Bioassessment Integration Study:
Systems Ecology Evaluation of US EPA Rapid Bioassessment Protocols in New Jersey
(Macroinvertebrates, Periphyton, Fish, and Habitat)

Patrick Center Project #830

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EXECUTIVE SUMMARY

The purpose of this project was to analyze existing fish, macroinvertebrate, and algal data to develop new methods for integrated stream bioassessment protocols. The goals of the project were divided into five main tasks with analyses focusing on data collected in New Jersey and adjacent states from sites in uplands physiographic regions (Piedmont, Ridge and Valley, Highlands):

1) Compile data on macroinvertebrate, fish and algal assemblages and associated site and watershed characteristics from the New Jersey uplands and adjacent upland areas. Compile data on mussels and odonates from the New Jersey Endangered and Nongame Species Program.

2) Collate and link data to allow joint analyses of intercorrelations among assemblages. This involves linking data from different taxonomic groups from identical or nearby sites, comparing sampling protocols from different data sources and selecting data to allow joint analysis.

3) Use literature on ecological response to stressors and existing bioassessment programs to define candidate metrics of assemblage structure. Calculate these metrics of assemblage structure for different taxonomic groups and compare the correlation structure of metrics within and among taxonomic groups.

4) Analyze relationships among metrics, environmental characteristics and stressors to determine the utility of various metrics as measures of environmental stress.

5) Pilot development of alternate methods of integrated analysis of indices from different taxonomic groups, with preliminary evaluation in conjunction with staff of NJ DEP.

Data were obtained from a variety of programs, including the NJ DEP fish index of biotic integrity (FIBI) program, the NJ AMNET macroinvertebrate program, the US EPA fish sampling program used for development of the NJ FIBI, the Long Island-New Jersey (LINJ) and Delaware USGS NAWQA programs, the Neversink sampling program of The Nature Conservancy, Philadelphia bioassessment data from the Philadelphia Water Department (PWD) and a series of studies conducted by the Academy of Natural Sciences of Philadelphia (ANSP). Data were compiled for 469 fish sites, 461 macroinvertebrate sites, 202 algal sites, 40 mussel sites, and 61 odonate sites. Watershed area and land use data were acquired for most sampling sites. Additional sampling data, such as habitat scores, were compiled as well, although these were generally not available in consistent form from enough sites to use extensively. Data from different programs were resolved to consistent taxonomy. Where necessary, data were standardized to provide comparable data. Random resampling of 100 individuals from macroinvertebrate samples was used to develop consistent 100-count macroinvertebrate data. The resampled data from each program were highly correlated with the original count data, indicating that the standardization preserved most of the information in the original data. A large series of candidate macroinvertebrate and fish metrics were calculated from the consistent data set, from which a subset were selected for further analysis. The primary metrics analyzed included the taxonomic-based metrics used in the NJ FIBI and AMNET indices. Additional metrics were calculated, relating to feeding guilds, habitat use, and taxonomic group. Diatom metrics were taken from ongoing studies developing algal indicators of ecological condition.

The correlation structure of metrics from each taxonomic group was analyzed. For macroinvertebrates, these showed high covariance among the AMNET metrics. Correlations among other macroinvertebrate metrics was variable. Principal component analyses (PCA) demonstrated several primary gradients of metric response. The first correlated with many of the AMNET metrics and other metrics commonly linked to impairment. This component was strong correlated with urban land and weakly correlated with watershed area. The second reflected abundance of hydropsychid caddisflies. This component was correlated with the amount of wetland-open water in the watershed, which reflects the abundance of these filter-feeders where there are sources of particulate food. The third and fourth components reflected patterns of abundance of chironomids and non-insects. These were also correlated with the amount of wetland in the watershed.

Fish metrics were not as strongly inter-correlated, which partly reflects the different aspects of fish assemblage condition considered in developing fish metrics. One of the metrics (proportion salmonids and centrarchids) was negatively correlated with a number of the other metrics. PCA was used to define components of variation. The first component was strongly related to species richness metrics and some other metrics. It is strongly correlated with watershed area and fish habitat score. The second component was correlated with metrics reflecting species tolerance. This metric was strongly correlated with the amount of urban and agricultural land in the watershed as well as with watershed area. The third component was correlated with metrics reflecting habitat preference. This component was strongly correlated with watershed area, and the amount of forest and wetland-open water in the watershed. The fourth component was controlled by two sites with assemblage structure very different from other sites. These sites had high numbers of the mummichog (*Fundulus heteroclitus*) and low numbers of a few other tolerant species. This assemblage reflects extreme urban impairment.

There was generally low covariance between individual fish and macroinvertebrate metrics, although the number of intolerant fish species was correlated with several of the AMNET metrics reflecting macroinvertebrate tolerance. The fish principal components of metrics were very significantly correlated with macroinvertebrate principal components. However, the magnitudes of the correlations were generally low, especially for more urban sites. Generally, the relationships between fish and macroinvertebrate components was weaker than relationships between those components and land uses.

A separate analysis of the fish and macroinvertebrate assemblages at the NJ FIBI sites and associated macroinvertebrate sites was done. The FIBI score was significantly correlated with the AMNET score. The correlation was relatively low, because of high variance in each score. The

FIBI scores for several metrics depend on watershed area. Examination of scores for different sites suggests that the size adjustment may not remove all watershed size dependence, and that there is watershed area dependence in other metrics. As a result, there may be a tendency to overestimate condition of small streams and underestimate condition of larger streams.

Algae metric and algae-macroinvertebrate relationships were determined for the entire algae dataset, and for a subset of samples collected for the NJ Algal Indicators of Eutrophication project and from the Raritan River sub-basin. PCA showed strong, positive correlations of the first 3 axis with autecological measures of siltation, general disturbance, and stream characteristics, respectively. The fourth axis related to diversity. The first 2 axes correlated with a macroinvertebrate impairment axis but only the first algae PC was related to urbanization and area. Autecological and diversity loadings were similar in sub-sample analyses to the overall dataset although coefficients were greater. Relationships with macroinvertebrate measures were weaker for subset analyses than for the overall dataset.

There were relatively few mussel species caught and no mussels were caught at a number of sites. Mussel assemblages were defined on the basis of species occurrence and co-occurrence, and assemblages were related to land use and macroinvertebrate metrics and principal components. A few mussel species were found in higher quality streams, based on land use or macroinvertebrate assessments. This reflects occurrence of these species in larger streams in areas of carbonate geology in the Ridge and Valley Province. Other mussel assemblages were found across a range of stream conditions.

Evidence (adult, larvae, exuvia, ovipositioning, or mating) of 105 odonate species were seen at sampling stations. All 61 sites had adult odonates present. Although the models weren't significant, macroinvertebrate metrics and land use variables accounted for 39.3% and 47.6% of the variation in Odonate adult and larvae richness, respectively. With the exception of odonate adult richness, regression of macroinvertebrate PCs against odonate richness metrics were not significant and did not account for substantial variation in odonate richness. Odonate adult richness was positively correlated with some macroinvertebrate impairment measures but models did not account for more than 20% of variation.

The various taxonomic groups all provide information on ecological condition and impairment. As such, integrated use of information from multiple groups is appropriate. Multiple indices can provide information on different types of stress or different ecological responses. Multiple indices may also be more robust by reducing influence of high variance of single metrics on assessments. Different responses to stressors may result in impairment of some taxa without corresponding impairment of some other taxa. As a result, use of multiple indices includes both a policy decision on how to weight impairment of various taxa to form an overall rating and statistical issues on reliability of results. Variance in each metric is likely to be high enough that misclassification of some sites is an issue. There are a number of ways of incorporating multiple indices into assessments, including averaging of index results, development of single or multiple indices using metrics from different taxonomic groups, and development of a decision system to develop overall ratings from ratings of the various study taxa. Depending on how indices are integrated, assessments may be designed to decrease the frequency of false positives (i.e., classifying an unimpaired site as impaired) or to decrease the frequency of false negatives (classifying an impaired site as unimpaired). Consistency among metrics of different indices and additional analyses or sampling of questionable sites may be valuable in developing reliable rating systems. Improvements in indices are likely to be important in developing an integrated program.

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1. INTRODUCTION

1.1 Project Objectives and Overview

The US EPA's objective of the Clean Water Act is to restore and maintain the chemical, physical, and biological integrity of the nation's waters. Biological integrity has been defined as "the ability to support and maintain a balanced, integrated adaptive assemblage of organisms having species composition, diversity, and functional organization comparable to that of natural habitat of the region." (Karr 1981, Karr et al. 1986). Because chemical monitoring alone can underestimate degradation in living systems, a number of biological measurements have been developed to provide a direct assessment of resource condition. If the biota is not present at the level expected, researchers have direct confirmation that anthropogenic influences are degrading steams and the environments that they drain.

In aquatic environments, biological monitoring can be focused on a variety of assemblages. These include fish, macroinvertebrates, algae, and, less commonly, reptiles, amphibians, and wading birds. Typically, bioassessment protocols translate taxonomic monitoring data into various metrics or indices of biological integrity (IBI). These indices are a synthesis of biological attributes that change predictably when perturbations of water or habitat quality are present and numerically depict associations between human influence and biological attributes. Metrics are usually based on either taxa richness (the number of taxa found at a study site) or the percentage of individual organisms which share common biological characteristics that increase or decrease along the gradient of human influence (e.g., percentage of individuals classified as pollution tolerant).

In keeping with the mandate of the Clean Water Act, the New Jersey Department of Environmental Protection (NJ DEP) conducts chemical and biological monitoring of water bodies, including sampling of macroinvertebrates, fishes, and chemical parameters. The State has developed indices of stream condition based on macroinvertebrate monitoring and fish communities, which are used to define impairment of water bodies. In addition, the State is also investigating use of algal communities for monitoring because of their sensitivity to nutrients and other anthropogenic effects. Because of the significant effort required to restore impaired reaches, it is important that the assessment procedure makes efficient use of the various data to provide defensible assessments. Although researchers have related macroinvertebrate (Kennen 1999) and algal (Ponader and Charles 2001) taxa to water quality conditions in New Jersey streams, there is currently no procedure to integrate assessments of different taxonomic groups (i.e., fish, macroinvertebrates, and algae) within sites or to compare information among sites to provide a robust assessment of attainment.

The purpose of this project was to analyze existing fish, macroinvertebrate, and algal data to develop new methods for integrated stream bioassessment protocols. Integrated analysis has the potential to provide several benefits. Since different indices (i.e., fish, invertebrate, algae) may be sensitive to different stressors and spatial patterns of indices can reveal scale and location of disturbances, examining and integrating multiple indices can provide more specific information on causes of impairment at different sites and possibly association of impairments among sites. Additionally, assessments are more robust by avoiding false determinations of impairment by reliance on a single index, resulting in more defensible determinations. This increased understanding can lead to more efficient monitoring protocols, (e.g., using stepwise analysis to determine potentially impaired sites from limited sampling, with follow-up sampling to provide more definitive assessments) and provide managers with a more powerful tool to focus their rehabilitation efforts. The goals of the project were divided into five main tasks with analyses focusing on data collected in New Jersey from sites in uplands physiographic regions (Piedmont, Ridge and Valley, Highlands):

1) Compile data on macroinvertebrate, fish and algal assemblages and associated site and watershed characteristics from the New Jersey uplands and adjacent upland areas. Compile data on mussels and odonates from the New Jersey threatened and endangered species program.

2) Collate and link data to allow joint analyses of intercorrelations among assemblages. This involves linking data from different taxonomic groups from identical or nearby sites, comparing sampling protocols from different data sources and selecting data to allow joint analysis.

3) Use literature on ecological response to stressors and existing bioassessment programs to define candidate metrics of assemblage structure. Calculate these metrics of assemblage structure for different taxonomic groups and compare the correlation structure of metrics within and among taxonomic groups.

4) Analyze relationships among metrics, environmental characteristics and stressors to determine the utility of various metrics as measures of environmental stress.

