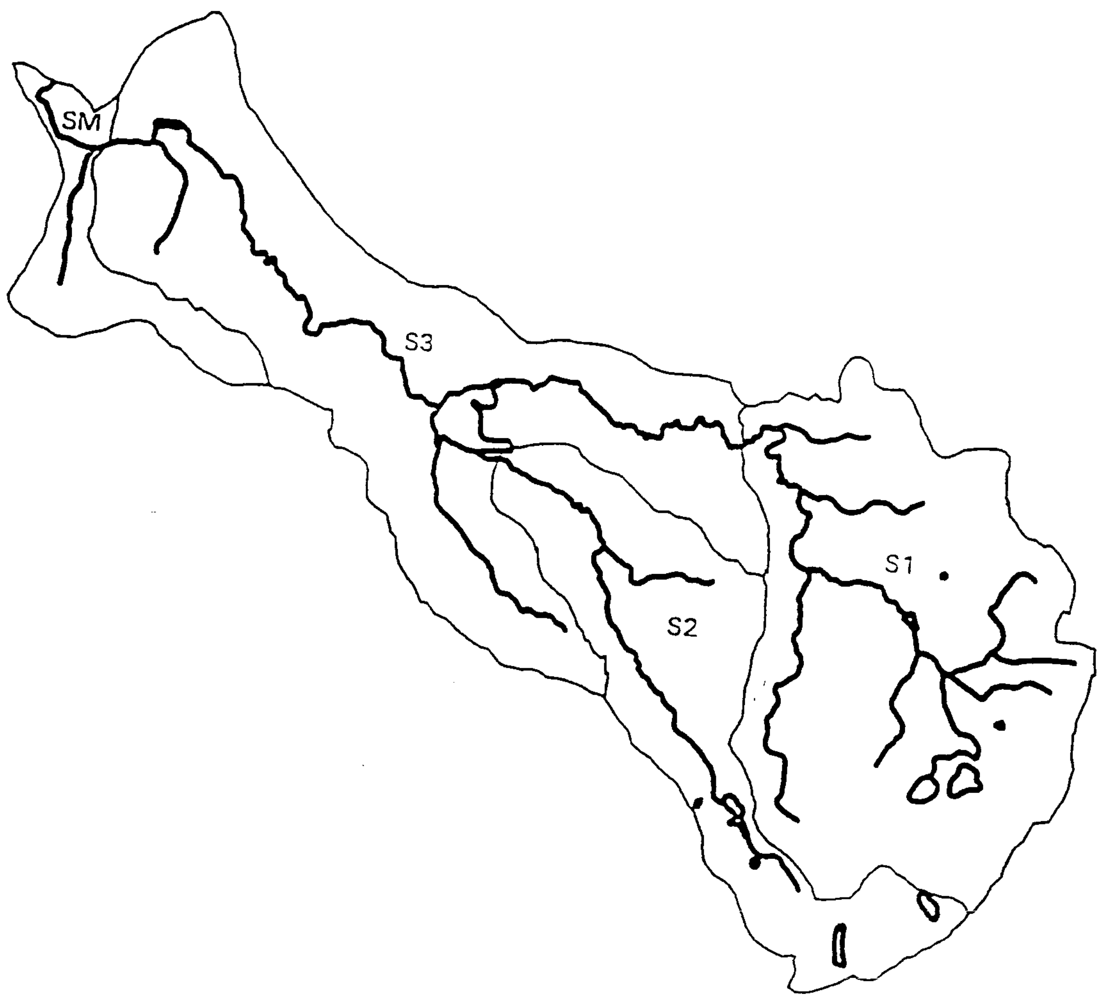
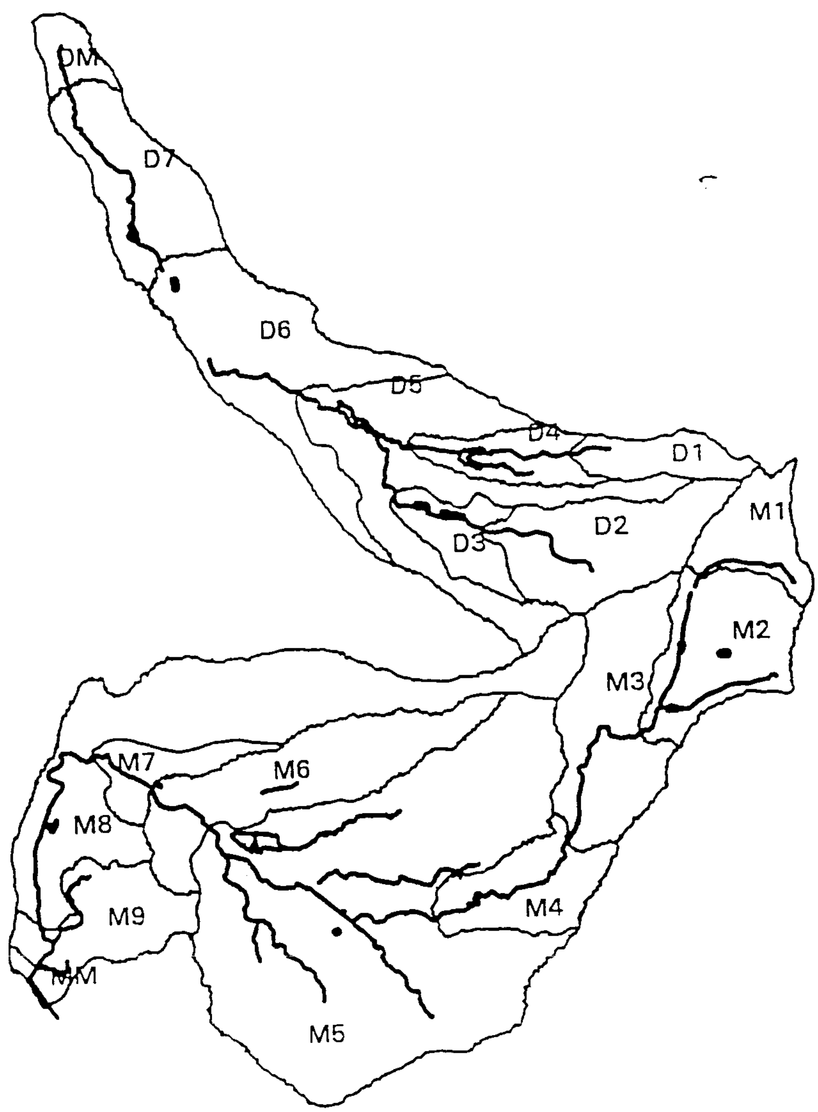


Figure 2. Map of Mill Creek and Doan Brook watersheds. Sample sites occur where the streams cross sub-watershed boundaries. Sub-watersheds are indicated by labels (D1 through D7 = Doan Brook; M1 through M9 = Mill Creek). DM and MM are the sub-watershed of the streams below most downstream sites.



tree cover and residential. Mill Creek and Doan Brook were selected as representing typical urban/suburban impacted streams and for which remediation efforts are being attempted and water quality sampling has been conducted (NEORSD, 1997). Salt Run was selected to reflect a less impacted suburban stream based on water chemistry data, ground surveys, and available knowledge. Upper sites of Salt Run and Doan Brook were located on branches of these streams (see CRWP 1998).

All three are headwater streams and have warm water habitat use designations from the OEPA (Table 1). Mill Creek has never exhibited full attainment of the warm water habitat use designation. Salt Run and Doan Brook have not been sampled by the OEPA.

Methods

Metrics. Biological metrics selected for analysis in this study are currently in use or are being considered for use by water quality managers charged with monitoring the study streams. Biological measures analyzed in this study include: the IBI, based on fish; the ICI, HBI, and percent Ephemeroptera, Plecoptera, Trichoptera taxa (%EPT) all based on macro invertebrates; the Trophic Diatom Index (TDI), the Langa-Bertalot Index (LBI), the Sensitive Species Index (SSI), the Tolerant Species Index (TSI), and the Generic Diatom Index (GDI) all based on diatoms.

The IBI (OEPA 1987) is an index compiled from 12 metrics which measure various attributes of fish in an ecosystem (Table 2). Each IBI metric is scored as 1, 3, or 5 with 5 indicating the best quality an attribute can obtain as measured by a metric at a site based on reference conditions. The highest possible index score for a site is 60 (5 x 12 metrics), but beneficial use attainment in Ohio is based on comparing the study site score with scores of reference sites within an ecoregion. All the sites in this study are located in the Erie Ontario Lake Plains region (Omerick 1987). IBI metric scores for this study were calculated based on the method for headwater or wading streams as used by the OEPA (1987). Scores for Mill Creek and some Doan Brook scores were calculated by NEORSD (1997, 1998) from electroshock samples. Scores for Salt Run and some Doan Brook scores were calculated for this study from seine net samples collected by CRWP (1998) and Curtis (1994).

The ICI (OEPA 1987) is similar to the IBI (OEPA 1987) in that an index score for a site is complied from a set of metric scores which measure attributes of the biota in the stream. The ICI is composed of 10 metrics which measure attributes of the macro invertebrate community (Table 2). The metrics are scored as 0, 2, 4, or 6 and, as with the IBI, the high score indicates conditions of best quality as compared with reference conditions. ICI scores from Salt Run were calculated by Stewart et al. (1998) from quantitative Surber samples except for metric 10, total number of qualitative EPT taxa, which was calculated from qualitative d-net samples. Scores for Mill Creek and Doan

Brook were collected by NEORSD (1997-98) from Hester-Dendy samples.

HBI (Hilsenhoff 1987) scores were calculated by NEORSD (1997, 1998) from semi-quantitative D-frame kicknet samples for Mill Creek and Doan Brook. The HBI is used as an indicator of organic and nutrient pollution in a stream. The HBI score is an index compiled from
tolerance values for arthropods. It is not tied to reference conditions, but, assuming physical habitability of sites to be equal, it provides the basis of a narrative assessment in which the higher scores represent worse conditions (Table 2).

Percent EPT taxa in a sample is also analyzed as a biotic measure of stream quality in this study. Ephemeroptera, Plecoptera, Trichoptera are sensitive organisms which usually are the first to disappear as conditions in a stream deteriorate, thus, a greater % EPT taxa in a sample suggests better quality conditions when compared to a sample with lower % EPT taxa. Percent EPT data used in this study are from NEORSD (1997, 1998) and Stewart et al. (1998).

The Trophic Diatom Index (TDI) (Kelly and Whitton 1994) was designed to serve the monitoring needs of the UK Environment Authority as an indicator of trophic status of streams impacted by wastewater treatment facilities. The index was developed by comparing the relationship between concentrations of filterable reactive phosphorous (FRP) with diatom species frequency at 70 "clean" sites in the UK. The index is calculated by a weighted average equation which accounts for both the indicator value as well as the pollution sensitivity of the index species. The index ranges from 1 (very clean water) to 5 (very polluted water). The TDI was calculated for sites in Mill Creek, Salt Run, and Doan Brook by the CRWP (1998) from samples collected from glass slides.

Lange-Bertalot (1979) examined the relationship between the ecology of globally abundant freshwater diatoms and water quality characteristics in the Rhine-Main river system in Germany from 1973-1977. The relationship between the saprobic conditions (terminology that roughly equates with BOD) in the Rhine-Main system and the relative abundance of the diatoms was used to assign pollution tolerance values of 1 (tolerant of pollution), 2 (less tolerant of pollution), or 3 (pollution sensitive) to diatom species. The Lange-Bertalot Index (LBI) was calculated for Salt Run, Doan Brook, and Mill Creek from glass slide samples by the CRWP (1998). The index was calculated as a simple average following its use by the Montana Department of Health (Bahls 1993) such that a high score indicates clean water.

The Sensitive Species Index (SSI) (Kentucky Department of Environmental Protection 1993, Sgro and Johansen 1998) is calculated as the proportion of sensitive species in total taxa richness in a sample. Weighted average analysis of nutrient data (PO_4, NH_2, NO_3, and NO_2) from Lake Erie estuaries was used to define indicator species for these variables. The SSI was calculated for Salt Run, Doan Brook, and Mill Creek by CRWP (1998) from glass slide samples. A higher percentage of sensitive species indicates cleaner water.

The percent Tolerant Species Index (TSI) (Kelly and Whitton 1995), the percentage of species tolerant to organic pollution in a sample, is used in Europe as a diagnostic feature of the TDI. The metric gives an indication of the amount of eutrophication that is associated with organic pollution. This feature allows the TDI to distinguish between effects of P concentration and the effects of organic pollution (BOD) in an aquatic system. Interpretation of the TSI according to Kelly and Whitton (1995) is given in Table 2.

The Generic Diatom Index (GDI) (Rumeau and Coste 1998) is the principle biological measuring tool for routine aquatic assessments used by the l'Agence de l'Eau Artois-Picardie in France. The GDI allows for water quality assessments with investigations being taken only to genus level. The GDI is calculated the same way as the TDI using a weighted average equation which accounts both for species indicator value as well as sensitivity to pollution. The index ranges from 1 (very polluted water) to 5 (very clean water) which is the opposite of the TDI. The GDI was calculated for Salt Run, Doan Brook, and Mill Creek from glass slide samples by CRWP (1998).

The QHEI (Rankin 1989; Yoder and Rankin 1996) evaluates habitat characteristics of a stream site. Specifically, the QHEI evaluates substrate type, in-stream cover, channel morphology, riparian zone and bank erosion, pool-glide and riffle-run quality, riffle depth and gradient. A stream site is assigned a score for each variable based on a subjective evaluation of the quality of the variable. The variable scores are compiled into an overall QHEI score for the stream site. The scores are positively correlated with environmental quality. The variables can be considered a link between land use in a watershed and biotic integrity of the aquatic system. The QHEI scores are used in this study as covariables with land use factors. QHEI scores used in this study are from NEORSD (1997) and Stewart et al. (1998).

Water chemistry data for this study were obtained from NEORSD (1997) and CRWP (1997-98). Values were averaged in the analysis when there were a series of temporal chemistry samples for a site. NEORSD (1997) samples are dry weather samples and CRWP samples are temporally random samples. Concentrations of Conductivity, NH_3, Cl, NO_2, NO_3, soluble reactive phosphorous (SRP), and suspended solids were considered as covariables in this study (Table 3).