5) Pilot development of alternate methods of integrated analysis of indices from different taxonomic groups, with preliminary evaluation in conjunction with staff of NJ DEP.

Based on sample data acquired from different groups, primary analyses were conducted on the following taxa:

A) Fish and macroinvertebrates, since a number of bioassessment programs have conducted studies of both taxa at the same or nearby stations, and since both taxa are used in the NJ Bioassessment Program;

B) Algae and macroinvertebrates, since NJ DEP has been developing an algal bioassessment program, and other groups (e.g., ANSP and NAWQA) have gathered consistent data on both taxa;

C) Unionid mussels and other macroinvertebrates, since NJ has developed a mussel sampling program; these data provide an opportunity to investigate relationships among data pertaining to different regulatory goals, i.e., biological integrity and support for species of special concern;

D) Odonates and macroinvertebrates; like the mussels, this provides information on relationships among data gathered for very different purposes.

1.2 Relationships among Different Measures of Assemblage Structure

Standard bioassessment protocols involve a series of numerical steps to derive impairment ratings from biological data. Additional steps are used in this study to improve statistical comparability among data sets and among metrics. As a result, the same assemblage may be described by a series of inter-related measures. The following types of measures are discussed in this report.

1) **Raw data** are the primary count data derived from analysis of a sample by the program protocol; typically, these involve a taxonomic identification and number of individuals counted within a standard sample size (algae, macroinvertebrates) or sample area (fish, mussels, etc.), although other types of data (numbers of anomalies of fish, etc.) may also be collected.

2) **Standardized data** are calculated from raw data so as to provide comparability among samples. Some types of standardization are routinely done within each assessment program, e.g., calculation of proportion of each taxon within the total sample. In this study, additional standardization procedures are done to provide comparability among data from different programs. These include subsampling raw counts to produce samples of the same numbers of individuals (macroinvertebrates), calculations of catch per sample area or length (fish), and resolution of taxonomic differences (i.e., to account for differences in the level of identification of some groups, or for differences in taxonomic nomenclature).

3) **Metric values** are numerical values derived from the standardized data. These typically include total proportions of different groups of taxa (e.g., those with similar feeding habits, habitat requirements, similar tolerance or intolerance to stress) and total richness of different groups of taxa. Proportions of some individual species may be defined as metrics where these are considered to be particularly indicative of impairment or reference conditions.

4) **Metric scores** are uniform levels of each metric, which collapse the range of observed values of each metric into ordinal classes. For fish metrics, these are typically scored as 1, 3 or 5, with 5 being the closest to reference conditions. For the AMNET program, scores are 0, 3 or 6, with 6 being the closest to reference conditions. Assignments of scores from metric values could be based on external criteria linking changes in metric values to classes of impairment. Typically, scores are derived from initial calibrations of metrics, with cutpoints between each score defined with respect to the range of observed values among presumed reference sites. Where variation of metrics with respect to factors other than impairment are known, different cutpoints may be defined for different levels of these factors. The most common such

adjustment is for variation in fish richness measures with stream size. For example, for the NJ FIBI, cutpoints of metric values between metric scores are defined by regressions of the richness values with watershed size on a logarithmic scale. Typically, the relationship between metric values and scores is monotonic, i.e., each metric is assumed to decrease (or increase for metrics like tolerance) with increasing impairment.

5) **Indices of integrity** are combined measures of site conditions, based on the metric scores. Usually, all metric scores for a sample are summed to produce the index value.

6) **Impairment ratings** are decisions on the state of a sample site based on the index of integrity for that site. As with the determination of the relationship between metric values and metric scores, the ratings may be defined from external information on condition of sites with various integrity indices, or by distribution of integrity scores among reference sites and other sites. This study is primarily concerned with relationships among metrics without *a priori* determinations of impairment, so that there is relatively little analysis of ratings.

7) **Adjusted metric values** are adjustments of metric values on the basis of external variables. For example, for analyses of the NJ FIBI metric values, fish species richness measures are adjusted by the richness expected for the watershed size of the sample site. This adjustment is similar to that used in defining metric scores, except that it is done on metric values and produces a continuous adjusted metric value, rather than ordinal classes.

8) **Transformed metric values** are mathematical transformations of metric values to provide a more normal distribution of data and improve linear correlations among metrics. Typical transformations are the square root of proportion metrics, and the ln of ratio metrics. Transformations of watershed variables (e.g., ln(watershed area), square root of proportional land uses) are also done to improve statistical analyses of relationships between metrics and site characteristics.

9) **Normalized values** are rescalings of values to produce a similar range and variance among different metrics. In this study, a metric value (transformed where appropriate) is normalized by subtracting the grand mean of that metric among all samples and dividing by the standard deviation among all samples. This produces a variable with mean 0 and standard deviation of 1. Normalization is convenient for comparing different metrics with different ranges of raw values. It is also convenient for interpreting a single metric value, since the normalized value automatically describes the relative position of the value among the entire sample set. For example, a normalized value of 2 indicates a value 2 standard deviations greater than the mean. Normalization of a set of metric values is analogous to calculation of metric scores from that set, in that both procedures produce data with the same range of values, simplifying comparison among metrics. The two differ in that the normalization used

here is defined on the basis of the observed distribution of values among a set of samples. In contrast, the metric scores are defined with respect to typical upper bounds (i.e., values for reference sites) for the metric, which may have been derived from a different data set or a subset of the entire data set.

10) **Principal components** (PCs) are combinations of values of metrics (usually transformed, standardized metric values for this study) for a group of samples, which are statistically uncorrelated with each other and are defined so that components partition variation among all metrics in decreasing order. PCs are useful for multivariate data sets where there are inter-correlations among the variables. Each PC expresses covariance among a group of the original variables, with the first few PCs usually expressing much of the variance in the entire data set. Thus, PCs are useful for describing inter-correlation among variables and allow analyses of a limited number of variables which express much of the variation among all variables. Mathematically, each PC is a weighted sum of normalizations of the original variables. Each variable is weighted by a coefficient for each PC. Highly inter-correlated variables will have coefficients of large absolute value on the same PC; positively-correlated variables will have coefficients of the same sign and negatively-correlated variables will have coefficients of the opposite sign.

The index of integrity produces a single index value for each sample. PCs for that sample can be considered as a group of indices, each reflecting variation among correlated groups of variables. The index of integrity is a single, unweighted sum of scores, which are a type of normalized value of the metric values. Analogously, each PC is a weighted sum of normalized scores. They differ in that the index of integrity scales to an absolute (defined by reference conditions), while the PCs scale to the distribution of metrics within the sample being analyzed. Thus, the PCs depend on the range of conditions among samples being analyzed. If the samples contain a mix of conditions approximating the distribution among streams, including reference sites, the PCs can be treated as measures of biological condition.

1.3 Organization of the Report

A general strategy of analysis was used in this study for each of the taxonomic groups and for the analysis of relationships among groups. The following steps were used:

1) Data from programs with sampling of multiple taxonomic groups were compiled. Initial data management was used to convert data into consistent formats.

2) Protocols of the different sampling programs were compared to determine comparability. Based on these comparisons, data were either standardized for use in further analysis or were excluded from further analysis. For macroinvertebrates, this involved extensive subsampling of data to provide comparisons across similar sample sizes.

3) Candidate metrics were defined and calculated for each taxonomic group. For fish and macroinvertebrates, metrics used in various existing bioassessment programs (including the NJ FIBI and AMNET) and additional metrics were used. Variant metrics for the same basic property (e.g., measures of number of intolerant species of fish using different lists of intolerant species) were included in the group of candidate metrics. For algae, metrics were derived from ongoing analyses of algal metrics at ANS. Because of the low number of mussel taxa found, *ad hoc* assemblages were defined for further use.

4) Inter-correlations among candidate metrics were calculated and used to select a subset for subsequent analysis. Typically, single metrics among groups of similar, highly-correlated metrics were selected. Metrics used by AMNET and the NJ FIBI were retained. Correlation analysis and principal component analysis (PCA) were the main tools for these analyses.

5) Principal components were defined for the primary set of metrics for fish, macroinvertebrates and algae. In addition, individual metrics (typically as normalized, transformed values) were retained for analysis, as well.

Results of the preceding steps are presented in separate sections for each of the taxonomic groups. This is followed by analyses of relationships among metrics for the pairs of taxonomic groups (fish-macroinvertebrates, macroinvertebrates-algae, and macroinvertebrates-mussels).

6) Relationships between metrics (usually normalized, transformed values) and watershed or site characteristics (land use, watershed area, site habitat scores) are analyzed. Analogous relationships between PCs and watershed and site characteristics are analyzed.

7) Relationships between PCs of each taxonomic pair are analyzed, i.e., relationships between fish PCs and macroinvertebrate PCs, etc.

8) Technical issues arising in these analyses are presented in the results section. More general hypotheses for observed correlations and lack of correlations are presented in the discussion section.

9) The observed relationships among metrics are used to suggest potential mechanisms for integrated use of multiple taxonomic indicators. These are presented in the discussion section.

2. STUDY DESIGN AND METHODS

2.1 Data Acquisition and Standardization

Biological and habitat data from sites in and around the uplands physiographic regions (Piedmont, Ridge and Valley, Highlands) of New Jersey were obtained from several agencies including NJ DEP, US Geological Survey National Water-Quality Assessment Program (NAWQA), and the Nature Conservancy (TNC), as well as from various Academy of Natural Sciences (ANS) projects (EPA Riparian Study, PA Dam Removal Study). The source and number of samples for each biotic group are shown in Table 2.1.1. Most macroinvertebrate and algae data used were collected between 1996 and 2002. Fish data were collected between 1990 and 2003. To ensure that data were compatible among groups, all data were examined and normalized to the lowest common factor in each group. Specific procedures applied to each group are described in the sub-sections below. In cases where sampling methods did not follow standard techniques or where sampling techniques resulted in substantially different capture efficiencies, data were excluded from the analyses. It was assumed that all data had undergone in-house QA/QC procedures for identification and data entry purposes prior to being released.

Table 2.1.1. Species assignments used in calculating metrics. A 1 indicates that that species is included in calculation of that metric. A -1 indicates that the species is used in calculating opposite metrics (e.g., 1 for tolerance, and -1 for intolerance).

Scientific Name	Native	Tolerant EPA	Riffle	Pool	Omnivore Low	Generalist	Top Carnivore or Salmonid	Salmon Centrarchid	Benthic Insectivore	Insect Cyprinid
Bullhead species	0	-1	0	1	0	0	0	0	1	0
Ameiurus natalis	0	-1	0	0	0	1	0	0	1	0
Ameiurus nebulosus	1	-1	0	1	0	1	0	0	1	0
Ambloplites rupestris	0	0	0	1	0	0	1	1	0	0
Anguilla rostrata	1	0	0	0	0	0	0	0	0	0
Aphredoderus sayanus	1	0	0	0	0	0	0	0	0	0
Campostoma anomalum	1	-1	0	0	1	0	0	0	0	0
Carassius auritus	0	-1	0	1	1	1	0	0	0	0
Catostomus commersoni	1	-1	0	0	0	1	0	0	1	0
Carpiodes cyprinus	1	0	0	1	0	1	0	0	0	0
Clinostomus funduloides	1	1	0	0	0	0	0	0	0	1
Cottus bairdi	1	1	1	0	0	0	0	0	1	0
Cottus caerulomentorum	1	0	1	0	0	0	0	0	1	0
Cottus cognatus	1	1	1	0	0	0	0	0	1	0
Cyprinella analostana	1	1	0	0	0	0	0	0	0	1
Cyprinus carpio	0	-1	0	1	0	1	0	0	0	0
Cyprinella species	1	0	0	0	0	0	0	0	0	1
Cyprinella spiloptera	1	0	0	0	0	0	0	0	0	1
Dorosoma cepedianum	1	0	0	0	1	0	0	0	0	0
Enneacanthus gloriosus	1	0	0	0	0	0	0	1	0	0
Erimyzon oblongus	1	1	0	0	0	1	0	0	1	0
Esox americanus	1	0	0	0	0	0	1	0	0	0
Esox niger	1	0	0	1	0	0	1	0	0	0
Esox species	0	0	0	0	0	0	1	0	0	0
Etheostoma olmstedi	1	0	0	0	0	0	0	0	1	0
Exoglossum maxillingua	1	1	0	0	0	0	0	0	1	1
Fundulus diaphanus	1	-1	0	0	0	0	0	0	0	0
Fundulus heteroclitus	1	0	0	0	1	1	0	0	0	0
Gambusia affinis	1	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	-1	0	0	1	0	0	0	0	0
Hypentelium nigricans	1	1	1	0	0	1	0	0	1	0
Hybognathus regius	1	0	0	0	1	0	0	0	0	0
Ictalurus punctatus	0	0	0	1	0	0	1	0	0	0
Lampetra aepyptera	1	0	0	0	0	0	0	0	0	0

Table 2.1.1 (continued). Species assignments used in calculating metrics. A 1 indicates that that species is included in calculation of that metric. A -1 indicates that the species is used in calculating opposite metrics (e.g., 1 for tolerance, and -1 for intolerance).

Scientific Name	Native	Tolerant EPA	Riffle	Pool	Omnivore Low	Generalist	Top Carnivore or Salmonid	Salmon Centrarchid	Benthic Insectivore	Insect Cyprinid
Lampetra appendix	1	1	0	0	0	0	0	0	0	0
Lepomis auritus	1	0	0	1	0	1	0	1	0	0
Lepomis cyanellus	0	-1	0	0	0	1	0	1	0	0
Lepomis gibbosus	1	0	0	1	0	1	0	1	0	0
Lepomis hybrid	0	0	0	1	0	0	0	0	0	0
Luxilus cornutus	1	0	0	0	0	1	0	0	0	1
Misgurnus anguicaudatus	0	1	0	0	0	0	0	0	1	0
Micropterus dolomieu	0	0	0	1	0	0	1	1	0	0
Micropterus salmoides	0	0	0	1	0	0	1	1	0	0
Morone americana	1	0	0	0	0	0	1	0	0	0
Morone saxatilis	1	0	0	0	0	0	1	0	0	0
Notropis amoenus	1	-1	0	0	0	0	0	0	0	1
Notropis bifrenatus	1	0	0	0	0	0	0	0	1	1
Notropis buccatus	1	0	0	0	0	0	0	0	0	1
Notemigonus crysoleucas	1	-1	0	1	1	1	0	0	0	0
Noturus flavus	1	0	1	0	0	0	0	0	1	0
Noturus gyrinus	1	0	0	0	0	0	0	0	0	0
Notropis hudsonius	1	0	0	0	0	0	0	0	0	1
Noturus insignis	1	0	1	0	0	0	0	0	1	0
Nocomis micropogon	1	0	0	0	0	1	0	0	0	0
Notropis procne	1	0	0	0	0	0	0	0	0	1
Notropis rubellus	1	0	0	0	0	0	0	0	0	1
Noturus species	1	0	1	0	0	0	0	0	1	0
Oncorhynchus mykiss	0	0	0	0	0	0	1	1	0	0
Perca flavescens	1	0	0	1	0	0	0	0	0	0
Petromyzon marinus	1	0	0	0	0	0	0	0	0	0
Percina peltata	1	1	1	0	0	0	0	0	1	0
Pimephales notatus	1	-1	0	0	1	1	0	0	0	0
Pimephales promelas	0	-1	0	0	1	1	0	0	0	0
Pomoxis annularis	0	0	0	1	0	0	1	1	0	0
Pomoxis nigromaculatus	0	0	0	1	0	0	1	1	0	0
Rhinichthys atratulus	1	-1	0	0	0	1	0	0	1	1
Rhinichthys cataractae	1	1	1	0	0	0	0	0	1	1
Salvelinus fontinalis	1	0	0	0	0	0	1	1	0	0
Salmo trutta	0	0	0	0	0	0	1	1	0	0
Semotilus atromaculatus	1	-1	0	0	0	1	0	0	0	1
Semotilus corporalis	1	0	0	0	0	1	0	0	0	1
Stizostedion vitreum	0	0	0	1	0	0	1	0	0	0
Umbra pygmaea	1	-1	0	0	0	1	0	0	0	0

2.1.1 Fish

Data were compiled from several datasets. Data were chosen to include reach-level sampling in uplands (i.e., non Coastal Plain) of New York through northern MD (Fig. 2.1.1). Data sets included:

1) The NJ FIBI program. Data through 2003 were provided by Brian Margolis of NJ DEP. Data were provided as an ACCESS database containing a primary table of sampling dates, locations and habitat conditions and a table containing information on the total number of individuals of each species collected. Data on two additional sites, sampled in September 2002, but not included in the database, were provided by Tom Belton of NJ DEP. All sites are in northern NJ.

2) EPA- NJ bioassessment data. Data from a number of sites which were sampled and used in the development of the NJ FIBI (Kurtenback 1994) were provided by James Kurtenbach of US EPA. Data were provided in EXCEL spreadsheet format. All sites are in northern NJ.

3) NAWQA. Data from the Long Island-New Jersey (LINJ) NAWQA unit and the Delaware NAWQA unit were supplied by Jonathon Kennan and Karen Riva-Murray of USGS. Data were provided in EXCEL spreadsheet formats. Data from Long Island were not used. Remaining data were from NJ and PA.

4) Neversink data. Data from the Neversink River drainage of NY were provided by Colin Apse of TNC in EXCEL spreadsheet format.

5) Philadelphia Water Department (PWD). Lance Butler and Joseph Perillo of PWD provided data from streams in the Philadelphia area (in Philadelphia and in headwaters or tributaries of streams in Philadelphia). These sites included several suburban and highly urban sites.

6) Academy of Natural Sciences. ANS provided data from several studies, including:

    a) A study of sites in forested-agricultural watersheds, contrasting sites with forested and nonforested riparian zones (Sweeney, et al.2004). Most sites were in Chester County, PA. A few sites were in the Piedmont of northern MD. No macroinvertebrate data were available for this study, but these sites provided information on fish assemblages.

    b) A study of sites along an urban gradient, contrasting sites with forested and nonforested riparian zones (Hession, et al. 2000). Most sites were in southeastern PA; a few sites were in northern DE and northern MD.

c) A study of sites with varying amounts and distribution of forested and agricultural land in non-urban watersheds. Most sites were in southeastern PA; one site was in NJ.

d) A study of sites above and below dams. Most sites were in southeastern PA, but a few sites were in northern MD.

Data were converted into three primary ACCESS databases. The first contains raw macroinvertebrate count data, data on subsetted samples (see below) and calculated metrics. The second database contains fish data in original formats and in consistent formats. The third database contains output from the macroinvertebrate database (species count data in standard format, and calculated metrics), the fish database (consistent count information, etc.) and other data. A unique station identifier was given to each separate station in the various data sets. A table was created which links this station number (of the form ANSi) with station identifiers from the source program. Where the same station was sampled in more than one program or for more than one taxon, synonyms were retained in the stations table. For NJ fish, mussel, algae and odonate data, the AMNET station name for the station was identified, allowing linkage to the macroinvertebrate data. Where no AMNET sample was taken at a station, the nearest AMNET station on the same stream and within 15 stream km of the station was identified by GIS. This nearest station, distance to the station, and direction (upstream or downstream of the station) were entered as fields in the stations table. Primary station information (latitude, longitude, water body name and location) from the source data were also kept in this table. Additional station information (e.g., land use) was converted into additional tables, using the original station identifier, the project source, and the ANS station number as identifiers. In addition to the water body name in the source data, a second, consistent water body name was developed, i.e., using consistent spellings and abbreviations. Where there was inconsistency in a water body name (e.g.,a different spelling or the same stream identified as "brook" or "creek" in different entries), these were resolved using latitude/longitude information, and using names consistent with the gazetteer in the Topo! Software for USGS topographic maps. A unique sample number was given each separate sample of fish and of macroinvertebrates, which was linked to station and the source sample identifier. Other sample information (date, etc.) was kept with the sample identifier data in separate tables for each taxon. Primary catch information (sample identifier, taxon, and number of individuals recorded) were converted into consistent tables for each taxonomic group. Designations of basin and subbasin of stations were defined by the AMNET regional codes or by location of sample sites on topographic maps.

2.1.1.1 Joining

Data were converted from original formats into ACCESS tables. Tables included station identifier tables, sample identifier tables, and catch tables with consistent taxonomic codes, sample identifiers, and number of each species and of all fish caught in the sample. In a few cases, samples contained fish identified only to genus level, in addition to a number of fish of the same genus identified to species. In these cases (mostly *Lepomis* and *Cyprinella*), fish were assigned to species in proportion to their occurrence in the sample. Some specimens identified as *Cyprinella* hybrids were treated in the same way. A few specimens could not be assigned to a species. These were retained in the database at genus level. In calculating metrics, these were assigned characteristics of probable species in the genus (e.g., *Noturus* species was assigned characteristics of *N. insignis*). Identifications were retained from the original source, except in one case where a correction was confirmed with the originating source.

Data from single-pass, reach level sampling were used for all analyses. Where additional passes were taken, these data were retained in the database, but not used in analyses. The NJ Headwater study involved sampling adjacent 75-m stream reaches. For analyses, the two reaches were treated as a single sample, since the NJ FIBI samples were 150 in length.

Curves of number of species versus number of individuals collected were calculated. These indicated general similarity among data sources, supporting joint analysis.

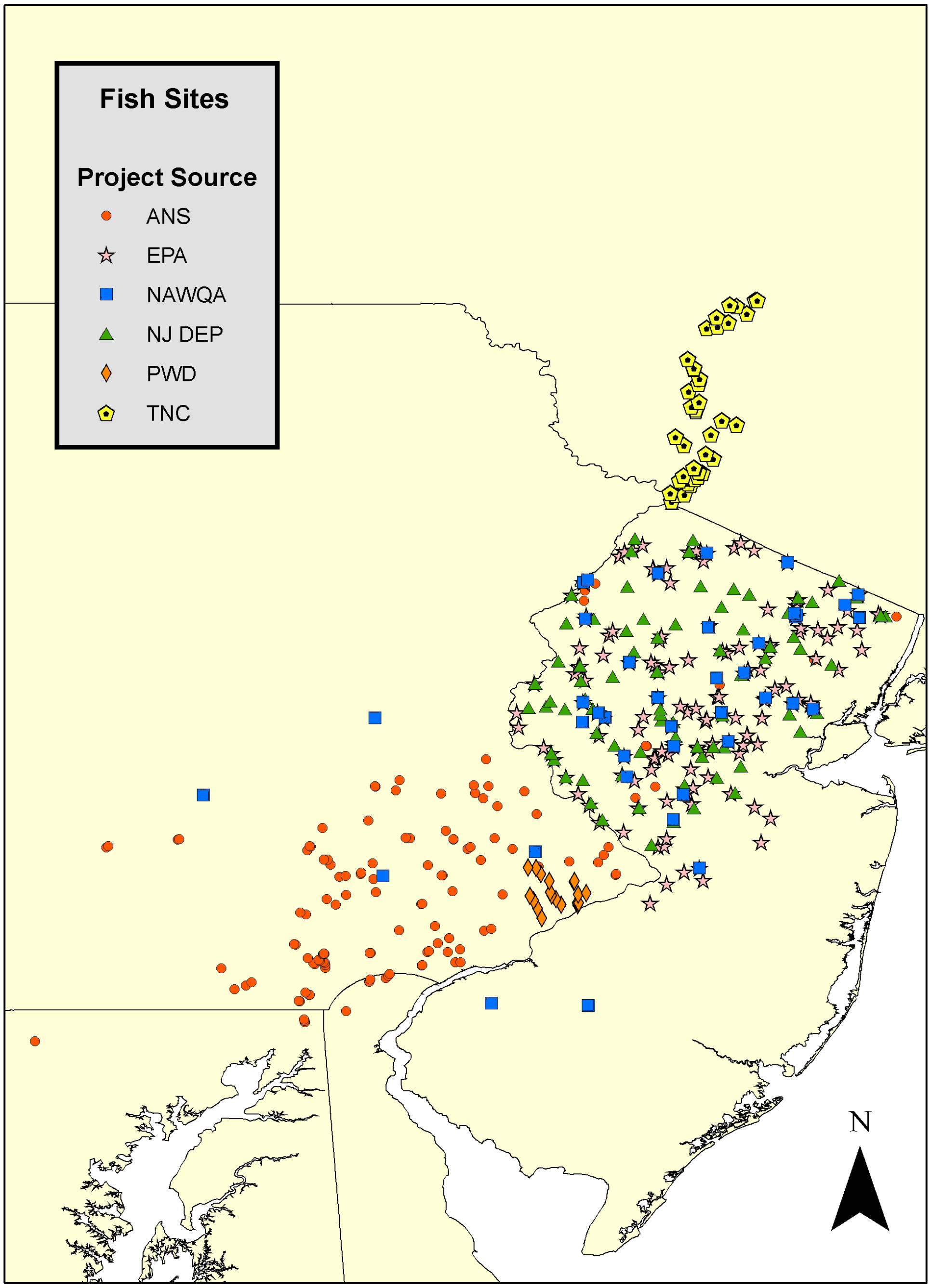


Figure 2.1.1. Location and source of sites with fish community data used in metric development and analyses.