Land use metrics against which the biological measures are analyzed in this study include measures of perviousness , undeveloped area, area urban soil classification, tree canopy cover, grass cover, residential density, and commercial/industrial density within the watersheds. Perviousness refers to the lack of solid surfaces which facilitate water runoff. Canopy indicates tree cover and grass indicates any vegetative cover other than trees. Residential area measures housing density and was distinguished from commercial/industrial areas by building shape and size as recognized on aerial photographs. Undeveloped areas were determined qualitatively and are areas with few or no buildings such as parkland or institutional lawns or golf courses etc. Urban soils data was taken from county soil maps and refers to the area of urban soil classification in the study area.

GIS pixel data was aggregated for the various parts of the watersheds under study. Perviousness, canopy cover, and grass area pixel data were calculated as mean values of these characteristics in the watersheds, subwatersheds, and 58 and 116 meter buffer zones. Commercial/industrial, residential, undeveloped, and urban soil pixel data are scores which can be regarded as a weighted averages of scores for these characteristics weighted by area (see Clapham and Sgro 2000).

The stream was defined for the purpose of calculating land use as the course of any of the three streams included in this study, or their tributaries, as defined on Digital Line graphs of hydrography, derived from US Geological Survey 7.5 minute quadrangles. Aggregate watershed areas are defined as the area in the watershed above each sample site. Sub-watershed areas are defined as the area in the watershed above each sample site to the next upstream site (Figures 1 and 2). The buffer zones are defined as 116 meters (4 pixels) or 58 meters (2 pixels) from the stream, given the nominal resolution of the TM imagery used in this study of 29 meters per pixel (see Clapham and Sgro 2000).

Data analysis. This pilot project is designed to probe relationships between land use variables and various measures of the stream biota. Data are available on fish, macroinvertebrate and diatoms indices in the study streams. However, although the data set is adequate for its purpose, a cautionary note must be given against over interpretation of the analysis. In particular, the data set is limited to just 3 streams and in the case of the diatoms to only 6 data points. The data do not represent a bivariate random sampling model as the data are nested within streams. Many of the relationships between IBI scores and land use variables are non-linear.

An analysis of the metrics which comprise the IBI and ICI was undertaken to evaluate in a statistical sense the information provided by the metrics for the study sites. For these analysis a total of 41 samples for IBI and 14 samples for ICI calculations from Salt Run, Doan Brook, and Mill Creek were used.

A Shannon index (Shannon and Weaver 1949) was used to characterize species diversity. This index attempts to combine species richness and evenness into a single value which characterizes abundance relationships in a community of S species and N individuals.

Spearman correlation coefficients (Hollander and Wolf 1973) were calculated pairwise on both IBI and ICI core metrics to determine interdependence or redundancy. It was arbitrarily decided that a high correlation coefficient (R) of >0.7 or <-0.7 between two metrics in an index indicated a statistical redundancy in contributing information to the structure of the community.

One way analysis of variance (ANOVA) along with Duncan multiple range tests (Box et al. 1987, Neter and Wasserman 1974) was performed with IBI metric scores as variables and sites as factor levels to determine if the metrics could distinguish between sites over time.

The relative contribution of the IBI and ICI metrics to the total variance between site scores and the indices was examined using stepwise regression (Belsly et al. 1980, Draper and Smith 1981, Durbin and Watson 1951). Stepwise regression uses forward selection to add variables to the model one at a time in order of greatest significance to the model. The best one-metric to four-metric models were determined in this way.

Scatter plots along with correlation analysis (Johnson and Wichern 1982) which generates correlation coefficients for sets of observed variables were used to discern correlations and relationships between land use and IBI (41 samples) and ICI (14 samples) metric scores, as well as environmental variables including water chemistry (27 samples for IBI and 11 samples for ICI calculations) and QHEI scores (37 samples for IBI and 11 samples for ICI calculations) from Salt Run, Doan Brook, and Mill Creek. SRP was LOG_e transformed for the correlations. Scatter plots and correlation analysis were also used to determine statistical correlations and relationships among the other indices in the study (HBI, %EPT, TDI, LBI, SSI, TSI, and GDI) and between these indices and environmental variables. Twenty-three samples in all from Doan Brook , and Mill Creek were used for the HBI and 11 samples for %EPT analysis and 6 samples in all from Salt Run, Doan Brook, and Mill Creek were used for the TDI, LBI, SSI, TSI, and GDI calculations (Appendix A).

Simple linear regressions (Draper and Smith 1981) of IBI scores against aggregate and sub-watershed perviousness and soils variables were performed. Residuals from these analyses were regressed against tree canopy and grass in the 58 meter buffers as well as QHEI scores in the streams. This residuals analysis might indicate if trees or habitat in the buffers could affect the IBI response to perviousness or soils. Simple linear regression was also used to explore relationships between land use variables.

A principal components analysis (Pielou 1984) was used to characterize the sites with respect to the land use variables. The non-transformed data were standardized and centered (Pielou 1984) to avoid polar axes. Eigenvalues computed from the species resemblance matrix were rank ordered such that the first few PCA axes or components upon which the samples are positioned represent the largest percentage of the variation that can be explained. All statistical calculations were performed with StatgraphicsPlus (Manugistics Inc. 1992).

RESULTS AND INTERPRETATION

Evaluation of metrics and indices. IBI indices calculated for Salt Run in 1997 and 1998, Doan Brook in 1997 and 1998, and Mill Creek in 1995 and 1998 were highest for Salt Run site 3 of all sites in the study (Appendix A). The Salt Run site 3 1998 sample with a score of 44 was the only sample considered "good" in the narrative rating (OEPA 1987). Neither of the 2 upstream sites on Salt Run scored above 20 in the 2 sample years. None of the Doan Brook sites attained an IBI index score above 22 ("poor"). The middle Doan Brook sites (sites 2, 3, 4, and 5) had higher scores than either the upstream or downstream sites. The highest scores for Mill Creek, like Salt Run, were from downstream sites. Sites 7, 8, and 9 each had scores of 26 in one sample year, however, these sites showed high variability with each of them also achieving scores in the "very poor" range during these sample years. Mill Creek site 1 never scored above a minimum score of 12.

IBI metrics 5, 6, 7, 8, 9, and 11 were unable to distinguish differences in the sites at a 95% confidence level with an ANOVA test. Metrics 2, 3, 4, and 10 were able to distinguish Salt Run site 3 (the best IBI index score) from all the rest. Metric 1 was able to distinguish Salt Run site 3 from all but Mill Creek site 9. Metric 12 distinguished Salt Run site 3 from all but Doan Brook sites 2, 3, 4, and 5 and Mill Creek sites 2, 4, 5, 7, 8, and 9.

Spearman rank correlations indicated that in a statistical sense redundant information was being contributed by the following metrics: metric 1 and metric 3 ($R=.85$, $p<.0001$); metric 1 and metric 11 ($R=.70$, $p<.0001$); and metric 3 and metric 10 ($R=.80$, $p<.0001$). Metric 5 contributed no information to distinguish one sample from another in this study. Metric 5 had a score of 1 for all samples.

The best single metric model, that is the metric that accounted for most variability in the IBI index score using stepwise regression, was metric 1 ($R^2=.66$). The best two metric model was metric 1 + metric 12 ($R^2 = .83$). The best 3 metric model was metric 1 + metric 12 + metric 10 ($R^2 = .89$) and the best 4 metric model was metric 1 + metric 12 + metric 10 + metric 9 ($R^2 = .93$). These 4 metrics were contributing in a statistical sense 93% of the information provided by the IBI in these streams. So, among the IBI metrics a simple richness measure was best able to distinguish the difference between these sites. Furthermore, richness was probably related to stable habitat at the sites.

An examination of the IBI scores for the sites (Appendix A) suggested, however, that the IBI was able to separate more impacted sites from the less impacted Salt Run 3 site. Additionally, the IBI scores for these sites were significantly correlated with Shannon Diversity ($R^2 = 62.5$, $p<.0001$). The decline in fish species diversity and sensitive fish species has been linked to imperviousness in previous studies (Klien 1979, Schueler and Galli 1992).

ICI indices calculated for Salt Run in 1994 and 1995, Doan Brook in 1998, and

Mill Creek in 1995 were highest for Doan Brook site 5 and Mill Creek site 7 (ICI = 38 for both) of all sites in the study (Appendix A). The narrative rating for Salt Run site 3 in 1994 was "fair" (ICI=28) and this improved to "good" (ICI=36) in 1995. ICI metric 8 (% other Dipteran and non-insect composition) was 0 for both years at this site. Doan Brook upstream and downstream sites (sites 1, 2, and 7) were considered "poor", while the middle sites (sites 3, 4, and 6) were considered "fair" with site 5 considered "good". Metric 2 (number of Mayfly taxa) was 0 for each of these sites. Mill Creek sites 2, 8, and 9 were ranked "fair" and site 7 was ranked "good". Metric 2 (number of Mayfly taxa) was 0 for each of these sites.

Spearman rank correlations among ICI metric scores for 14 samples total from Salt Run, Doan Brook, and Mill Creek indicated a statistical redundancy in contributing information to the structure of the macroinvertebrate community among the following metrics: metrics 1 and 4 (R=.75, p=.0065); metrics 5 and 3 (R=.82, p=.0031); metrics 3 and 6 (R=.95, p=.0006) and metrics 5 and 6 (R=.72, p=.009).

Metric 6 (% Caddisfly composition) accounted for the most variability in the ICI scores based on stepwise regression (R^2=.73) for 14 samples total from Salt Run, Doan Brook, and Mill Creek, thus, it was the best single metric for distinguishing differences in these sites. The best 2 metric model was composed of metrics 6 + 4 (R^2=.92). The best 3 metric model was composed of metrics 6+ 4+9 (R^2=.97). The best 4 metric model was composed of metrics 6+4+9+1 (R^2=.99). These four metrics were essentially the only ICI metrics providing information to distinguish these urban systems apart. Caddisfly taxa were habitat sensitive. This implies that the ICI is distinguishing these sites primarily on the quality of the habitat. Better sites were those with less impacted habitat. However, this metric could not inform as to whether it is benthic or terrestrial habitat or both that was degraded.

The impacts of imperviousness and urbanization on macroinvertebrate populations, as with fish populations, have been linked to a decline in sensitive species and diversity in several studies (Schueler 1994). The ICI scores in these systems, however did not correlate with insect Shannon Diversity at a 95% confidence level (R = 0.18, p =.5438). The metric measuring total number of taxa, metric 1, was variable in these samples ranging from 0 to 6. On the other hand, the pollution sensitive Mayfly taxa, as measured by metric 2, were greatly reduced. This metric was scored 0 for all sites in these streams. The sensitive species as measured by metric 5, % Mayfly composition and metric 3, number of Caddisfly taxa, were positively correlated suggesting a similar fate for these taxa in these urban systems.

Percent EPT taxa scores were compiled for 8 sites on Mill Creek in 1995. Mill Creek site 3 had the lowest %EPT score (7%). Site 6 had the highest (82.7%). Other sites ranged from 21.4% to 61.7%. The Mill Creek scores were higher than those for Doan Brook. Only 3 Doan Brook scores were available from 1994 and 1998. These scores ranged from 0 (site 6) to 14.29 (site 4) (Appendix A).

HBI scores were compiled for 8 sites on Mill Creek in 1995. Site 6 had the lowest HBI score (4.56, "good") while site 3 had the highest (6.75, "fairly poor") (Appendix A). HBI scores for 7 sites were compiled for Doan Brook from 1994 - 98. These scores ranged from 5.18, "good", (site 2) to 7.16, "fairly poor", (site 6) (Appendix A).

The negative correlation apparent in the %EPT scores and the HBI scores was examined with linear regression. HBI regressed against %EPT scores for 10 samples taken in 1994-95 gave a significant negative correlation (R^2=89.3, $p<.0001$).

The % EPT and Hilsenhoff are *biotic indices* based on macroinvertebrate indicator species sensitive to organic pollution. They are not tied to regional reference conditions. The narrative ranking of sites in Mill Creek and Doan Brook were generally better than the narrative rankings given these sites by the IBI and ICI.

The algal indices, like the Hilsenhoff and % EPT indices, are not tied to regional reference conditions. They are based on indicator species sensitive to eutrophication and organic pollution. The algal communities have a faster turnover rate than the macroinvertebrate communities so should represent more recent conditions than might be reflected in the macroinvertebrate community. The 5 algal indices examined in this study (TDI, LBI, SSI, GDI, and TSI) were nearly unanimous in ranking the two Salt Run sites (for which there are algal metric scores) as the best quality sites. Doan Brook site 5 was given the lowest ranking by all algal indices, except the SSI, of the measured sites; this site was also given a relatively low ranking by the Hilsenhoff index. (Table 4). The algal indices, based on a linear regression of the 6 1998 samples, correlated with each other except for the GDI (Table 5).

All of the biological measures examined in this study gave relatively high scores to the Salt Run 3 site (Appendix A). However, beyond discerning this better site, the associations between the measures were less clear. The ICI scores were not significantly correlated with IBI scores based on a regression of 10 samples from approximately the same dates and sites from Doan Brook and Mill Creek at a 95% confidence level (R= 0.47, p =.1655). The TSI was the only diatom index to correlate with the IBI. The TSI correlated with averaged IBI scores for the 6 sites (R = -0.87, p = .0247). The diatom indices did not correlate with ICI score based on data from 5 sites. The algal indices gave the lowest ranking to Doan Brook site 5, while the ICI ranked this site highest (Appendix A).

Perviousness is often used as a proxy for the effects of urbanization. Perviousness in this study was significantly (95% confidence level) correlated with all the other variables except grass cover in the aggregate watershed using linear regression analysis. The R^2 values ranged from 44.1% (p = 0.0019) for perviousness regressed against residence to 90.0% (p < 0.0001) for perviousness regressed against undeveloped area. This suggested that, except for grass cover, the statistical information these variables are providing is somewhat redundant, though only the perviousness against undeveloped area regression has an R^2 value greater than 85%. Grass was difficult to quantify from GIS information because it was often obscured by tree canopy.