2.1.2 Macroinvertebrates

2.1.2.1 Datasets

The majority of macroinvertebrate data was obtained from the NJ DEP's AMNET program and from ANS projects (Fig. 2.1.2). All data sources used similar methods to collect macroinvertebrate samples: a D-frame dip net and kick-sampling technique. However, mesh-size of the net differed among groups. Although smaller mesh size can result in a greater abundance of smaller taxa collected, it was not possible to correct for this source of error.

Macroinvertebrates were not identified to the same level of taxonomic resolution by all data sources. To create a taxonomically consistent dataset, it is necessary to resolve ambiguous taxa (not identified to the same taxonomic level). This can be achieved by "lumping" taxa to the highest taxonomic level used in identifications (i.e., deleting children taxa of ambiguous parent taxa and adding their abundances to the parent), or by excluding samples containing ambiguous taxa identifications. Both of these methods can be inappropriate for a quantitative analysis because they can result in a significant reduction in the number of taxa records leading to a very coarse analysis, or lead to a significant reduction in samples and a subsequent loss of power. Because all data sources contained ambiguous taxonomic identifications, we included all taxa records identified at and below the family level. In certain cases, where all data sources used the same ambiguous identification (e.g., Oligochaeta), records were included in the dataset.

2.1.2.2 Subsetting

Because the number if macroinvertebrates collected in a single sample can be prohibitive to identify, most samples are sub-sampled to a set number of organisms. For example, macroinvertebrate samples collected by TNC were sub-sampled to 300 individuals for identification, while ANS projects were typically subsampled to 100 or 200 organisms. Because the lowest sub-sampled size was 100 organisms, all macroinvertebrate samples exceeding a count of 100 organisms were electronically sub-sampled to 100 individuals using the sub-sampling package in the command-line program "R" (Ihaka and Gentleman 1996). This package produces simulated random sub-samples of species counts in a sample-by-species matrix and mimics the pre-identification sub-sampling procedure. Because "R" followed laboratory or in-field sub-sampling procedures in that a sub-sample was sorted in its entirety even after the target number was reached, some samples exceeded 100 organisms. Most samples contained between 100 and 115 organisms following subsetting procedures with 2 samples exceeding 150 organisms. Some samples (n=33) were not sub-sampled because they contained less than 100 organisms from the onset.

Regression analyses were conducted to identify the relationship between raw and sub-sampled data using the NJ DEP BF&BM metrics that comprise the AMNET Stream Bioassessment Protocol. These metrics include: Taxa Richness (total families), E+P+T Index (EPT, Total number of Ephemeroptera, Plecoptera, and Trichoptera families), % Dominant Family (%DF), Percent EPT taxa (%EPT), and Modified Family Biotic Index (FBI) (see Hilsenhoff 1988).

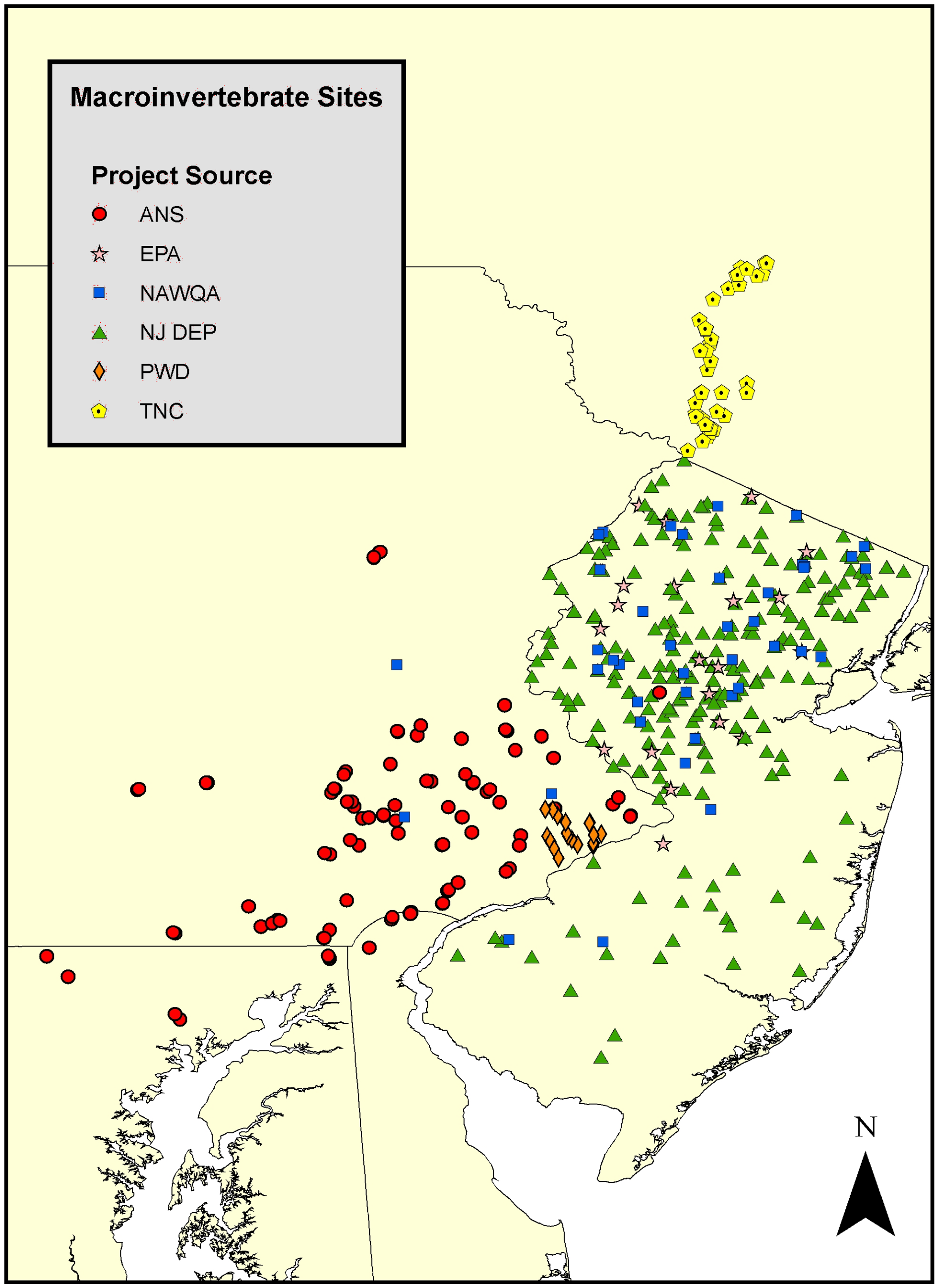
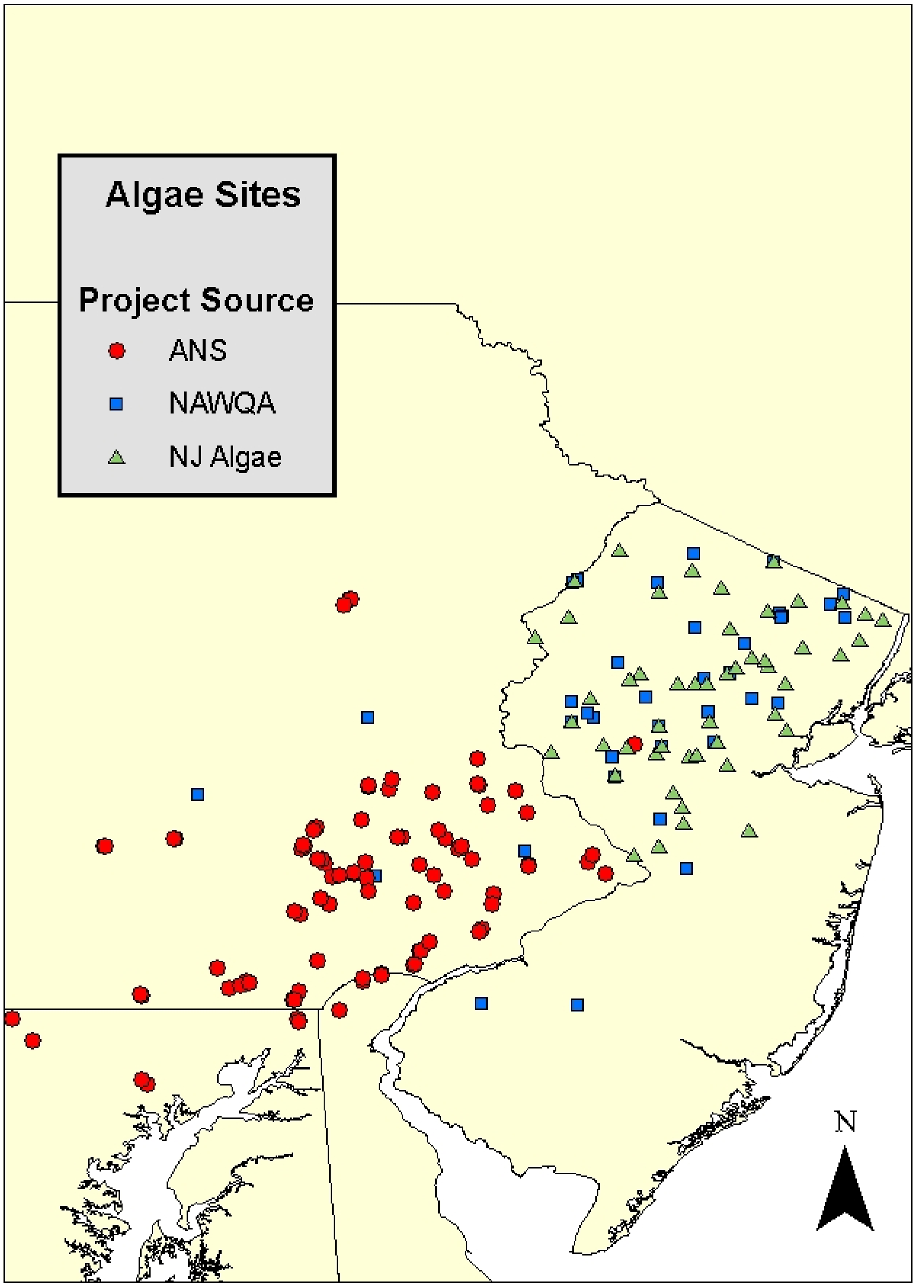


Figure 2.1.2. Location and source of sites with macroinvertebrate community data used in metric development and analyses. Sites located in the Coastal Plain region were used only in analyses of relationships with Rare and Endangered species (Odonates and Mussels).