A general characterization, based on PCA analysis of the aggregate watersheds is one of Mill Creek being relatively commercial, Doan Brook being relatively residential, and Salt Run being relatively undeveloped. The first PCA axis in the aggregate watersheds analysis (Fig. 3) accounted for 65.78% of the variance in the data set. Along this axis the less urban Salt Run sites were separated from the more urban Doan Brook and Mill creek sites by having more tree canopy, undeveloped area and perviousness. The more urban sites were separated from the Salt Run sites by having more urban soils, as well as more residence and commercial area. Doan Brook and Mill Creek sites were separated along the vertical axis which accounts for 29.26% of the variance by grass cover and commercial area, both of which were more plentiful in the Mill creek watershed.

The first PCA axis in the 58 meter buffer strips analysis (Fig. 4) accounted for 52.99% of the variance in the data set. Along the first axis the Salt Run sites were separated from the Mill Creek and Doan Brook sites by having more undeveloped and more pervious buffer strips. Generally, there was more commercial area and more urban soils in the Mill Creek buffer zones than in the Salt Run buffers. Doan Brook was separated from Mill Creek and Salt Run generally along the second axis which accounted for 19.31% of the variance and represented more residential structures in these areas.

Associations between metrics and indices and water chemistry and habitat parameters. A standardized summed nutrient parameter (Conductivity, NH_3, No_{2+3}, and SRP), LOG_e SRP, and Cl were correlate (95% confidence level) with various land use metrics in the aggregate and sub-watersheds and the 116 and 58 meter buffers, though SS was not (see Figs. 5 and 6 for representative plots). The SS values in this study were influenced by the SS concentrations of the Salt Run sites, particularly site 3. Salt Run site 3 had the highest averaged SS values of the samples despite the relatively good quality of this site in other respects (Appendix A).

The best correlation at the aggregate watershed level was between Cl and perviousness ($R = -0.93$, $p < 0.0001$). All land use variables except grass correlated with Cl at the 95% probability level. Additionally, in the aggregate watershed SRP correlated best with soils ($R = -0.83$, $p = 0.0005$) and also correlated with undeveloped, residential, and perviousness area in that order at the 95% probability level. The summed nutrient parameter correlated best with commercil area ($R = 0.79$, $p = 0.0012$). Perviousness, tree canopy, and undeveloped area also correlated in that order with the summed nutrient parameter at the 95% probability level.

Similarly, the best correlation at the sub-watershed level was between Cl and perviousness (Fig. 6, $R = -0.95$, $p < 0.0001$) and again all variables except grass correlated

Figure 3. Principal components plot of sites ordinated by land use variables of the aggregate watersheds (S = Salt Run, M = Mill Creek, D = Doan Brook; numbers indicate site).

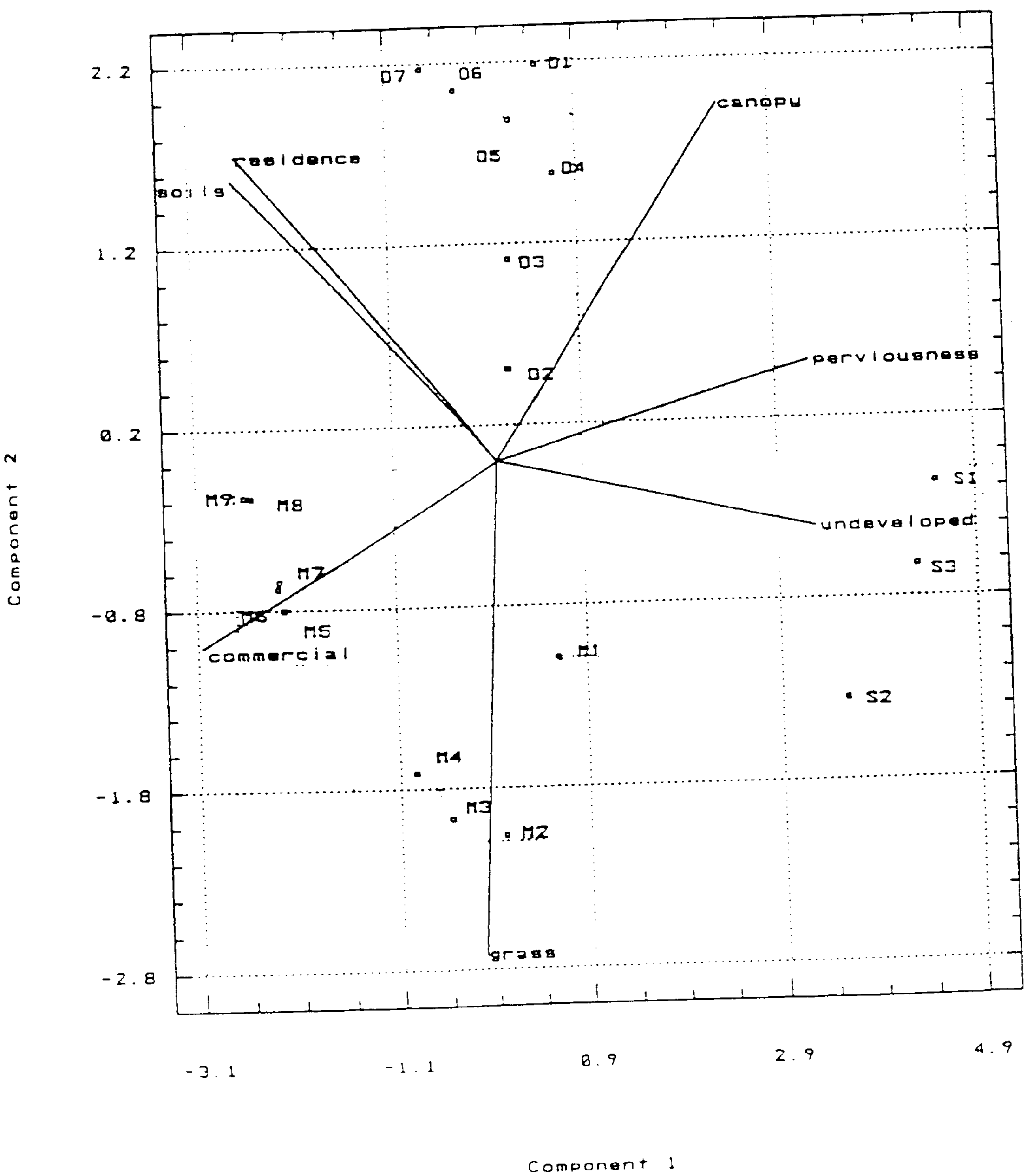


Figure 4. Principal components plot of sites ordinated by land use variables of the 58 meter buffer zones (S = Salt Run, M = Mill Creek, D = Doan Brook; numbers indicate site).

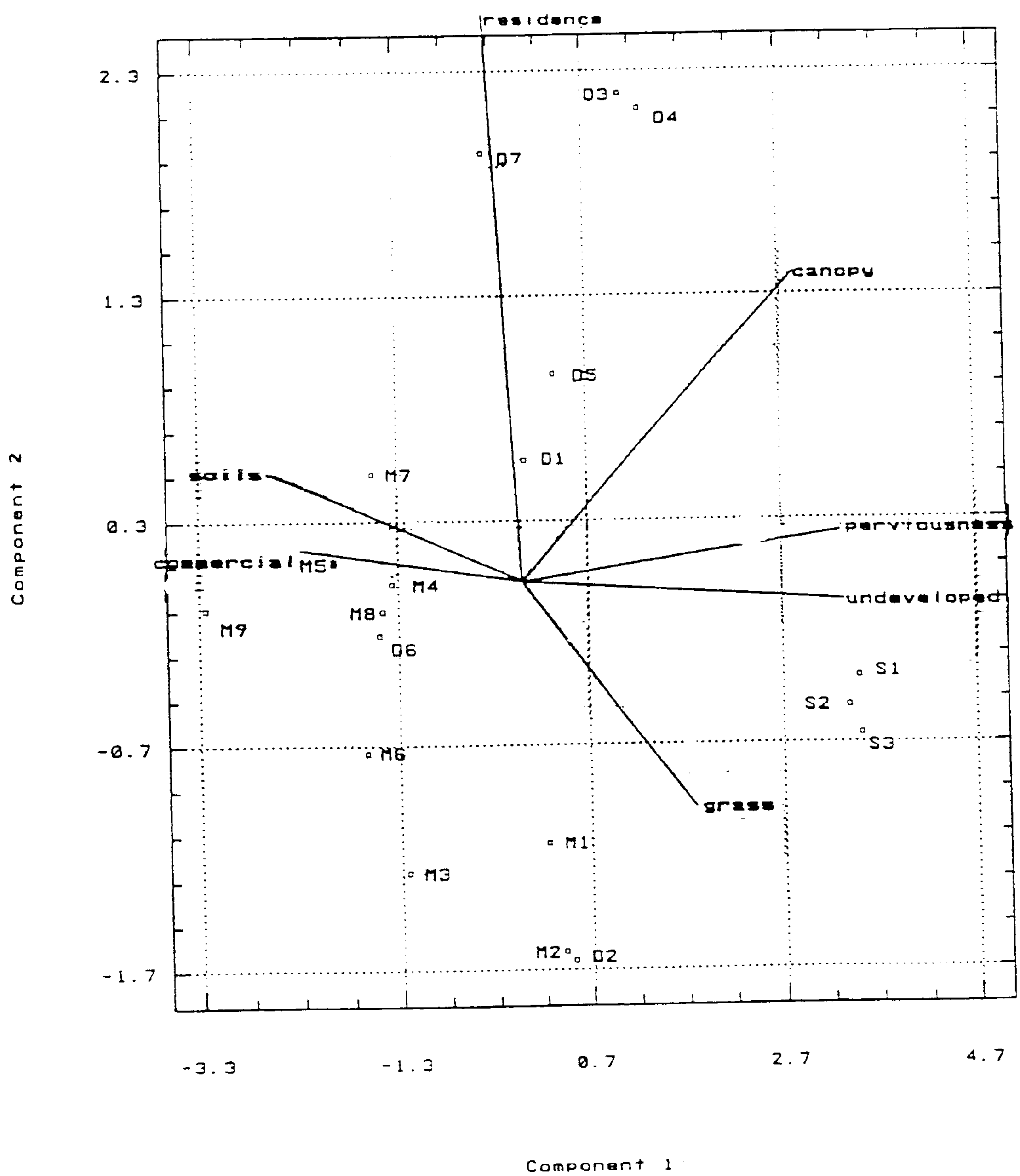


Figure 5   Plot of % commercial area vs nutrients (see methods) for sites in the aggregate watersheds.

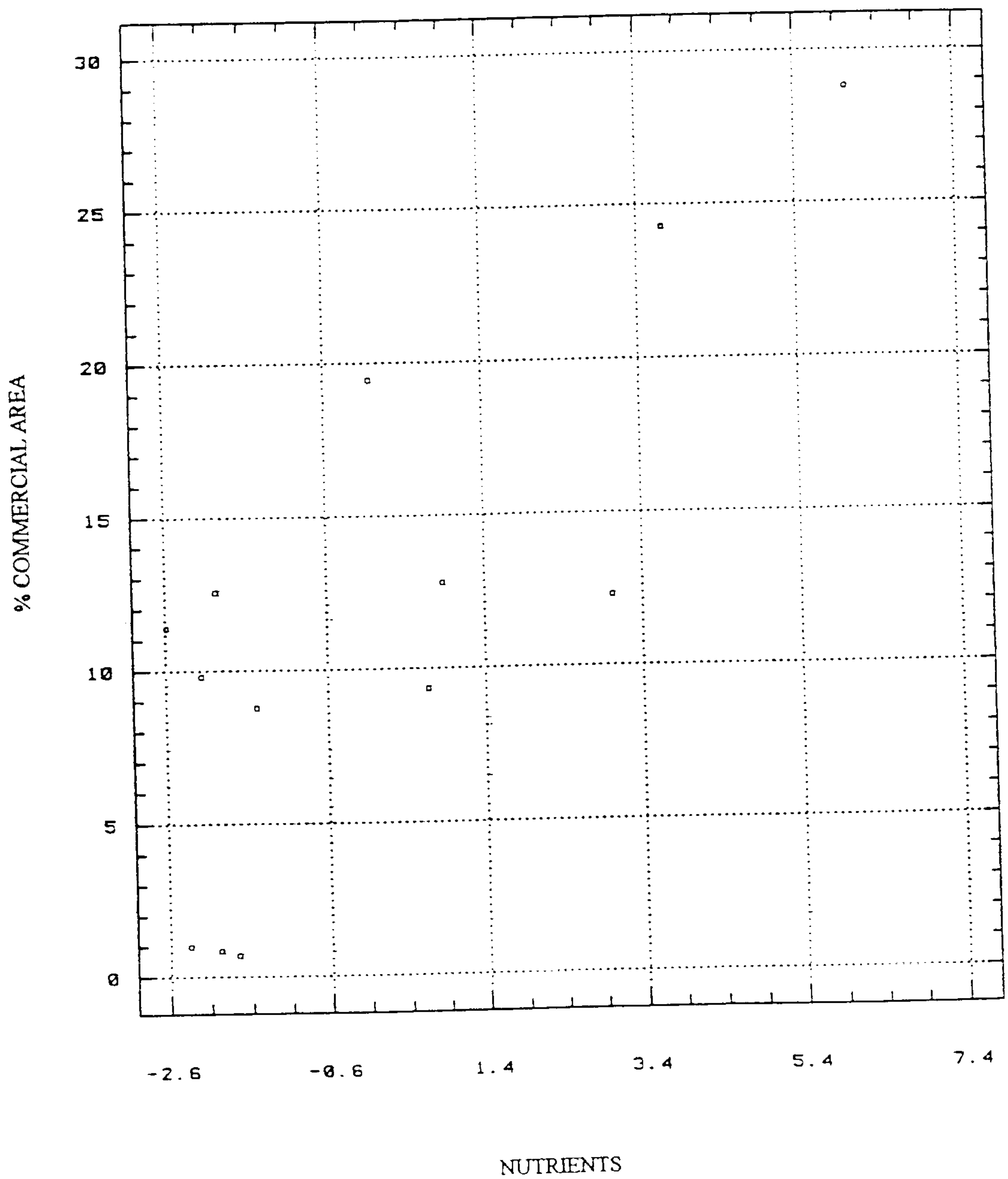
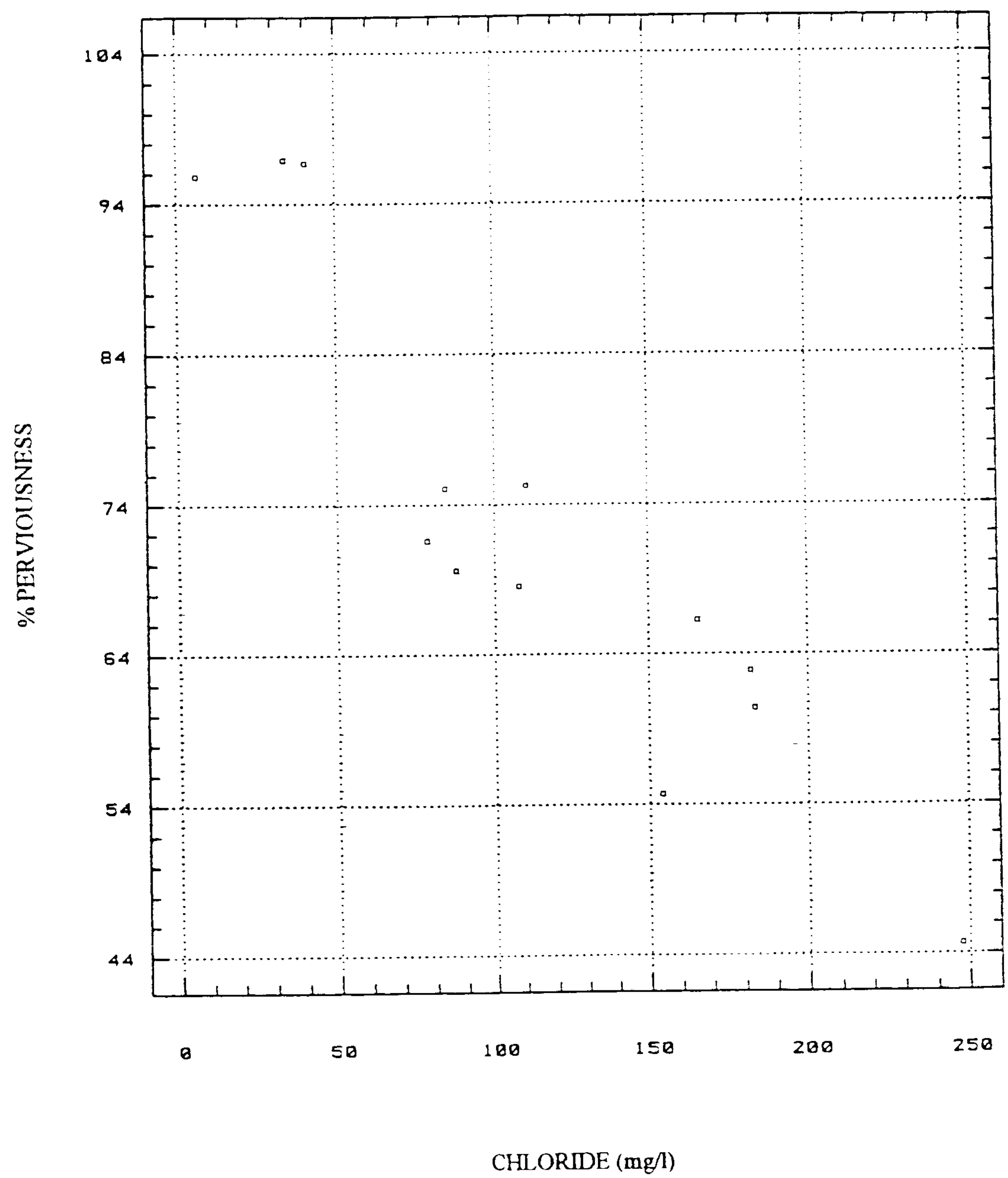


Figure 6 Plot of % pervious area vs chloride concentrations for sites in the aggregate watersheds.



with Cl at the 95% probability level. SRP correlated best ($R = 0.83$, $p = 0.0004$) with soils in the sub-watersheds. Undeveloped, residence, and perviousness variables in that order also correlated with SRP at a 95% probability level. Additionally, the summed nutrient parameter correlated best with commercial area and, as in the aggregate watersheds, also correlated with perviousness, tree canopy and undeveloped area.

The best correlation in the 116 meter buffers was again perviousness and Cl ($R = -0.83$, $p = 0.0004$) and all variables but grass and residence correlated with Cl in these buffers at a 95% probability level. SRP correlated best ($R = -0.78$, $p = 0.0017$) with undeveloped area in these buffer areas. Soils and perviousness also correlated with SRP at a 95% probability level. The summed nutrient parameter correlated with all variables except grass and residence in these buffers having the best correlation with commercial area ($R = 0.80$, $p = 0.0009$).

Undeveloped area and Cl correlated best ($R = -0.82$, $p = 0.0006$) in the 58 meter buffer areas and, as in the 116 meter buffers, all variables but residence and grass correlated with Cl. SRP correlated best ($R = -0.81$, $p = 0.0009$) with perviousness in these buffer areas. Undeveloped area also correlated at a 95% probability level with SRP. The standardized nutrient parameter correlated at the 95% probability level with all land use variables but grass and residence and, again like the 116 meter buffers and the sub-watersheds, the best correlation was with commercial area ($R = 0.79$, $p = 0.0014$).

The IBI index scores ($R = 0.71$, $p < 0.0001$), as well as individual IBI metrics 1, 2, 3, 4, 7, 8, 11, and 12, were correlated with SS at a 95% confidence level. However, this is a counter-intuitive correlation (as SS increases, so do IBI scores) and, as discussed above, can be interpreted as an artifact of a small data set.

The IBI index scores correlated with SRP ($R = -0.52$, $p = 0.0039$) and Cl ($R = -0.38$, $p = 0.0449$) in the streams. The standardized nutrient parameter correlated with individual IBI metric 12; Cl with metrics 10 and 12; and SRP with metrics 1, 9, 11, and 12.

The ICI index scores were not correlated with water chemistry parameters in this study. However, ICI metric 5 correlated with SRP and metric 9 correlated with the standardized nutrient parameter.

The relationship between QHEI scores and the land use variables in the aggregate watersheds and the 58 meter buffer strips were examined with correlation analysis. Commercial area and tree canopy correlated with QHEI scores in the aggregate watersheds ($R = 0.60$, $p = 0.0113$) and 58 meter buffer zones ($R = -0.63$, $p = 0.0070$) respectively. However, the signs of the R values indicated counter-intuitive correlations. That is, it would be expected that a higher QHEI score would have correlated with decreasing, rather than increasing, commercial area in the watersheds and increasing, rather than decreasing, tree canopy in the buffers. This phenomenon may be an artifact of a small data set. However, the lower Mill Creek sites had higher QHEI scores than the smaller streams or upstream portions of streams (Appendix A) and may be more resilient to urban impacts.

IBI index and metric scores for these sites were not significantly correlated at a 95% confidence level with QHEI scores. Despite the fact that both the IBI and ICI respond to habitat disturbances (OEPA 1987), neither of these indices were significantly correlated with the QHEI which measures habitat quality at the site. However, ICI metric 1 (number of taxa) was correlated with the QHEI ($R = 0.70$, $p = .0058$).

The HBI scores correlated with SRP and the EPT, the other macroinvertebrate metric, correlated with SS concentrations in the streams. Again, as discussed above, this association with SS concentrations was probably an artifact of a small data set which is skewed by the relatively high concentrations of SS concentrations in the Salt Run sites.

The algal indices, LBI, TDI, TSI, GDI, and SSI, did not correlate with water chemistry parameters in this study. There is much written in the literature substantiating the sensitivity of algae, particularly diatoms, to water chemistry (see Sgro and Johansen 1995, Weitzel 1979 for reviews). However, the water chemistry data set in this study was temporally averaged for each site and did not represent a synoptic sample with the diatoms. Water chemistry is often highly variable and the diatoms respond quickly to changes in the chemistry. In Doan Brook changes in the diatom community were detectable within 4 days following a storm event (Johansen 1999 unpublished). The macroinvertebrates and fish better integrate water chemistry conditions over time and thus are more likely to correlate with averaged water chemistry values.

Associations between biological and land use metrics. When IBI scores for sites examined in this study were plotted against perviousness they revealed a pattern which is consistent with previous studies of the effects of urbanization on fish populations (Klein 1979, Schueler and Gali 1992). There was a precipitous drop-off of IBI scores for sites with less than approximately 90% perviousness in the aggregate or sub-watershed such that below this level there was no score above 35, approximately the level considered "good" for this ecoregion (Fig. 7, sub-watershed plot not shown). The 4 squares in the lower right of Figure 7 represent Salt Run sites 1 and 2. No reason is apparent from this study why these sites with such a high level of perviousness had relatively low IBI scores. There is a gap in the Figure 6 plot due to a lack of sites between 75% and 90% perviousness. An examination of the plot would lead to a speculation that such watersheds in this range would have IBI scores in the mid to upper 20's.

The aggregate, sub-watershed, 116, and 58 meter buffer scatter plots of IBI scores against residence, tree canopy, soils, undeveloped, and commercial variables examined in this study were all similar to the IBI scores plotted against perviousness as shown in Figure 7 (see Figs. 8 and 9 for representative plots). The plot of IBI scores and grass cover did not resemble the other plots and there was no discernable pattern in this plot.

Figure 7 Plot of IBI scores for sites vs % perviousness in the aggregate watersheds.

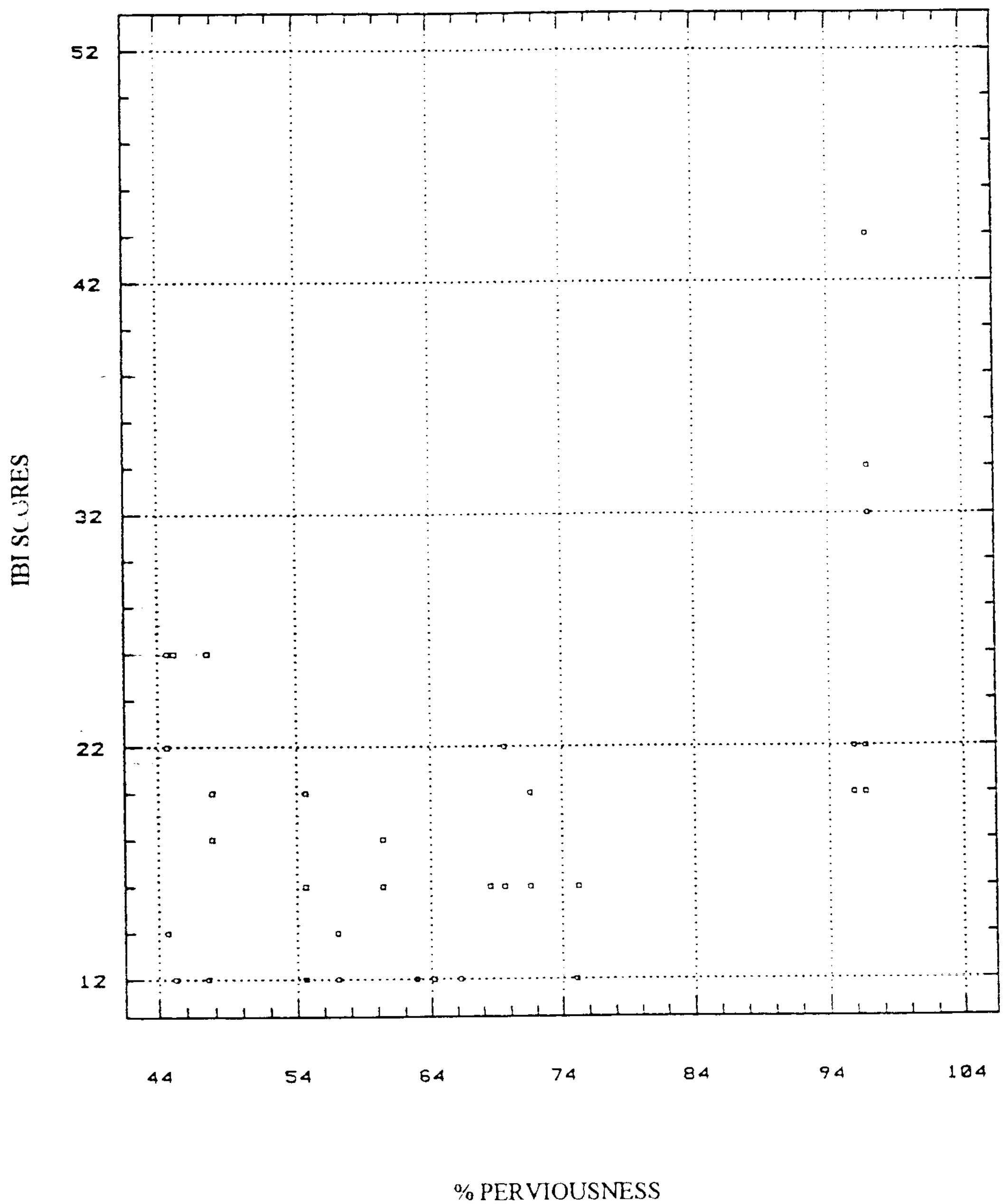


Figure 8. Plot of IBI scores for sites vs soils (see methods) in the sub-watersheds.