2.1.3 Algae

Algal metrics were obtained from the Delaware and Long Island-New Jersey (LINJ) NAWQA program, and from Academy projects including: Algal Indicators of Eutrophication for New Jersey Streams, two riparian projects (EPA and GG2), and a dam effects project (GG2) (Fig. 2.1.3). Database issues prevented obtaining raw data for some datasets so algal data consist of existing metrics calculated by the ANS Phycology Section's diatom database. All metrics were normalized prior to analyses by subtracting the dataset standard deviation (std dev) of the metric from the metric value and dividing the result by the mean metric value for the dataset ([metric value]-[dataset std dev])/[dataset mean]).



2.1.3. Location and source of sites with algae metric data.

2.1.4 Rare and Endangered Species

2.1.4.1 Mussels

The New Jersey Division of Fish, Game and Wildlife (NJDFGW) surveyed mussels at 41 sites throughout New Jersey in 2000-2002 (Fig. 2.1.4). All sites were at or near AMNET sites. These did not generally correspond to NJ FIBI sites, and some mussel sites did not have the range of habitats requisite for NJ FIBI sampling. NJDFGW conducted fish surveys at sites without NJ FIBI samples, but these data were not available for inclusion in this report. Mussel data were summarized as total number of live individuals found at each station, and number of additional taxa for which shells were found. A few unidentified mussels were noted. These were from stations where other mussels were found; since they may represent species already recorded, they are not included in data summaries. Two mussels noted as possible paper pondshells (*Anodonta imbecilis*) were noted from Scotland Run. New Jersey is out of the native range of this species (e.g., Parmalee and Bogan 1998, as *Utterbackia imbecillis*), but the species has been introduced into New Jersey and is found in streams throughout Gloucester and Salem Counties (J. Bowers (pers. comm.). These records were not used in summaries of mussel distribution. Since the site is on the Coastal Plain, this does not affect comparisons of macroinvertebrate and mussel distribution. One site on the Paulins Kill was linked to AMNET site AN0025A in mussel database. Site AN0025A is listed as being on Blair Creek in the macroinvertebrate database. The mussel sample was taken on the Paulins Kill (J. Bowers, pers. comm.).

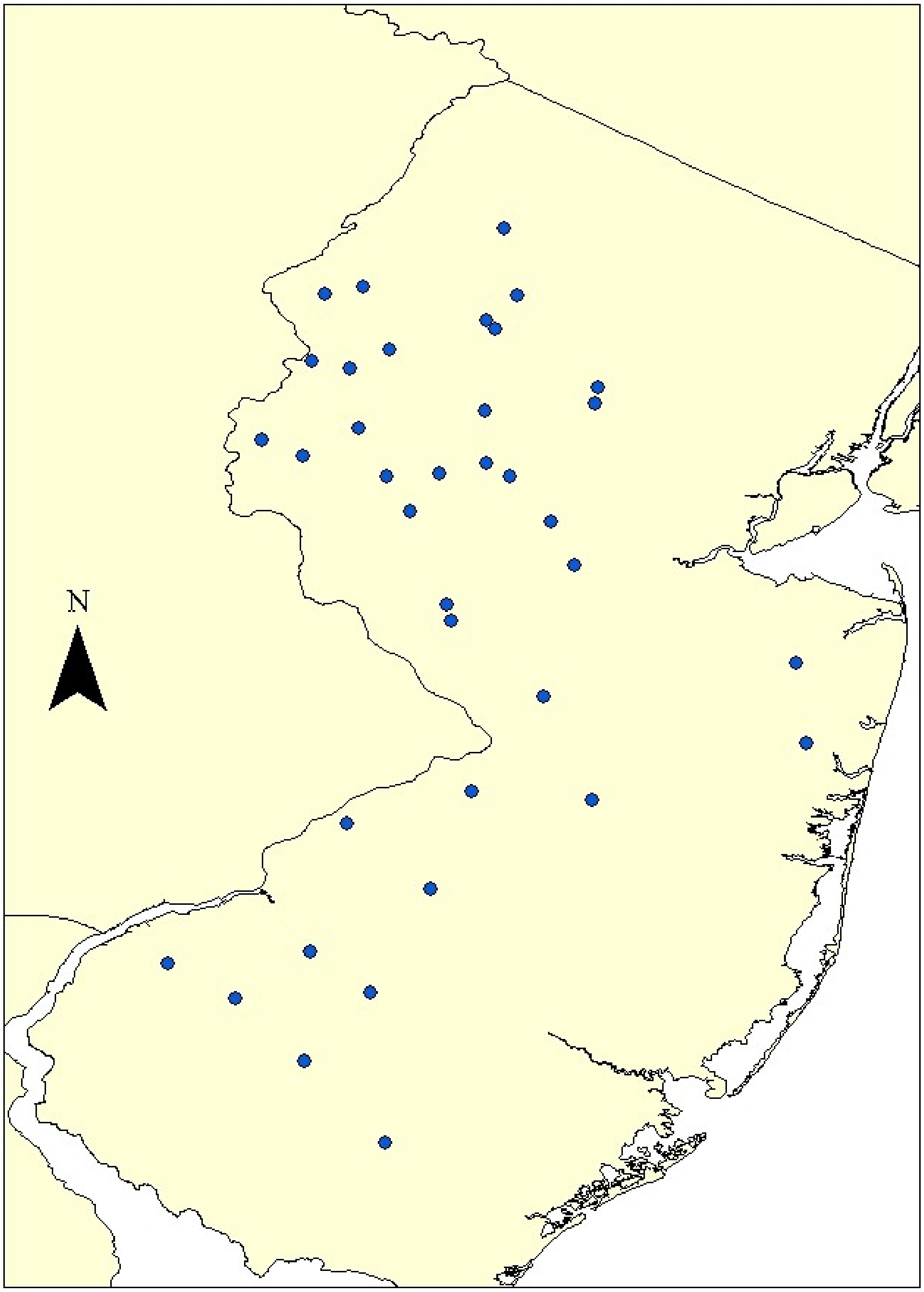


Figure 2.1.4. Location of mussel sites used in metric development and analyses.

2.1.4.2 Odonates

The NJDFGW surveyed odonates at or near AMNET sites (n=60) throughout New Jersey in 2000-2002 (Fig. 2.1.5). Species richness data (presence/absence) was available for adult, larvae, exuvia, ovipositioning, and mating odonates for 68 samples. Adult, ovipositioning, and mating odonates observed at the site were identified and recorded. In-stream and bank surveys for larvae and exuvia were conducted in all suitable habitats. Relative abundance data was not collected. Corresponding AMNET metric data PCs were related to odonate species richness measures using multiple regression. Multiple regression analyses were also used to relate macroinvertebrate metrics and land use variables to Odonate richness measures. Fish data were available for only 18 sites and, therefore, not examined.

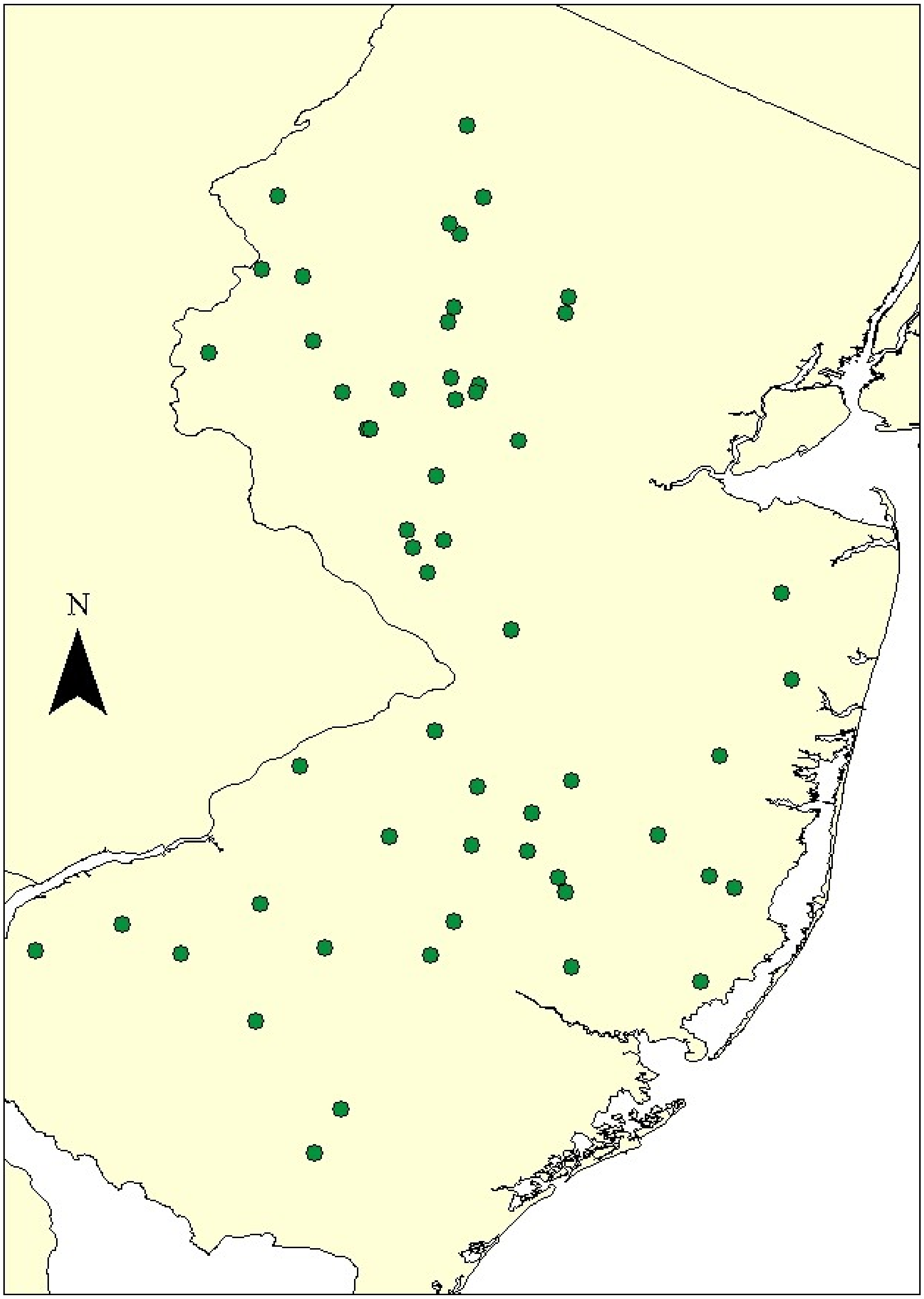


Figure 2.1.5. Location of odonate sites used in metric development and analyses.

2.2 Site and Watershed Characteristics

2.2.1 Habitat Data

The primary habitat data used in the analyses were based on the EPA's Rapid Bioassessment Protocol (RBP) for high gradient streams. The EPA RBP evaluates streams based on 10 measurements: epifaunal substrate/available cover, embeddedness, velocity/depth regimes, sediment deposition, channel flow status, channel alteration, frequency of riffles (or bends), bank stability, bank vegetative protection, and bank riparian vegetative zone width. Each of these measurements is given a score between 0 and 20 (with 20 being the optimum) and summed together to give a total habitat score for the site. The last four metrics (i.e., those relating to bank or riparian zone conditions) are given separate scores between 0 and 10 for each bank, and the sum used for the site metric score. One data source, EPA, did not provide scores for the individual parameters but only the total habitat score.

Because all data followed the same method, they were consistent across data sources with one exception. Scores from data collected by the Philadelphia Water Department were the average of the assessment of either two or three workers. The multiple assessments were conducted to ensure accuracy in the habitat scoring and we do not feel that it contributes any source of error to the data. The RBP habitat score was not assessed during field sampling for the ANS-urban gradient study sites. For these sites, habitat conditions were noted at points along a transect. RBP scores were developed for these sites based on the field transect notes, other field notes, site maps made at the time of sampling and photographs.