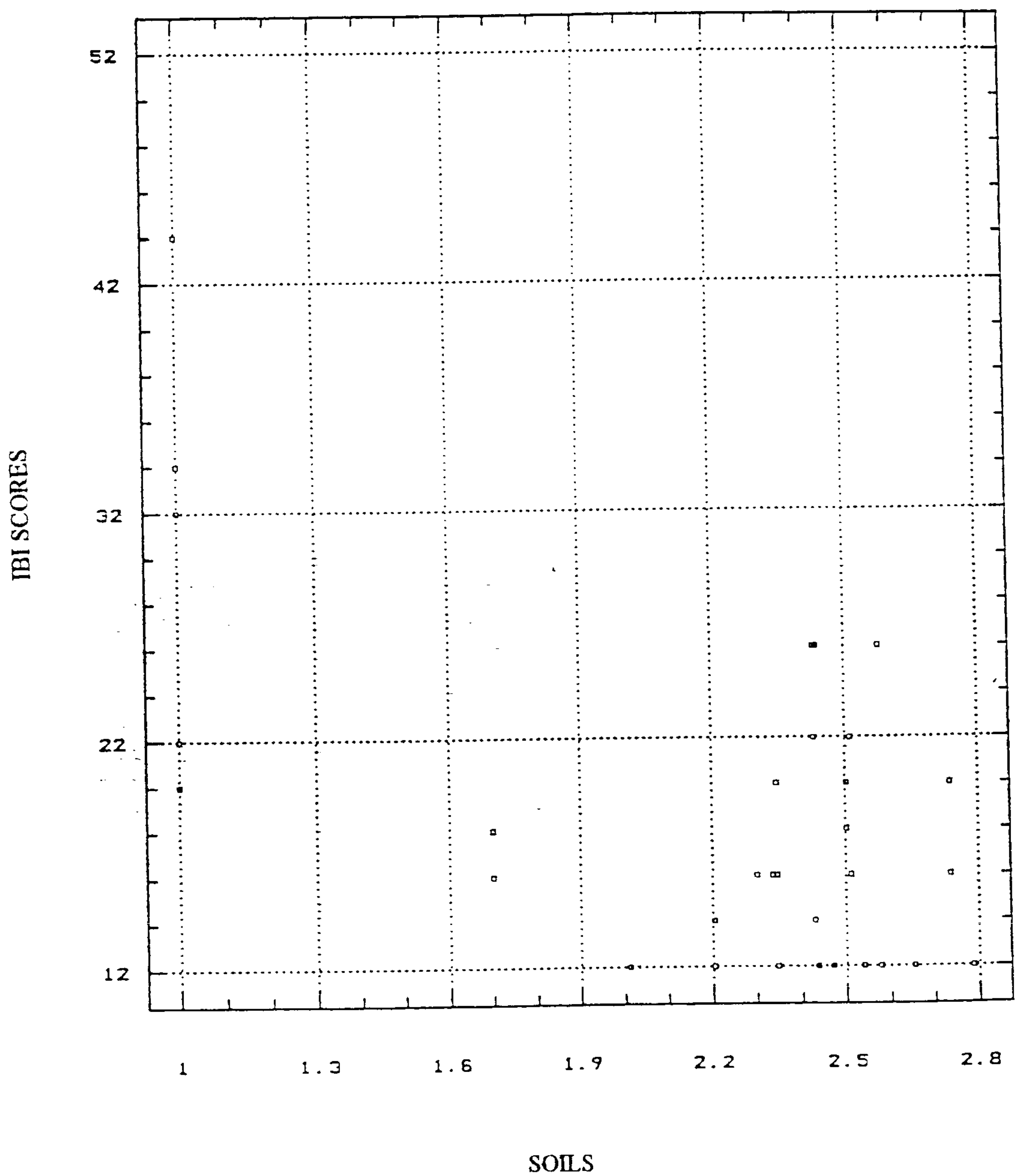
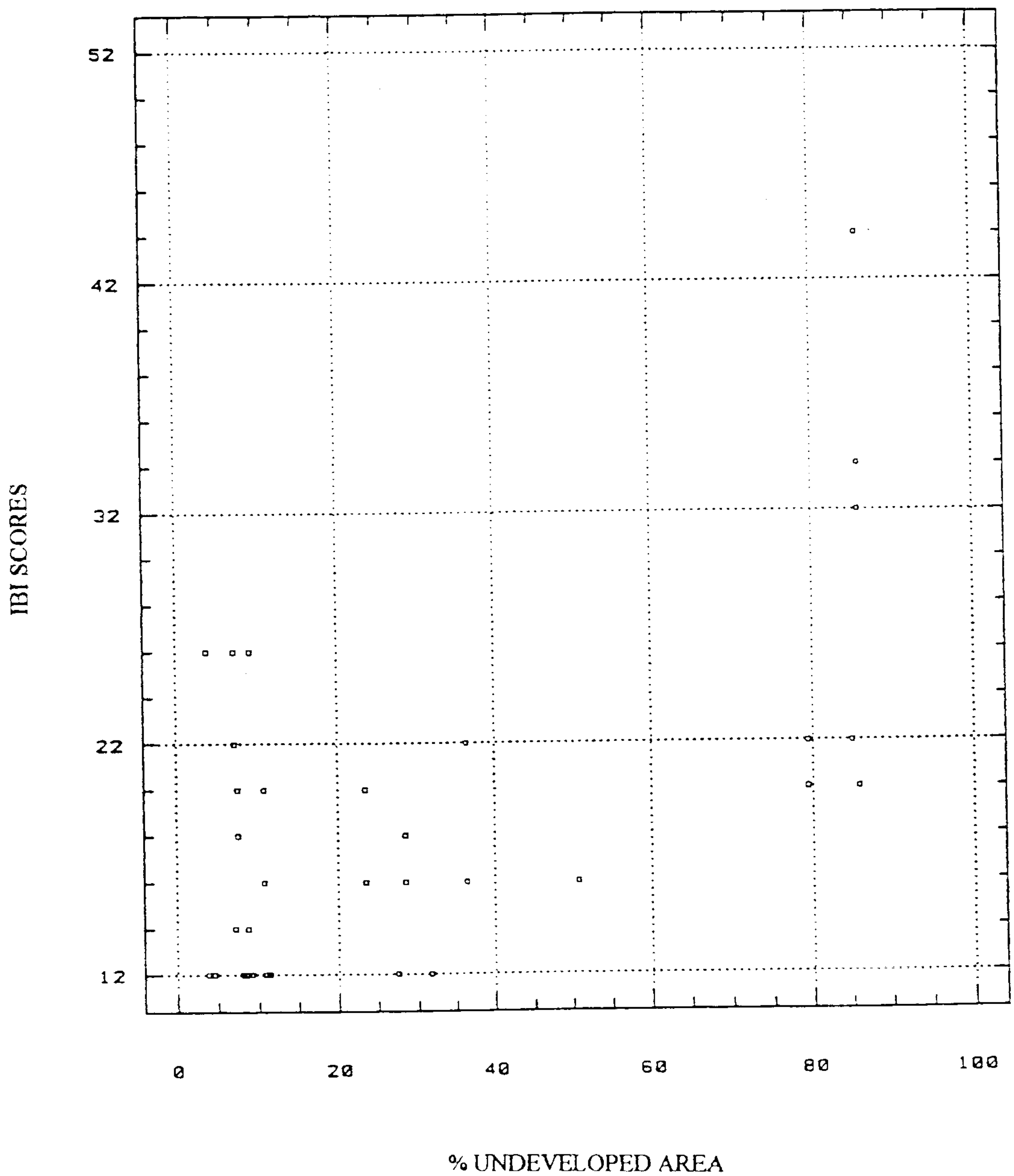


Figure 9 Plot of IBI scores for sites vs % undeveloped area in the 58 meter buffer zones



Correlation analysis associated the IBI scores in the aggregate watersheds most closely with soils ($R = -0.57$, $p < 0.0035$). The IBI scores also correlated with soils, residential area, undeveloped area, and perviousness in that order at a 95% confidence level in the aggregate watersheds. The associations were similar at the sub-watershed level with soils ($R = -0.56$, $p = 0.0001$), residential area, undeveloped area, and perviousness again correlated with IBI scores in that order. The correlations between IBI scores and land use variables closely reflected the correlations between SRP and land use variables.

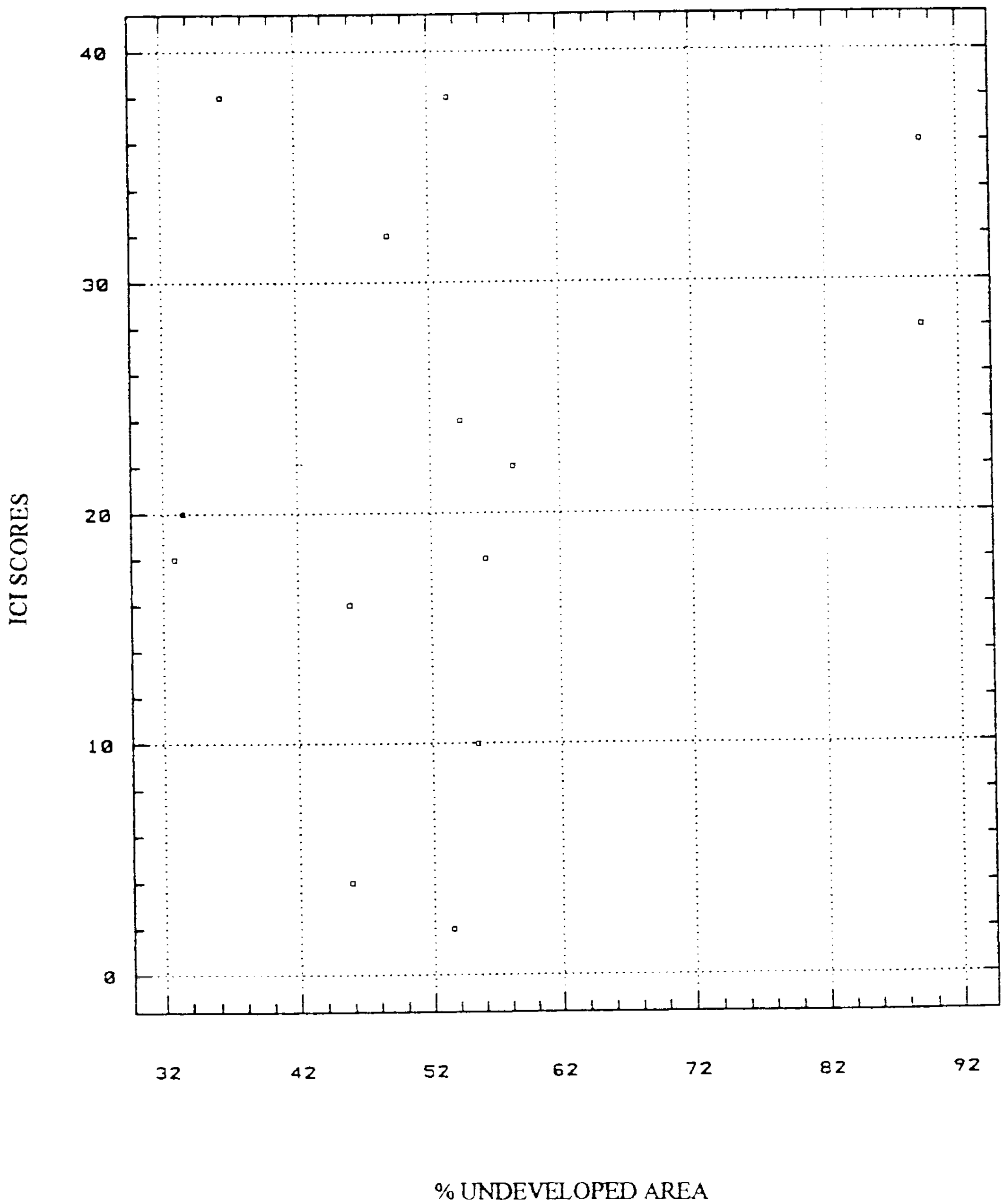
The IBI scores in the 116 and 58 meter buffers were correlated most strongly with perviousness ($R = -0.54$, $p = 0.0003$ and $R = 0.57$, $p = .0001$ respectively). This variable was followed by undeveloped area, soils, and tree canopy in the 116 meter buffer and by undeveloped area, and tree canopy in the 58 meter buffers. Associations between land use variables and individual metrics are given in Table 6.

The ICI scores were not correlated at a 95% confidence level with land use variables in the aggregate, sub-watershed, 116 meter buffer or 58 meter buffer areas in this study. The plot shown in Figure 10, ICI scores against undeveloped area in the aggregate watershed, is representative of plots for ICI scores and land use variables. Though not statistically significant and plotted with a very small data set, this plot may be able to separate sites in this study in which there is greater than 60% undeveloped land (no score less than 26). Macroinvertebrate diversity and index scores have been shown to be negatively impacted by imperviousness in other studies (Klien 1979, Schueler and Gali 1992).

Individual ICI metrics 1, 2, and 9 correlated at a 95% confidence level with various land use variables. Metric 1 correlated with grass, tree canopy, and commercial area in the aggregate watersheds. The relationship between ICI metric 1 with tree canopy and commercial area was counterintuitive (as the metric score goes up so does commercial area in the aggregate watershed, while tree canopy goes down). This may reflect the relatively better QHEI scores in the lower Mill Creek sites. Also, as discussed above, the larger size of the lower Mill Creek may provide a resiliency to urban impacts which accounts for better QHEI scores and more aquatic insect taxa at these sites. Metric 2 correlated with perviousness in the 58 meter buffers; residence area in the aggregate and sub-watersheds; commercial area in the aggregate watersheds; undeveloped area in the aggregate and sub-watersheds and the 116 and 58 meter buffers, and soils in the aggregate and sub-watersheds. However, with only 1 score above 0 in all samples, metric 2 may be considered a statistical artifact rather than an informative metric. Metric 9 correlated with perviousness in the aggregate and sub-watersheds and 116 and 58 meter buffers and with undeveloped area in the aggregate and sub-watersheds and 116 meter buffers.

Both the HBI and EPT indices correlated with land use variables only at the aggregate watershed level. Both these indices correlate best ($R = -0.49$, $p = .0091$ and R

Figure 10. Plot of ICI scores for sites vs % undeveloped area in the aggregate watersheds.



= 0.7409, p = 0.0091 respectively) with commercial area (see Fig. 8 for %EPT plot) and also with tree canopy, perviousness and undeveloped area. Again the correlation with commercial area and tree canopy was counterintuitive and the macroinvertebrate communities measured by these indices may be responding to better habitat conditions or larger stream size or both in the lower Mill Creek sites.

Only The LBI and the TSI of the 5 algal indices were correlated with land use variables. In the aggregate watersheds both LBI and TSI were best correlated with soils (R = -0.94, p = .0046 and R = 0.91, p = 0.0131 respectively). Both of these indices also correlated with residence density and the LBI also correlated with undeveloped area in the aggregate watersheds. These indices again correlated best with soils in the sub-watersheds. They also correlated again with residence density and the LBI with undeveloped area This reflected the relationship between SRP and IBI scores in the aggregate and sub-watersheds. There were only 6 data points used for these analysis, but the plots were convincing (Figs. 11 and 12).