Other habitat and site data were available for some of the programs, including the NJ FIBI program. Some of the FIBI habitat data were used for separate analyses of the FIBI data.

2.2.2 Land Use Data

Land use data were collected from multiple sources. The majority of sites were used in NJ DEP (AMNET, FIBI) or NJ DEP-sponsored projects (EPA sites for FIBI development) so data were available from NJ DEP. Most land use data were generated from 1995 USGS land use and land coverages (LULC), but a subsample of sites (n=15, NJ Headwater sites) has land use data based on 2003 USGS LULC. Land use for remaining sites was calculated as part of the project. For example, most of the ANS sites calculated land use using land use coverages from Multi-Resolution Land Characteristics Consortium (MRLC) which compiles data from the USGS, EPA, National Oceanic and Atmospheric Administration (NOAA), U.S. Forest Service (USFS), National Atmospheric and Space Administration (NASA), and the Bureau of Land Management (BLM).

Land use is typically broken down into five main categories: agriculture, urban, forest, water, and wetlands. Much of the land use data obtained from NJ DEP combined water and wetlands into a single category (%Wet). As a result, %wet and %water were combined for remaining sites to prevent exclusion from analyses due to missing data.

2.2.3 Watershed Area

The drainage area of each sampling site (defined as the watershed area of the downstream boundary of the site) was calculated using GIS for most study sites. Watershed areas for most New Jersey sites (i.e., AMNET, FIBI, and NJ-EPA bioindicator development sites) were generated by NJ DEP. Data for the NAWQA sites were generated by and obtained from the USGS. Data for ANS sites were generated as part of individual studies by ANS.

2.3 Metric Development and Analyses

2.3.1 Fish

2.3.1.1 Raw Metrics

Metrics were calculated from data on numbers of individuals in each sample. No metrics relating to individual fish condition (e.g., number of fish with anomalies or parasites) were calculated. A series of candidate metrics was calculated, including metrics used in the NJ FIBI, and regional metrics (McCormick, et al. 2001, Daniels, et al. 2002), and a few additional metrics relating to habitat (e.g., % of species typically occurring in riffles and % of pool species) and trophic level (e.g., using variant definitions of omnivory). Many of these metrics are close analogues, frequently defined to measure similar aspects of the assemblage, but with different assignments of species to tolerance, habitat or trophic class. Correlation and principal components analyses indicated high intercorrelation among many metrics (Table 2.3.1). In particular, analogous metrics were usually highly correlated. Based on the observed correlations, a set of primary metrics was selected for further analysis (Table 2.3.2), which included single metrics within each type. All NJ FIBI metrics were retained for further study.

2.3.1.2 Transformation and Normalization of Metrics

In addition to analyses using the raw metrics, some metrics were transformed to provide better distributions of data. Proportional metrics were square-root transformed. The ratio of %chironomids to %EPT was ln(transformed).

Metrics were normalized to allow scale-independent comparisons. Raw metrics (number of taxa) or transformed metrics (square-root transformed proportions, etc.) were normalized to the mean and standard deviation of the primary set of samples. For fish, the primary sample set included the first pass of all standard reach-level samples, excluding two pairs of sites (in Muddy Run and Gunpowder River, both in the Susquehanna drainage), which were seen to have different faunal patterns than the remaining sites. The primary set includes sites which do not have matching macroinvertebrate samples, and thus includes a larger group of sites than used for the joint analyses of fish-macroinvertebrate relationships. For macroinvertebrates, the primary sample set was the group of AMNET sites selected to match fish sites and data from other programs taken at sites where fish samples were taken. For comparisons of macroinvertebrate and algal metrics, a different normalization was performed, using only the sites for which there were associated algal and macroinvertebrate samples.

2.3.1.3 Watershed-size Standardized Metrics

Empirical relationships between species richness measures and log(watershed area) were used to estimate predicted species richness for a given watershed area. The ratio of the observed measure to the predicted measure was used to standardize for these watershed area effects. The standardization was used for the four FIBI metrics (number of species, number of benthic invertivores, number of insectivorous cyprinids and number of salmonid and centrarchid species) for which these watershed area relationships are designed. These metrics were calculated for the four metrics for the NJ FIBI sites for analysis of macroinvertebrate-fish metric relationships.

The FIBI scores these metrics by watershed area using graphs of linear relationships between the metrics and log(watershed area).These graphs provide three lines for each metric, defining the range of metrics and breakpoints between scores. The middle line of each of these relationships was used to form the predicted species richness:

Nspecpred = 4.8 + 2.625 * log(watershed area in square miles)

Nbipred = 2.0 + 1.2 * log(watershed area in square miles)

Nsalcentpred = 1.0 + 2.05 * log(watershed area in square miles)

Nintolpred = 1.0 + 0.75 * log(watershed area in square miles), and

Rnspec = Nspec/Nspecpred

Rbi = Nbi/Nbipred

Rsalcent = Nsalcent/Nsalcentpred

Rintol = Nintol/Nintolpred.

Table 2.3.1. Pearson correlations among part of candidate metric set for 482 fish samples. Similar metrics with different species assignments are indicated by source.

		Nnat	NSpec	Nbi	Nbic	Nbi_w	Nlith	Ncytol	Nbe_m	Nsc	Nint_nj	Nint_e	Ntol_e	Ntol_p	Nres	Nwc	Nter
# of native species	Nnat	1.00	0.94	0.89	0.77	0.61	0.54	0.82	0.74	0.54	0.31	0.64	0.55	0.75	0.99	0.73	0.68
Total # of species	NSpec		1.00	0.82	0.69	0.57	0.46	0.77	0.72	0.76	0.33	0.57	0.62	0.80	0.94	0.79	0.64
# of benthic invertivores (nj)	Nbi			1.00	0.93	0.65	0.61	0.88	0.63	0.38	0.34	0.68	0.52	0.69	0.90	0.57	0.79
# of benthic invertivorous cyprinids	Nbic				1.00	0.50	0.59	0.90	0.43	0.25	0.19	0.62	0.48	0.60	0.78	0.50	0.86
# of benthic invertivores (nawqa)	Nbi_w					1.00	0.54	0.59	0.74	0.27	0.72	0.84	0.04	0.26	0.61	0.14	0.41
# of lithophils	Nlith						1.00	0.52	0.46	0.07	0.42	0.58	0.22	0.29	0.55	0.13	0.60
# of cyprinids, excluding tolerant spp.	Ncytol							1.00	0.51	0.36	0.28	0.70	0.41	0.58	0.83	0.58	0.84
# benthic species (emap)	Nbe_m								1.00	0.41	0.47	0.68	0.33	0.48	0.74	0.40	0.33
# of salmonids & centrarchids	Nsc									1.00	0.27	0.21	0.39	0.53	0.54	0.70	0.25
# of Intolerant species (nj)	Nint_nj										1.00	0.62	-0.16	-0.05	0.31	-0.13	0.19
# of Intolerant species (epa)	Nint_e											1.00	0.08	0.25	0.64	0.23	0.51
# of tolerant species (epa)	Ntol_e												1.00	0.89	0.57	0.66	0.52
# of tolerant species (emap	Ntol_p													1.00	0.75	0.79	0.57
# of resident species	Nres														1.00	0.73	0.71
# of water column species	Nwc															1.00	0.44
# of terete cyprinids	Nter																1.00

Table 2.3.1. (Cont.) Pearson correlations among part of candidate metric set for 482 fish samples. Similar metrics with different species assignments are indicated by source.

		Nins	Ngen_w	Pint	Pcacom	Pgen_nj	Pgen_w	Pic	Pbi	Pbi_w	Pbic	Pter	Pwc	Ptcs	Ptol_e	Priff	Ppool	Pomn
# of native species	Nnat	0.85	0.75	-0.19	0.29	-0.24	-0.09	0.06	-0.02	0.31	0.02	0.12	0.01	-0.32	-0.23	0.24	0.00	-0.04
Total # of species	NSpec	0.81	0.82	-0.20	0.31	-0.21	-0.09	-0.04	-0.14	0.27	-0.08	0.10	0.07	-0.26	-0.26	0.20	0.13	-0.04
# of benthic invertivores (nj)	Nbi	0.89	0.63	-0.15	0.26	-0.19	-0.06	0.27	0.23	0.34	0.23	0.21	-0.13	-0.34	-0.08	0.30	-0.24	-0.09
# of benthic invertivorous cyprinids	Nbic	0.81	0.56	-0.16	0.22	-0.16	0.02	0.40	0.32	0.22	0.37	0.30	-0.15	-0.34	0.01	0.21	-0.29	-0.08
# of benthic invertivores (nawqa)	Nbi_w	0.70	0.24	0.09	0.12	-0.34	-0.11	0.22	0.06	0.60	0.13	-0.01	-0.28	-0.18	-0.15	0.54	-0.17	-0.11
# of lithophils	Nlith	0.45	0.34	0.01	0.07	-0.12	0.11	0.38	0.27	0.25	0.34	0.22	-0.19	-0.24	0.07	0.23	-0.26	-0.07
# of cyprinids, excluding tolerant spp.	Ncytol	0.87	0.56	-0.09	0.22	-0.22	-0.11	0.23	0.13	0.24	0.18	0.26	-0.06	-0.25	-0.16	0.22	-0.15	0.00
# benthic species (emap)	Nbe_m	0.66	0.52	-0.07	0.20	-0.32	-0.11	-0.05	-0.13	0.48	-0.10	-0.07	-0.09	-0.18	-0.27	0.38	0.07	-0.05
# of salmonids & centrarchids	Nsc	0.42	0.65	-0.09	0.22	-0.15	-0.13	-0.27	-0.36	0.15	-0.30	-0.01	0.14	0.00	-0.34	0.09	0.40	-0.10
# of Intolerant species (nj)	Nint_nj	0.35	-0.02	0.37	-0.01	-0.33	-0.11	0.23	0.06	0.37	0.14	-0.02	-0.34	0.12	-0.12	0.44	-0.20	-0.12
# of Intolerant species (epa)	Nint_e	0.74	0.27	0.07	0.10	-0.30	-0.11	0.29	0.13	0.46	0.20	0.07	-0.21	-0.19	-0.13	0.41	-0.21	-0.10
# of tolerant species (epa)	Ntol_e	0.40	0.82	-0.39	0.29	0.22	0.15	-0.04	0.01	-0.04	-0.02	0.10	0.26	-0.37	0.17	-0.09	0.05	0.08
# of tolerant species (emap	Ntol_p	0.57	0.90	-0.40	0.33	0.03	0.10	-0.03	-0.07	0.09	-0.04	0.06	0.21	-0.41	0.00	0.02	0.09	0.04
# of resident species	Nres	0.85	0.77	-0.19	0.28	-0.21	-0.07	0.08	0.01	0.31	0.04	0.15	0.02	-0.32	-0.21	0.23	-0.01	-0.03
# of water column species	Nwc	0.59	0.84	-0.32	0.26	0.04	-0.06	-0.23	-0.23	0.01	-0.23	0.08	0.39	-0.26	-0.23	-0.07	0.30	0.05
# of terete cyprinids	Nter	0.71	0.57	-0.14	0.22	-0.06	0.03	0.30	0.25	0.15	0.26	0.39	-0.07	-0.30	0.03	0.06	-0.23	0.03
# of insectivores	Nins	1.00	0.49	-0.11	0.22	-0.22	-0.23	0.11	0.07	0.37	0.06	0.15	-0.05	-0.27	-0.22	0.29	-0.11	-0.05
# of generalists (nawqa)	Ngen_w		1.00	-0.36	0.33	0.01	0.15	-0.06	-0.13	0.10	-0.07	0.11	0.25	-0.36	-0.07	0.00	0.20	0.03
Prop. Intolerant spp.	Pint			1.00	-0.19	-0.25	-0.41	-0.09	0.05	0.06	-0.15	-0.04	-0.31	0.81	-0.34	0.13	-0.19	-0.09
Prop. White sucker	Pcacom				1.00	-0.18	0.19	-0.18	0.11	0.08	-0.17	-0.11	-0.13	-0.17	0.13	0.04	-0.05	-0.02
Prop. generalists (NJ)	Pgen_nj					1.00	0.05	-0.10	0.13	-0.32	-0.06	0.15	0.53	-0.21	0.53	-0.25	-0.07	0.07
Prop. Generalists (nawqa)	Pgen_w						1.00	0.42	0.11	-0.28	0.45	0.01	-0.01	-0.43	0.61	-0.17	0.06	0.07
Prop. Insectivorous cyprinids	Pic							1.00	0.64	0.01	0.98	0.46	-0.45	-0.34	0.44	0.20	-0.54	-0.19
Prop. Benthic invertivores	Pbi								1.00	0.10	0.67	0.25	-0.38	-0.17	0.51	0.18	-0.76	-0.22
Prop. Benthic invertivores (nawqa)	Pbi_w									1.00	-0.07	-0.11	-0.33	-0.16	-0.29	0.64	-0.15	-0.13
Prop. Benthic invertivorous cyprinids	Pbic										1.00	0.43	-0.42	-0.34	0.47	0.18	-0.53	-0.18
Prop. Terete cyprinids	Pter											1.00	-0.19	-0.15	0.01	-0.16	-0.19	0.00
Prop. Water coloumn spp.	Pwc												1.00	-0.20	-0.06	-0.30	0.48	0.30
Prop. Top carnivores/salmonids	Ptcs													1.00	-0.40	-0.09	0.11	-0.05
Prop. Tolerant spp. (epa)	Ptol_e														1.00	-0.13	-0.39	-0.02
Prop. Riffle species	Priff															1.00	-0.19	-0.10
Prop. Pool species	Ppool																1.00	0.03
Prop. Omnivores	Pomn																	1.00