The TSI correlated best in the buffer zones with residence density (R = 0.88, p = 0.0213 for the 116 meter zones) and the LBI with undeveloped area (R = 0.93, p = 0.0071 for the 116 meter zones). Both of these indices correlated with undeveloped area, residence density, and perviousness in the 116 and 58 meter buffer zones, except the TSI and perviousness did not correlate at a 95% confidence level in the 116 meter zone.

Finally, results of residuals analyses testing QHEI scores and tree canopy in the buffers as covariables with perviousness and soils v.s. IBI scores in the watersheds were not significant at a 95% confidence level. No covariables were identified which might affect the IBI sensitivity to soils and perviousness in the aggregate and sub-watersheds.

DISCUSSION

The IBI metrics in Ohio are calibrated using a "least impacted" reference set which establishes warm water habitat criteria for the ecoregion. There is a similarity in the way the urban stream fish communities in this study depart from the reference set for the ecoregion. In particular, the urban systems are composed of a greater number of tolerant fish species and a reduced number of sensitive fish species when compared with the reference set. Metric 5, the sensitive species metric, never scored above a 1 in any sample suggesting that sensitive species are rare if not extirpated in all of these sites which are almost all largely dominated by tolerant species, measured by metric 6. This similarity in the type and degree of fish community response in the urban systems accounts for the failure of six IBI metrics to effectively distinguish differences in the study sites. The IBI in these degraded systems functions mainly as a richness measure which is associated with land use impacts. The IBI index score was able to discriminate Salt Run site 3 from the other more impacted sites in the study, but discrimination between sites beyond this is not clear.

Figure 11. Plot of LBI scores for sites vs soils (see methods) in the aggregate watersheds.

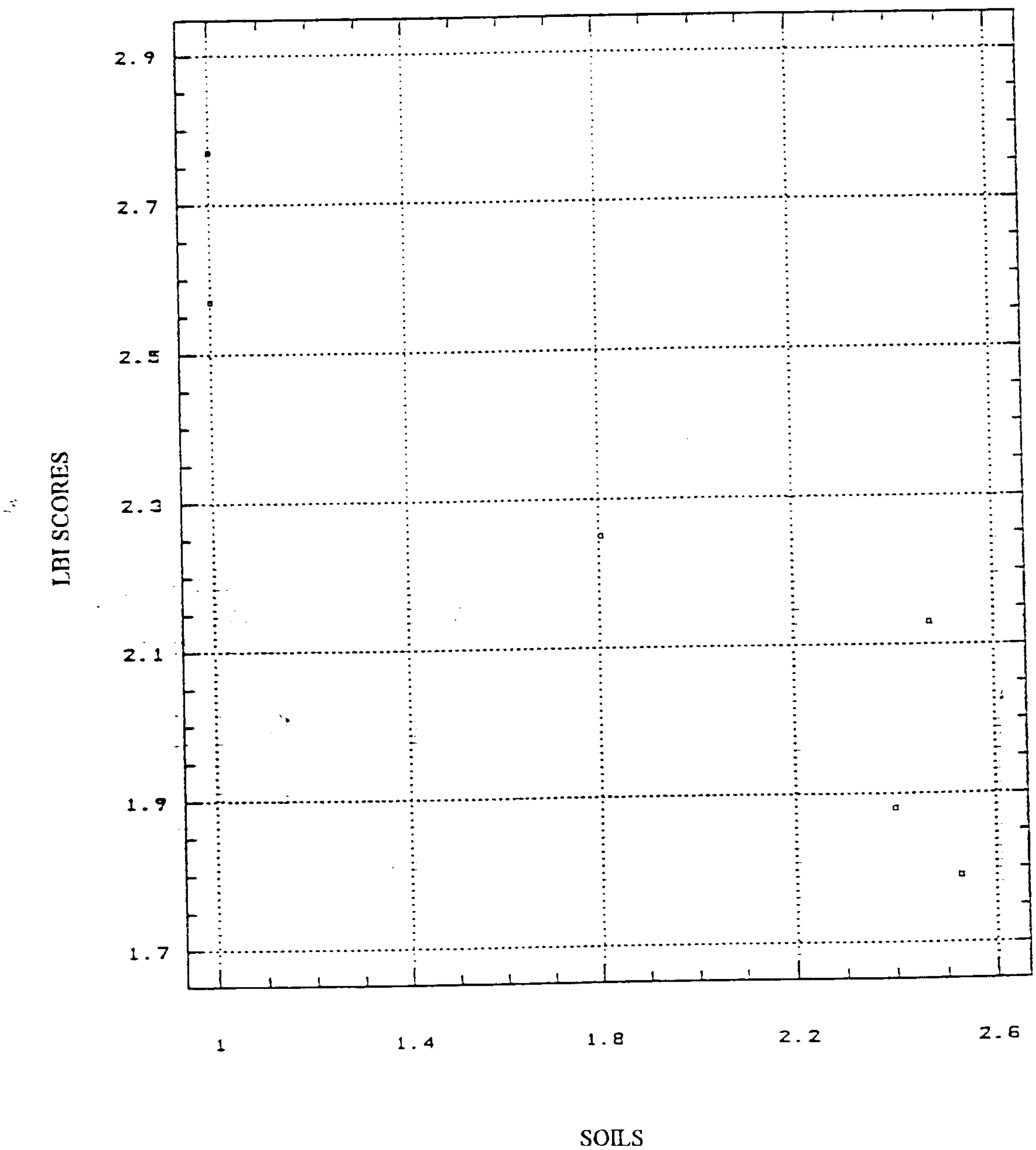
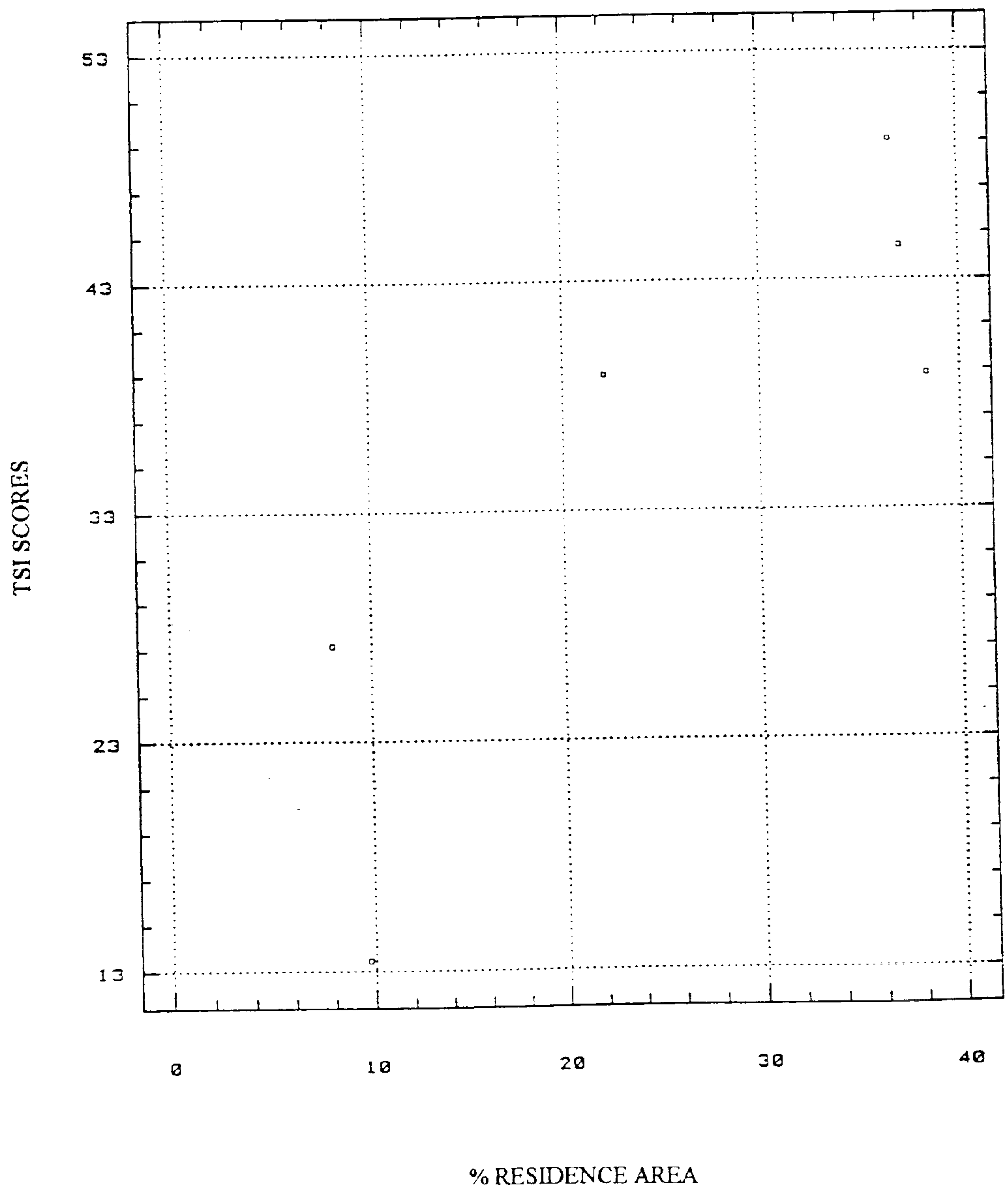


Figure 12. Plot of TSI scores for sites vs % residence area in the aggregate watersheds.



It is difficult to interpret the ICI scores for these systems. The ICI does not statistically correlate with the IBI nor does it correlate with land use variables or water chemistry in these systems. Most of the variance is accounted for in only the Dipteran and Caddisfly metrics. The other individual metrics which comprise the ICI give scant information on which to interpret the relationship between the macroinvertebrates and the environment of these systems. The ICI may separate greater urban impacts from lesser urban impacts (such as amount of undeveloped area) based on examination of scatter plots, but does not clearly indicate the Salt Run 3 site as the least impacted site. The %EPT and HBI, the other macroinvertebrate indices evaluated in this study reveal a provocative counterintuitive relationship with commercial area in this study and may be responding to the size of the lower Mill Creek or perhaps to habitat of these sites, though there is no correlation between these indices and QHEI scores.

Five diatom indices were also evaluated in this study. These indices were able to discern Salt Run 3 as the least impacted site. The LBI and TSI correlated statistically with land use variables suggesting that these metrics respond to urban impacts within these degraded urban streams, though there was no correlation between these measures and water chemistry in this data set.

The TSI correlates with the IBI index scores in these systems. Except for this correlation, the fish index , the macro invertebrate indices, and the diatom indices do not appear to strongly correlate between organismal groups. This means that for the degraded sites, the relative level of stream system health at a site indicated by one organismal group may not serve as a proxy for the relative level of stream system health at that site indicated by another organismal group. For example, Doan Brook site 5 had the best ICI score and the worst LBI score.

The traditional biological metrics are limited in their ability to discern urban impacts in these highly degraded systems. Even biological metrics and indices which seem to be associated with land use impacts in this study, such as the IBI, do not clearly distinguish between the highly degraded sites. The traditional metrics function well in assessing use attainment, but would be ambiguous measures of effectiveness of incremental remediation efforts.

The Washington DC NRUP study (NVPDC 1979) reports no correlation between SS and imperviousness and concludes that SS is a function of watershed size. The hypothesis is that perviousness in a watershed contributes little to stream sediments which are determined mostly by bank erosion. Larger watersheds have more bank length to erode than smaller watersheds. The steep undeveloped banks of Salt Run in the park area along with a possible store of sediments from past agricultural activities may contribute relatively high SS concentrations and thus the counterintuitive positive relationships with biological index scores found in this study.

Other water chemistry pollutants such as SRP are related to imperviousness (Schueler 1987, USEPA 1983). There were correlations between every other water chemistry parameter measured and at every level of watershed area examined in this study. Some of these correlations were convincing especially the association between Cl concentrations and perviousness, and SRP and tree canopy. However, the relationship between water chemistry and the biological measures was less clear.

Water chemistry impact is a factor in urban stream degradation that is not currently measured in a way that has meaning for biological integrity of urban streams. The IBI and ICI index scores do not clearly reflect water chemistry changes. The best measures of water chemistry impacts in these systems appear to be the IBI index which correlates with Cl and SRP in these systems and metric 12 (DELT anomalies) which correlates with several water chemistry and land use metrics, but again it is not clear which of these associations has a causal link with these measures. Diatoms, though not correlated with water chemistry in this study, should be further tested as a water chemistry monitor in these urban streams because of its effectiveness as a pollution monitor in other systems.

The process of urbanization brings a suite of assaults to aquatic life including altered hydrology, geomorphology, habitat, and water chemistry (Schueler 1987). The response of aquatic life to these assaults is complex and can be measured at the individual, population, community, ecosystem, and landscape levels. Urban land use metrics can broadly summarize these assaults and can be used as predictors of biological response due to landscape level effects. For example, much can be explained by imperviousness because of its effect on hydrology, geomorphology, sediment load, and chemistry in an urban watershed. A 10% - 15% watershed imperviousness measure is sited as a threshold for aquatic life degradation (Schueler 1982).

The land use metrics are useful for examining landscape level effects, however, they cannot be the basis of a classification by which to calibrate biocriteria until a reasonable model is developed for the complex interplay of cause and effect in these systems. Though it is probably true that an urbanized stream will not be able to achieve warm water habitat criteria, it would be difficult at this point to pick a proxy with which to measure urbanization. Based on this study the correlations between land use and the biological measures vary depending on what biological measures were being used and what landscape level was being studied.

Some sites such as the lower Mill Creek sites and the upper Salt Run sites can be viewed as outliers in this study as they performed biologically better or worse respectively than would be expected given the current knowledge of these sites. The existence of covariables which could explain these outliers must exist at some level in the watersheds, but were not revealed in this study. Tree canopy in the buffer zones as well as QHEI scores are not associated in this study with biological metrics or with other land use measures such as perviousness or soils as covariables.