Table 2.3.2. Names and abbreviations of primary fish metrics used in analyses of relationships with macroinvertebrate metrics, watershed area and land use, and site characteristics. NJ FIBI indicates whether the metric is part of the NJ FIBI.

Metric	Abbr.	NJ FIBI	Notes
Number of species	Nspec	Yes	
Number of benthic invertivore species	Nbi	Yes	
Number of salmonid and centrarchid species	Nsalcent	Yes	
Number of intolerant species	Nintol	Yes	
%White sucker	%cacom	Yes	
%Generalists	%gen	Yes	
%Insectivorous cyprinids	%inscyp	Yes	
%Top Carnivores/salmonids	%topcarn	Yes	
Number of individuals	Nind	Yes	
% Intolerant species	%intol	No	
%Tolerant species (EPA RBP definition)	%Tol	No	
%Riffle habitat specialist species	%Riffle	No	
%Pool habitat specialist species	%Pool	No	
%Omnivorous species	%Omnlow	No	Omnivory includes detritus, plant and invertebrate foods
R Number of species	Rnspec	No	Nspec/(Nspec predicted by watershed area)
R Number of benthic invertivore species	Rbi	No	Nbi/(Nbi predicted by watershed area)
R Number of salmonid and centrarchid species	Rsalcent	No	Nsalcent/(Nsalcent predicted by watershed area)
R Number of intolerant species	Rintol	No	Nintol/(Nintol predicted by watershed area)
Square root transformed metric X	Norm%X	No	Square root transformation of raw metric for proportion metrics
Ln transformed metric Y	LnY	No	Ln(Y+1) transformation
Normalized value of metric X	NormX	No	Normalized to mean=0 and standard deviation = 1

2.3.2 Macroinvertebrates

Metrics at both the family and genus level were developed for all invertebrate data using sub-sampled data. All metrics, definitions, and predicted response to environmental stress are shown in Table 2.3.3. Metrics were developed based on their relationships to water and/or habitat quality and community health or structure. Pearson correlation analyses were used to determine covariance among metrics. When two metrics were highly correlated (correlation coefficients exceeding ±0.70), one of the metrics was excluded from analyses. Decisions on which metric to exclude were based on correlation results among other metrics, biological importance of the metric, and ease with which metric information is obtained (e.g., Taxa Richness is often positively correlated with Shannon-Wiener Diversity Index but is a more parsimonious calculation).

Table 2.3.3. Macroinvertebrate metrics, taxonomic resolution, and predicted response to stress developed for testing for the integrated analyses dataset.

Metric	Description	Taxonomic Resolution		Response to Impairment
		Family	Genus	
Taxa Richness	Total number of individual taxa	X	X	Decrease
# Dipteran Taxa	Total number of Dipteran taxa	X	X	Increase
# Ephemeropteran Taxa	Total number of Ephemeropteran taxa	X	X	Decrease
# Plecopteran Taxa	Total number of Plecopteran taxa	X	X	Decrease
# Trichopteran Taxa	Total number of Trichopteran taxa	X	X	Decrease
# non-insect	Total number of non-insect taxa	X		Increase
# EPT Taxa	Total number of EPT taxa	NA	NA	Decrease
% Baetidae	Percent composition of Baetidae	X		Decrease
% Chironomidae	Percent composition of Chironomidae	X		Increase
% Diptera	Percent composition of Diptera	NA	NA	Increase
% Dominant Taxa	Percent composition of the most dominant taxa	X	X	Increase
% Hydropsychidae	Percent composition of Hydropsychidae	X		Decrease
% Intolerant Taxa	Percent composition of Intolerant taxa (tolerance values 0-2)	X		Decrease
% Tolerant Taxa	Percent composition of Tolerant taxa (tolerance values 8-10)	X		Increase
% non-insect Taxa	Percent composition of non-insect taxa	NA	NA	Increase
% EPT Taxa	Percent composition of Ephemeropteran, Plecopteran, and Trichopteran taxa	NA	NA	Decrease
Family Biotic Index (FBI)	Weighted mean of sample pollution tolerance values (0-10) based on Hilsenhoff (1987)	X		Increase
Chironomidae:EPT	Ratio of chironomidae individuals to EPT individuals	X		Increase
Simpsons Diversity Index	General measure of diversity		X	Decrease
Simpsons Evenness	Measures the evenness, or equitability, of the community		X	Decrease
% Collector-Filterers	Percent composition of organisms classified as filterers of minute particles from the water column (e.g. net-building caddisflies)	NA	NA	Variable
% Scrapers	Percent composition of organisms classified as grazers of algae	NA	NA	Decrease
% Collector-Gatherers	Percent composition of organisms which feed on fine particles of decomposing organic material	NA	NA	Variable
% Shredders	Percent composition of organisms which feed on dead plant material (leaves, algae, grasses, and rooted aquatic plants)	NA	NA	Decrease
% Predators	Percent composition of macroinvertebrates which feed on other insects	NA	NA	Variable

2.3.3 Algae

Seven algae diversity and autecological metrics were examined:

1) *Diatom Species Richness* - number of diatom species in a count. This metric was calculated for each sample where diatoms were counted. High species richness indicates absence of severe environmental stress.
2) *% Dominant Taxon*– Percent of total diatom valves made up by the most abundant taxon. Samples where the percent abundance of a dominant taxon is low contain a higher diversity.

3) *Percent Dominants*– Percent of total diatom valves made up of taxa that occurred in >10%abundance.

4) *Shannon-Wiener Diatom Diversity Index*– The SWDI for diatoms was calculated on the basis of relative abundance of diatom valves in a count, measures diversity of the diatom assemblage. The formula used for calculations is:

$$H' = -\sum_{I-1}^{S} (p_i)(\mathrm{Log}_2 p_i)$$

   Where H'= Shannon-Wiener index of species diversity, s= number of diatom species, p_i=proportion of species I in the total diatom count.

5) *Siltation Index*– This index is a percentage relative abundance of motile diatoms *Amphiprora, Aneumastus, Cavinula, Craticula, Cylindrotheca, Diadesmis, Entomoneis, Fallacia, Gyrosigma, Hantzshia, Kobayasiella, Luticola, Navicula, Nitzschia, Placoneis, Plagiotropis, Pleurosigma, Proshkinia, Sellaphora, Stenopterobia, Surirella,* and *Tryblionella* in a diatom count. These diatoms are able to move through silt particles and are associated with fine sediments. This index has been used to detect siltation in Montana rivers (Bahls et al. 1992) and may be related to bank erosion and agricultural practices.

6) *Percent of* Achnanthidium minutissimum– This metric is a relative abundance of valves of *Achnanthidium minutissimum,* a small monoraphid diatom. It is one of the most common diatoms in fresh waters, known by its wide ecological amplitude and ability to tolerate stress. *Achnanthidium minutissimum* is known as an early-successional species (Peterson and Stevenson 1992) and often is the first to colonize river beds after scouring during spates. Additionally, *A. minutissimum* was often the dominant species in Montana streams receiving mining discharge and other chemicals (Barbour et al. 1999).

7) *Centrales/Pennales ratio*– Ratio of centric to pennate diatoms approximately shows the proportion of planktonic taxa in the diatom community. This ratio tends to be higher in large lowland rivers where the plankton community is well developed or in

streams that drain lakes, ponds, or reservoirs. A high ratio of Centrales/Pennales in small rivers may indicate increased nutrient loading.

Pearson correlation analyses were used to examine relationships among metrics. All datasets had diatom sample counts of 600 valves with the exception of the Academy's EPA riparian project which had a sample count of 300. Because raw data were not available, it was not possible to subsample datasets for sample size consistency. To determine if metric relationships differed due to sample counts, sites were categorized based on sample count and separate Pearson correlation analyses of normalized data were conducted. When two metrics were highly correlated (correlation coefficients exceeding ±0.70), one of the metrics was excluded from analyses. Decisions on which metric to exclude were based on correlation results among other metrics, biological importance of the metric, and ease with which metric information is obtained.

The relationship among environmental variables (land use (square root proportion), area (ln), and EPA habitat scores) for algae samples was examined using Pearson correlation analyses. As with metric analyses, correlation coefficients between two environmental variables exceeding ±0.70 resulted in the elimination of one of the variables.

Principal components analysis (PCA) was used to identify patterns among algae metrics. Algae-macroinvertebrate metric relationships were examined using regression analyses of algae principal components (PC) with macroinvertebrate PCs. Similar regressions related algae PCs with habitat, land use, and drainage area. To further discern patterns in algae metrics and associations with environmental variables, PCA analyses, correlation and metric-environmental variable regressions were conducted based on samples in individual datasets (e.g., NJ Algal Indicators) and on drainage sub-basins (e.g., Raritan River sub-basin).

3. RESULTS

3.1 Data Summary

A summary of the number of sites and data sources is shown in Table 3.1.1. There were substantially more sites with fish and macroinvertebrate data than other taxa groups. Almost 60% of the fish sites were derived from ANS projects and sites used in the development of the NJ FIBI. The remaining 40% of sites came from NJ DEP's FIBI program, and monitoring conducted by the Philadelphia Water Department and the Nature Conservancy. There were slightly fewer sites with macroinvertebrate data. Over half of the sites were from the NJ DEP's AMNET program, although some of the sites (n=16) from the Coastal Plain region were not included in the analyses. Of the available macroinvertebrate and fish sites, 312 were identical or within 15 km from one another and used to determine relationships between taxa groups and metrics. There were 202 algae sites with the majority of sites (159) coming from ANS projects. Included in this number are the 51 sites from the project examining Eutrophication of NJ Streams. Over 96% of these sites had corresponding macroinvertebrate data which were used to examine relationships between the two groups. Mussel and odonate data were available at 40 and 61 sites, respectively.