The associations between the biological metrics and land use examined in this study are engaging, but they are correlative and provide little information with which to interpret differences in biological metric scores. Rather, several questions are raised by these correlations, for example: what is the causal relationship between land use and biological measures; why do some biological metrics correlate with land use while others do not; what is the cause of counterintuitive correlations; what is the nature of redundancy in land use correlations with biological metrics; how are the shifting correlations between biological metrics and land use at different watershed levels to be interpreted; why is there a lack of correlation between QHEI scores and biological metrics in these sites; and to what extent are water chemistry associations with biological metrics redundant?

A very quantitative measure of land use was necessary for this study. The usual methods of quantifying land use, for example, circling residential areas on a map, lose much biologically relevant information. The method used in this study of assessing the status of the landscape by classifying imagery from the Landsat Thematic Mapper and from Digital Aerial Photography generated a flexible and quantitative data set well suited to the heterogeneous nature of these small urban watersheds.

Future study should focus on the causal links between urban impacts and the stream biota. Specific research to be undertaken might include:

1) statistical techniques such as path analysis using a more expansive data set which includes at least 100 sites, hydrology data, and data from additional biological organisms to examine causation between variables;

2) manipulative studies which might include artificial stream studies;

3) process based studies of aquatic life and stream and riparian habitat characteristics;

4) paired monitoring protocols involving sites undergoing different remediation efforts or BMPs;

5) an effort to identify and define features of urban stream outliers (sites which perform better or worse than expected) in terms of biological integrity;

6) comparative studies of stream reaches which involves detailed assessments, temporal changes, and spatial studies such as patch analysis to determine watershed scale impacts;

7) assessing remediation efforts by targeting specific biological goals (not necessarily the traditional metrics) appropriate for the site;

8) additional measures of urban ecosystem health should be explored with GIS

technology, particularly those that could be incorporated into a predictive model for the effects of urban land use on the biota.

Finally, this study was conducted as a pilot project using a statistically limited data set. Though it serves well as an exploratory study with many intriguing findings, it is unable to provide strong evidence to support these findings. A purpose of the pilot project is to foreshadow were future efforts along these lines could lead. Exploring the relationship between the measures of biological integrity and features of urbanization as done in this study refines our ability to set goals for urban streams and leads to the development of management tools which can assist in the decision process of remediation efforts. A predictive model for urban streams with causal links identified should be the goal.

Table 1. Characteristics of the study streams (NEORSD 1997, OEPA 1999b).

Name	Use Designation	Length km.	Drainage Area $km.^2$
Doan Brook	WWH, AWS, IWS, PCR	13.0	30.3
Mill Creek	WWH, AWS, IWS, PCR	14.5	46.8
Salt Run	SRW, WWH, AWS, IWS, PCR		

WWH - Warm Water Habitat
AWS - Agricultural Water Supply
IWS - Industrial Water Supply
PCR - Primary Contact Recreation
SRW - State Resource Water

Table 2. Interpretation of biological indices and metrics analyzed in this study (NEORSD 1995, OEPA 1987, Sgro and Johansen 1998)

Metric or **Index**	Interpretation
IBI	Compared with reference conditions; higher scores equate with better conditions
1) Total number of indigenous fish species	Positively correlated with environmental quality
2) Number of darter species	Sensitive to physical and chemical disturbance; Positively correlated with environmental quality
3) Proportion of headwater species	Indicates permanent habitat with low environmental stress; positively correlated with environmental quality
4) Number of minnow species	Positively correlated with environmental quality
5) Number of sensitive species	Indicates stress due to perturbations, loss of habitat, lack of water; positively correlated with environmental quality
6) Percent abundance of tolerant species	Increase as a proportion of the community in degraded conditions; negatively correlated with environmental quality
7) Percent of omnivores	Indicates disruption of food chain; negatively correlated with environmental quality
8) Proportion of insectivores	Indicates degradation of insect food base; positively correlated with environmental quality
9) Number of pioneering species	Indicates habitat under stress; negatively correlated with environmental quality
10) Number of individuals in a sample	Decreases when trophic relationships are disturbed; positively correlated with environmental quality

11) Number of simple lithophilic species	Need clean gravel and substrate particles to spawn; positively correlated with environmental quality
12) Number of individuals with deformities, eroded fins, lesions, and tumors	Negatively correlated with environmental quality
ICI	Compared with reference conditions; higher scores equate with better conditions
1) Total number of taxa	Positively correlated with environmental quality
2) Number of Mayfly taxa	Pollution sensitive taxa; positively correlated with environmental quality
3) Number of Caddisfly taxa	Need optimal habitat and appropriate food type; positively correlated with environmental quality
4) Number of Dipteran taxa	Have range of pollution tolerances; positively correlated with environmental quality
5) Percent Mayflies	Pollution sensitive taxa; positively correlated with environmental quality
6) Percent Caddisflies	Need optimal habitat and appropriate food type; positively correlated with environmental quality
7) Percent Tanitarsini Midges	Very pollution sensitive; positively correlated with environmental quality
8) Percent other Diptera and non-insects	Become predominant under adverse water quality conditions; negatively correlated with environmental quality
9) Percent tolerant organisms	Tolerant to organic pollution and toxic impact; negatively correlated with environmental quality

10) Qualitative EPT taxa	A measurement of habitat quality and type; positively correlated with environmental quality
HBI	Indicator of organic and nutrient pollution: 0.00 - 3.5 Excellent 3.51 - 4.50 Very Good 4.51 - 5.50 Good 5.51 - 6.50 Fair 6.51 - 7.50 Fairly Poor 7.51 - 8.50 Poor 8.51 - 10.00 Very Poor
%EPT	Positively correlated with environmental quality
TDI	Indicator of eutrophication; negatively correlated with environmental quality
LBI	Correlates with BOD; positively correlated with environmental quality
SSI	Correlates with concentrations of PO_4, NH_2, NO_3, and NO_2 in Lake Erie estuaries; positively correlated with environmental quality
TSI	Correlates with organic pollution, used along with TDI to assess how much eutrophication is the result of organic pollution: $<$20% Free of organic pollution 21 - 40% Some evidence of organic pollution 41 - 60% Organic pollution contributes significantly to eutrophication $>$61% Heavily contaminated with organic pollution
GDI	Indicator of general pollution; positively correlated with environmental quality

Table 3. Water chemistry values for study sites (NEORSD 1994-95, CRWP 1997-98) (S = Salt Run; D = Doan Brook; M = Mill Creek).

Site	Date	Cond Units	NH_3 mg/L	Cl mg/L	NO_{2+3} mg/L	SRP mg/L	SS mg/L
S1	Sep. 97	880	0.0	49.3	0.49	0.0	9.5
S1	May 98	773	0.045	32.2	0.15	0.0	4.2
S2	Sep. 97	878	0.042	7.5	0.28	0.0	5.4
S2	May 98	702	0.062	5.0	0.14	0.0	9.4
S3	Sep.97	815	0.028	31.9	0.06	0.0	8.1
S3	May 98	673	0.068	36.8	0.07	0.0	6.8
S3	Jun. 98	660	0.068	32.9	0.14	0.006	91.5
D1	May 98	746	0.058	97.0	1.01	0.143	1.3
D1	Jun. 98	580	0.058	70.3	0.92	0.045	20.7
D2	May 98	813	0.0	108.1	0.04	0.011	1.3
D4	Aug. 94	NA	0.002	126.0	NA	0.04	1.0
D4	May 98	641	0.066	95.5	0.25	0.01	3.9
D5	May 98	640	0.165	103.5	0.15	0.007	0.1
D5	Jun. 98	379	0.182	52.2	0.24	0.038	22.2
D6	Aug. 94	NA	0.5	198.0	NA	0.24	2.0
D6	May 98	901	0.017	133.6	0.63	0.019	1.9
D7	May 98	1152	0.07	182.1	0.54	0.03	2.4
M2	May 95	NA	0.1	358.0	NA	0.03	6.0
M2	Sep. 97	NA	0.0	85.3	2.39	0.088	25.7
M2	May 98	1066	0.036	145.3	0.13	0.003	4.9
M2	Jun. 98	846	0.056	145.0	0.38	0.005	60.1
M4	May 95	NA	0.1	228.0	NA	0.06	5.0
M4	Sep. 97	637	0.0	76.5	1.95	0.073	31
M4	May 98	1090	0.047	157.3	0.27	0.017	2.9
M9	May 95	NA	1.2	322.0	NA	0.01	18.0
M9	May 98	1319	0.631	173.0	1.9	0.001	12.6

Table 4. Algal index scores for 6 sites sampled in 1998 (CRWP 1998). (SR=Salt Run, DB=Doan Brook, ML=Mill Creek; **bold** = best, <u>underline</u> = worst).

Site	TDI	LBI	SSI	GDI	TSI
SR1	4.01	2.62	26.31	3.33	27.18
SR3	**2.85**	**2.77**	**53.12**	**3.9**	**13.45**
DB1	3.72	2.13	25.69	3.03	44.42
DB5	<u>4.36</u>	<u>1.79</u>	11.57	<u>2.25</u>	<u>49.09</u>
ML2	3.87	2.25	23.59	2.86	38.87
ML9	3.86	1.88	<u>10.23</u>	3.03	44.42

Table 5. Correlations between algal indices based on linear regression of 6 1998 study site samples.

TDI v.s. LBI	(R^2=50.74, p=.1123) negative correlation
LBI v.s. TSI	(R^2=80.51, p=.0153) negative correlation
TDI v.s. TSI	(R^2=70.55, p=.0364)
GDI v.s. TSI	(R^2=20.27, p=.3704) negative correlation
SSI v.s. TSI	(R^2=73.85, p=.0283) negative correlation
SSI v.s. LBI	(R^2=80.88, p=.0145)
SSI v.s. TDI	(R^2=78.39, p=.0190) negative correlation
SSI v.s. GDI	(R^2=41.46, p=.1677)
GDI v.s. TDI	(R^2=62.94, p=.0596) negative correlation
GDI v.s. LBI	(R^2=46.01, p=.1386)

Table 6. Associations based on correlation analysis (α = 0.05) between individual IBI metrics and land use variables in the aggregate (A), Sub-watershed (S), 116 meter buffers (116), and 58 meter buffer (58) areas.

IBI	Tree	Residence	Undeveloped	Soils	Commercial	Perviousness	Grass
1	-	-	116, 58	A, S	-	116, 58	-
2	116, 58	A, S	A, S, 116, 58	A, S, 116	A	A, S, 116, 58	-
3	-	-	58	A, S	-	-	-
4	-	A	-	A, S	-	-	-
5	-	-	-	-	-	-	-
6	-	-	-	-	-	-	-
7	-	-	-	-	-	-	A,S
8	116	-	-	-	-	-	-
9	-	-	-	-	-	-	-
10	116	S, 116	A, S, 116, 58	A, S, 116, 58	S	A, S, 116, 58	-
11	-	S, 116, 58	A, S, 116, 58	A, S, 116	-	A, S, 116, 58	-
12	116, 58	S	A, S, 116, 58	A, S, 116, 58	A, S	A, S, 116, 58	-

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Appendix A

Indices and Metrics

Sources: Curtis (1992), Cuyahoga River Watershed Project (1998), Northeast Ohio Regional Sewer District (1997), Ohio Environmental Protection Agency (1999a), and Stewart et al. (1998)

Appendix A - Table 1. Index of Biological Integrity index and metric scores (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers).

Site	Date	IBI-1	IBI-2	IBI-3	IBI-4	IBI-5	IBI-6
S1	11/97	1	1	1	1	1	1
S1	6/98	1	1	1	1	1	1
S2	11/97	3	1	1	1	1	1
S2	6/98	1	1	1	1	1	1
S3	11/97	3	3	3	5	1	1
S3	6/98	5	3	5	5	1	3
S3	92	3	1	3	3	1	1
M1	7/95	1	1	1	1	1	1
M1	9/95	1	1	1	1	1	1
M2	7/95	1	1	1	1	1	1
M2	9/95	1	1	1	1	1	1
M2	6/98	1	1	1	1	1	1
M3	7/95	1	1	1	1	1	1
M3	9/95	1	1	1	1	1	1
M4	7/95	1	1	1	1	1	1
M4	9/95	1	1	1	1	1	1
M4	6/98	1	1	1	3	1	1
M5	7/95	1	1	1	1	1	1
M5	9/95	1	1	1	1	1	1
M6	6/95	1	1	1	1	1	1
M6	8/95	1	1	1	1	1	1
M7	7/95	1	1	1	1	1	1
M7	9/95	1	1	1	3	1	1
M8	6/95	1	1	1	1	1	1
M8	8/95	3	1	3	1	1	3
M9	8/91	3	1	3	5	1	1
M9	6/95	1	1	1	1	1	3
M9	7/95	3	1	3	1	1	1
M9	6/98	3	1	1	3	1	1
D1	7/97	1	1	1	1	1	1
D1	6/98	1	1	1	1	1	1
D2	7/97	1	1	1	1	1	1
D3	7/97	1	1	1	1	1	1
D3	6/98	1	1	1	1	1	1
D4	7/97	1	1	1	1	1	1
D5	7/97	1	1	1	1	1	1
D5	6/98	1	1	1	1	1	1

Site	Date	IBI-1	IBI-2	IBI-3	IBI-4	IBI-5	IBI-6
D6	6/98	1	1	1	1	1	1
D7	7/97	1	1	1	1	1	1
D7	6/98	1	1	1	1	1	1

Appendix A Table 1. Continued.

Site	Date	IBI-7	IBI-8	IBI-9	IBI-10	IBI-11	IBI-12	IBI-T
S1	11/97	1	1	5	1	1	5	20
S1	6/98	1	1	5	1	3	5	22
S2	11/97	1	1	3	1	3	5	22
S2	6/98	1	1	3	1	3	5	20
S3	11/97	1	3	1	3	3	5	32
S3	6/98	3	5	1	5	3	5	44
S3	92	3	3	3	5	5	3	34
M1	7/95	1	1	1	1	1	1	12
M1	9/95	1	1	1	1	1	1	12
M2	7/95	5	1	1	1	1	1	16
M2	9/95	5	1	1	1	1	3	18
M2	6/98	1	1	1	1	1	5	16
M3	7/95	1	1	1	1	1	1	12
M3	9/95	1	1	3	1	1	1	14
M4	7/95	1	1	1	1	1	1	12
M4	9/95	5	1	1	1	1	5	20
M4	6/98	1	1	3	1	1	1	16
M5	7/95	5	1	1	1	1	5	20
M5	9/95	5	1	1	1	1	3	18
M6	6/95	1	1	1	1	1	1	12
M6	8/95	1	1	1	1	1	1	12
M7	7/95	1	1	1	1	1	1	12
M7	9/95	5	1	5	1	1	5	26
M8	6/95	1	1	1	1	1	1	12
M8	8/95	1	1	3	3	3	3	26
M9	8/91	1	1	5	1	1	1	24
M9	6/95	1	1	1	1	1	1	14
M9	7/95	1	3	1	1	5	1	22
M9	6/98	1	3	5	1	1	5	26
D1	7/97	1	1	1	1	1	1	12
D1	6/98	1	1	1	1	1	1	12
D2	7/97	1	1	1	1	1	5	16
D3	7/97	1	1	1	1	1	5	16
D3	6/98	1	5	3	1	1	5	22
D4	7/97	1	1	1	1	1	5	16
D5	7/97	1	1	1	1	1	5	16
D5	6/98	1	5	5	1	1	1	20
D6	7/97	1	1	1	1	1	1	12

Site	Date	IBI-7	IBI-8	IBI-9	IBI-10	IBI-11	IBI-12	IBI-T
D6	6/98	1	1	1	1	1	1	12
D7	7/97	1	1	1	1	1	1	12
D7	6/98	1	1	1	1	1	1	12

Appendix A Table 2. Invertebrate Community Index and metric scores (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

Site	Date	ICI-1	ICI-2	ICI-3	ICI-4	ICI-5	ICI-6
S1	NA	NA	NA	NA	NA	NA	NA
S2	NA	NA	NA	NA	NA	NA	NA
S3	7/94	2	2	6	4	2	6
S3	6/95	2	0	6	6	2	6
D1	8/98	0	0	0	2	0	0
D2	8/98	2	0	0	2	0	0
D3	8/98	2	0	6	0	2	6
D4	8/98	2	0	0	4	0	0
D5	9/98	4	0	4	6	2	6
D6	9/98	2	0	4	4	0	6
D7	8/97	2	0	0	2	0	0
D7	9/98	2	0	2	4	2	2
M1	NA	NA	N	N	N	N	N
M2	9/95	6	0	0	6	0	0
M3	NA	NA	NA	NA	NA	NA	NA
M4	NA	NA	NA	NA	NA	NA	NA
M5	NA	NA	NA	NA	NA	NA	NA
M6	NA	NA	NA	NA	NA	NA	NA
M7	9/95	6	0	6	6	6	6
M8	8/95	4	0	2	6	0	2
M9	8/95	2	0	4	4	2	2

Appendix A Table 2. Continued.

Site	Date	ICI-7	ICI-8	ICI-9	ICI-10	Total
S1	NA	NA	NA	NA	NA	NA
S2	NA	NA	NA	NA	NA	NA
S3	7/94	2	0	6	6	36
S3	6/95	2	0	4	0	28
D1	8/98	0	0	0	0	2
D2	8/98	2	0	4	0	10
D3	8/98	0	0	6	2	24
D4	8/98	4	0	6	2	18
D5	9/98	6	2	6	2	38
D6	9/98	6	2	6	2	32
D7	8/97	0	0	0	0	4
D7	9/98	2	0	0	2	16
M1	NA	NA	NA	NA	NA	NA
M2	9/95	4	0	4	2	22
M3	NA	NA	NA	NA	NA	NA
M4	NA	NA	NA	NA	NA	NA
M5	NA	NA	NA	NA	NA	NA
M6	NA	NA	NA	NA	NA	NA
M7	9/95	2	2	2	2	38
M8	8/95	2	0	0	4	20
M9	8/95	2	0	0	2	18

Appendix A Table 3. Trophic Diatom Index (TDI), Langa-Bertalot Index (LBI), %Tolerant Species Index (TSI), Generic Diatom Index (GDI), and Sensitive Species Index (SSI) scores (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

Site	Date	TDI	LBI	TSI%	GDI	SSI
S1	8/98	4.07	2.57	27.18	3.33	26.31
S2		NA	NA	NA	NA	NA
S3	8/98	2.85	2.77	13.45	3.9	53.12
D1	8/98	3.72	2.13	44.42	3.03	25.69
D2		NA	NA	NA	NA	NA
D3		NA	NA	NA	NA	NA
D4		NA	NA	NA	NA	NA
D5	9/98	4.36	1.79	49.09	2.25	11.57
D6		NA	NA	NA	NA	NA
D7		NA	NA	NA	NA	NA
M1		NA	NA	NA	NA	NA
M2	8-9/98	3.87	2.25	38.87	2.86	23.59
M3		NA	NA	NA	NA	NA
M4		NA	NA	NA	NA	NA
M5		NA	NA	NA	NA	NA
M6		NA	NA	NA	NA	NA
M7		NA	NA	NA	NA	NA
M8		NA	NA	NA	NA	NA
M9	8/98	3.86	1.88	38.85	3.51	10.23

Appendix A Table 4. Percent Ephemeroptera, Plecoptera, Trichoptera taxa index (%EPT) and Hilsenhoff Biotic Index (HBI) (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

Site	Date	%EP	HB
S1			N
S2			N
S3			N
M1			N
M2	9/95	55.	5.4
M3	9/95	7.	6.75
M4	9/95	21.4	6.29
M5	9/95	38.3	5.63
M6	9/95	82.7	4.56
M7	9/95	61.7	5.5
M8	9/95	55.	5.06
M9	9/95	54.1	4.77
D1	8/98	NA	6.0
D2	6/98	NA	6.48
D2	7/98	NA	5.18
D3	9/94	NA	5.42
D3	6/97	NA	5.98
D3	8/97	NA	5.35
D3	6/98	6.27	
D3	8/98	NA	5.84
D4	9/94	14.29	6.2
D4	8/98	NA	6.06
D5	9/98	NA	6.26
D6	9/94		7.16
D6	6/97	NA	6.01
D6	8/97	NA	5.93
D6	9/98	NA	6.35
D7	9/98	NA	6.23

Appendix A Table 5. Qualitative Habitat Evaluation Index (QHEI) (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

Site	Date	QHEI
S1		NA
S2		NA
S3	6/94	57.75
S3	6/95	53
D1	8/98	61
D2	8/98	61
D3	9/94	56.75
D4	9/98	62.5
D5	9/98	69.25
D6	9/98	66.25
D7	8/98	57.75
M1	9/95	56.5
M2	9/95	78
M3	9/95	74
M4	9/95	62.25
M5	9/95	62
M6	9/95	70.25
M7	9/95	69.5
M8	8/95	70.5
M9	8/95	72

Appendix A Table 6. Land use values of the aggregate watersheds given as mean percentages as described in methods. (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

site	area	canopy	grass	pervious	residence	com.	undevel.	soils
D1	1393881.8	57.692	33.924	75.054	37.029	9.329	53.465	2.471
D2	4064666.5	43.16	41.307	68.519	31.826	12.554	55.509	2.337
D3	5589684	46.678	38.18	69.602	34.156	11.405	54.324	2.385
D4	2997610.7	54.664	37.335	75.232	34.895	8.766	56.16	2.381
D5	14791892	51.801	34.815	71.572	36.561	9.805	53.506	2.529
D6	20697742	47.968	32.589	66.248	38.697	12.306	48.889	2.603
D7	24469702	45.506	31.49	62.868	41.26	12.799	45.844	2.611
M1	2003253.1	39.964	49.909	64.198	24.056	12.828	62.954	2.012
M2	5846754.5	28.11	51.851	60.404	22.129	19.427	58.316	1.806
M3	10474245	24.124	50.671	57.01	23.38	22.256	54.272	1.982
M4	13107331	23.82	48.859	54.645	25.216	24.26	50.434	2.048
M5	33004390	21.063	43.544	47.853	34.07	29.01	36.858	2.323
M6	38118092	21.063	42.944	47.529	34.806	28.778	36.358	2.351
M7	39492176	21.313	42.778	47.539	34.995	28.461	36.485	2.353
M8	49047584	21.599	40.04	45.222	38.247	28.222	33.478	2.399
M9	51793516	21.293	39.812	44.688	38.319	28.77	32.859	2.4
S1	2553174.2	54.375	35.845	96.699	8.003	0.693	91.287	1
S2	1376873	45.204	47.155	95.804	16.349	0.846	82.788	1
S3	6690406.5	52.437	40.402	96.931	9.74	0.99	89.245	1

Appendix A Table 7. Land use values of the sub-watersheds given as mean percentages as described in methods. (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

site	area	canopy	grass	pervious	residence	com.	undevel.	soils
D1	1393881.8	57.692	33.924	75.054	37.029	9.329	53.465	2.471
D2	4064666.5	43.16	41.307	68.519	31.826	12.554	55.509	2.337
D3	1525017.2	56.019	29.798	72.453	40.408	8.332	51.135	2.513
D4	1603718.3	52.032	40.301	75.387	33.039	8.277	58.503	2.302
D5	6204608.5	55.069	30.525	71.562	39.549	8.842	51.493	2.737
D6	5905849	38.431	27.051	53.001	44.011	18.531	37.401	2.787
D7	3770715.2	32.202	25.548	44.608	55.112	15.46	29.394	2.653
M1	2003253.1	39.964	49.909	64.198	24.056	12.828	62.954	2.012
M2	3843501.2	22.045	52.859	58.476	21.13	22.817	55.941	1.7
M3	4627490.5	18.886	49.147	52.549	24.976	25.951	49.028	2.204
M4	2633086.5	22.393	41.298	44.747	32.868	32.412	34.645	2.346
M5	19897058	19.256	40.065	43.427	39.935	32.105	27.916	2.503
M6	5113704	20.955	38.647	44.907	40.118	27.274	32.568	2.539
M7	1374082.1	28.057	37.093	46.543	41.434	20.091	38.43	2.437
M8	9555409	22.824	29.055	36.041	51.232	27.233	21.503	2.577
M9	2745933.5	15.961	35.838	35.4	39.568	38.311	22.088	2.43
S1	2553174.2	54.375	35.845	96.699	8.003	0.693	91.287	1
S2	1376873	45.204	47.155	95.804	16.349	0.846	82.788	1
S3	2760359.2	54.494	41.05	97.752	7.83	1.345	90.789	1

Appendix A Table 8. Land use values of the 116 meter buffer zones given as mean percentages as described in methods. (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

site	area	canopy	grass	pervious	residence	com.	undevel.	soils
D1	1393881.8	43.974	38.746	44.896	14.373	1.842	37.142	2.358
D2	4064666.5	26.474	47.364	44.21	6.632	1.576	32.1	1.496
D3	1525017.2	49.347	35.019	65.181	28.284	2.611	43.439	1.737
D4	1603718.3	52.384	39.942	71.691	27.071	5.324	54.216	1.985
D5	6204608.5	44.774	31.808	50.26	18.217	1.388	31.08	1.809
D6	5905849	17.53	22.384	26.49	12.886	2.049	12.262	2.09
D7	3770715.2	36.095	25.191	42.909	30.29	3.297	16.571	1.71
M1	2003253.1	28.137	52.622	35.996	9.804	4.086	42.436	1.813
M2	3843501.2	19.625	54.768	52.483	7.426	6.63	40.264	1.204
M3	4627490.5	12.017	38.727	27.046	6.077	4.216	14.857	1.646
M4	2633086.5	24.785	36.884	36.231	14.755	12.793	17.755	1.864
M5	19897058	18.066	35.413	30.225	16.531	10.462	11.669	2.157
M6	5113704	16.316	33.553	24.246	7.988	3.497	8.713	1.911
M7	1374082.1	25.17	33.503	27.98	18.065	6.425	15.757	2.137
M8	9555409	17.24	26.256	16.458	14.031	4.646	6.122	1.433
M9	2745933.5	14.493	35.449	23.933	13.151	16.244	10.481	2.738
S1	2553174.2	51.804	35.512	96.723	5.968	0.516	89.016	1
S2	1376873	43.912	45.226	96.629	10.546	0.58	82.093	1
S3	2760359.2	51.513	39.794	98.048	4.333	0.911	89.053	1

Appendix A Table 9. Land use values of the 58 meter buffer zones given as mean percentages as described in methods. (S=Salt Run, M=Mill Creek, D=Doan Brook; numbers following site codes indicate site numbers; NA=not available).

site	area	canopy	grass	pervious	residence	com.	undevel.	soils
D1	1393881.8	41.381	34.262	32.635	9.537	0.734	27.458	2.178
D2	4064666.5	19.303	46.327	32.578	3.214	0.585	23.455	1.257
D3	1525017.2	44.22	34.785	56.965	23.211	1.575	36.397	1.44
D4	1603718.3	48.899	39.755	64.376	21.841	3.321	50.82	1.736
D5	6204608.5	38.217	29.842	39.125	13.68	0.823	23.405	1.511
D6	5905849	12.757	17.36	18.794	7.1	0.888	7.999	1.707
D7	3770715.2	32.318	21.365	34.832	20.93	1.839	11.446	1.299
M1	2003253.1	27.172	50.129	27.014	5.04	2.208	31.619	1.774
M2	3843501.2	15.604	48.381	40.467	4.758	3.081	28.434	1.171
M3	4627490.5	9.263	30.865	18.865	3.713	2.656	8.737	1.502
M4	2633086.5	22.119	32.002	28.029	9.817	6.973	10.747	1.645
M5	19897058	15.226	31.165	23.131	11.152	5.943	7.536	2.059
M6	5113704	11.718	24.915	15.202	4.948	1.901	4.584	1.816
M7	1374082.1	22.731	28.902	19.486	12.269	4.034	9.113	2.018
M8	9555409	13.314	20.771	10.818	9.68	2.015	3.816	1.345
M9	2745933.5	11.791	32.618	18.015	8.371	9.889	7.133	2.806
S1	2553174.2	46.872	35.706	97.317	3.261	0.313	85.923	1
S2	1376873	41.869	42.338	97.175	5.254	0.418	79.337	1
S3	2760359.2	44.861	39.7	97.605	2.777	0.712	85.898	1

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EVALUATION OF STANDARD BIOTIC METRICS IN URBANIZED WATERSHEDS