Sites included a range of watershed sizes and land uses (Fig. 3.1.1). Several of the programs, including the ANS riparian study and the NJ DEP-ANS headwater program, sampled small streams (less than the 5 square mile threshold for the NJ FIBI). Several of the studies, including the EPA IBI development study, the NAWQA studies, the NJ DEP-ANS headwater study and the ANS urban gradient study, selected stations to span a range of sites from undeveloped to heavily urban watersheds. The ANS riparian study and the TNC Neversink study primarily sampled undeveloped sites. In contrast, most of the PWD stations were in highly urban watersheds in Philadelphia.

Several land uses (%urban, %forest and %agriculture) were significantly correlated with ln(watershed area) (Figs. 3.1.1-3.1.4), although correlations were low: %urban (negative correlation, $r^2 = 0.01$, $p<0.001$), %forest (positive correlation, $r^2= 0.01$, $p<0.001$), %agriculture (negative correlation, $r^2= 0.06$, $p<0.01$). Land use types were negatively correlated, as would be expected (Figs. 3.1.5 and 3.1.6), e.g., across all sites, $r^2 = 0.43$ for %Urban and %Forest, and $r^2 = 0.13$ for %urban and %agriculture.

Table 3.1.1. Number and source of sites with biological community data.

	Taxonomic Group				
Project Source	**Fish**	**Macro**	**Algae**	**Mussels**	**Odonates**
ANS	124	98	159	0	0
EPA	156	0	0	0	0
NAWQA	40	47	43	0	0
NJ DEP	86	252	0	40	61
PWD	22	19	0	0	0
TNC	41	45	0	0	0
Total	**469**	**461**	**202**	**40**	**61**

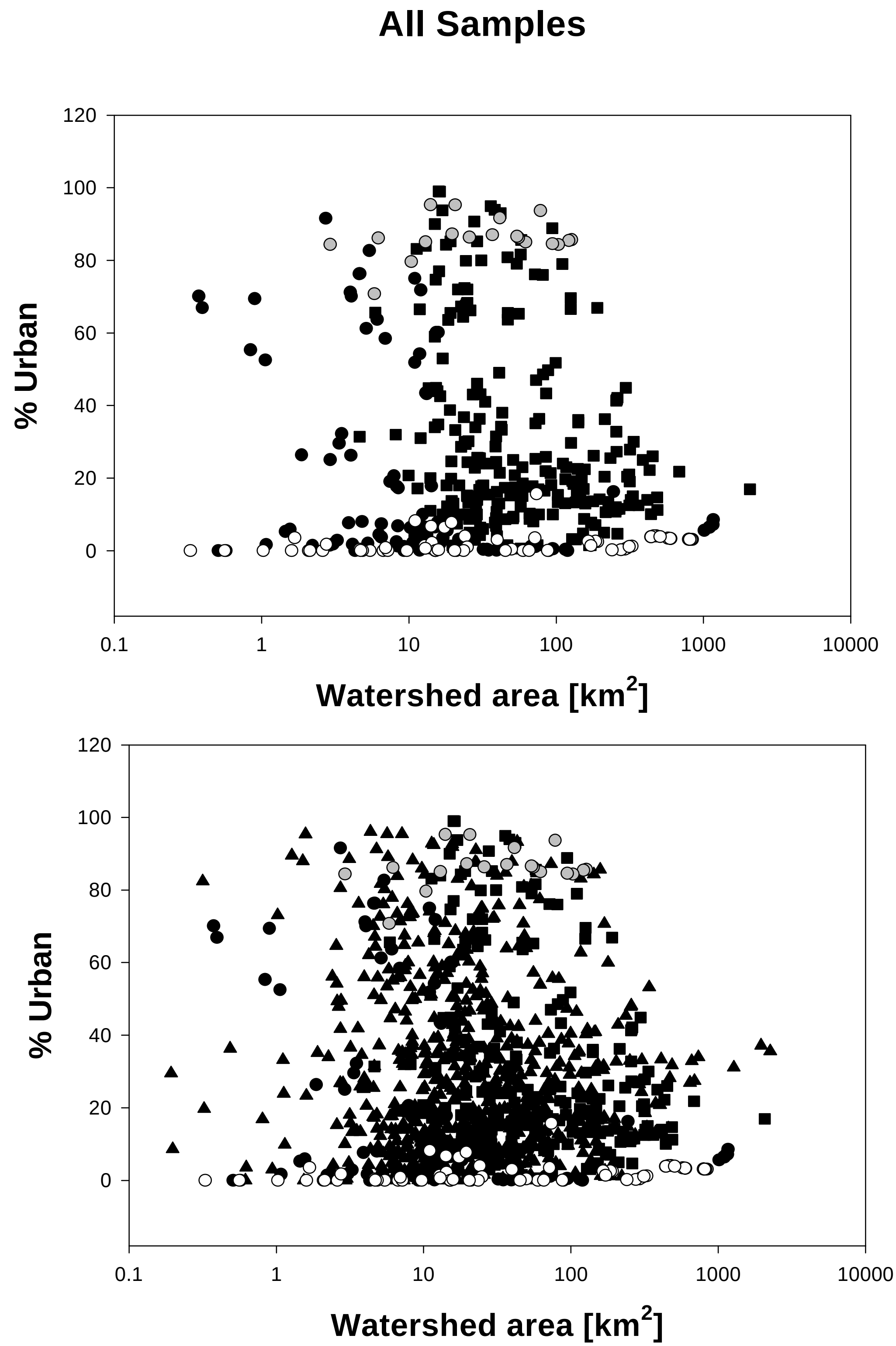


Figure 3.1.1. Relationship between %urban and watershed area for fish and algae samples (top) and all samples (bottom). Sites are coded by source: AMNET (closed triangles), ANS (closed circles), EPA, NJ FIBI, and NAWQA (closed squares), PWD (gray circles), and TNC (open circles).

All Samples

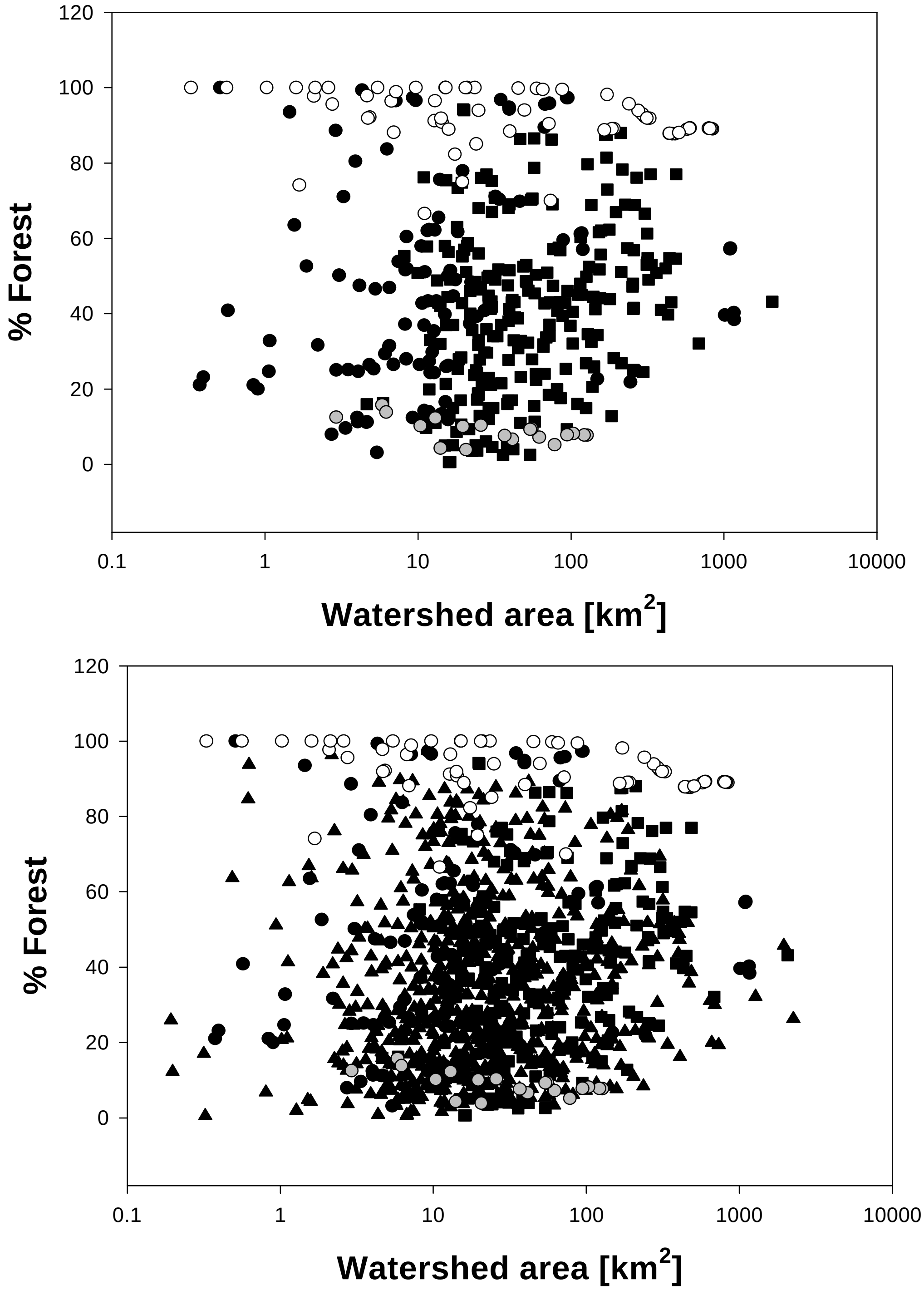
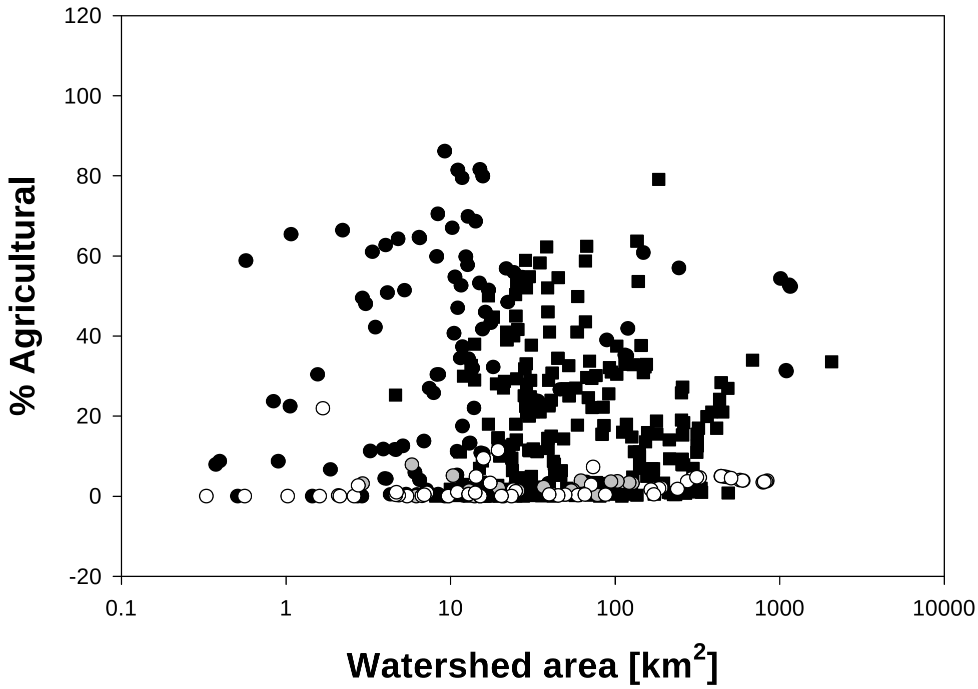


Figure 3.1.2. Relationship between %forest and watershed area for fish and algae samples (top) and all samples (bottom). Sites are coded by source: AMNET (closed triangles), ANS (closed circles), EPA, NJ FIBI, and NAWQA (closed squares), PWD (gray circles), and TNC (open circles).

All Samples



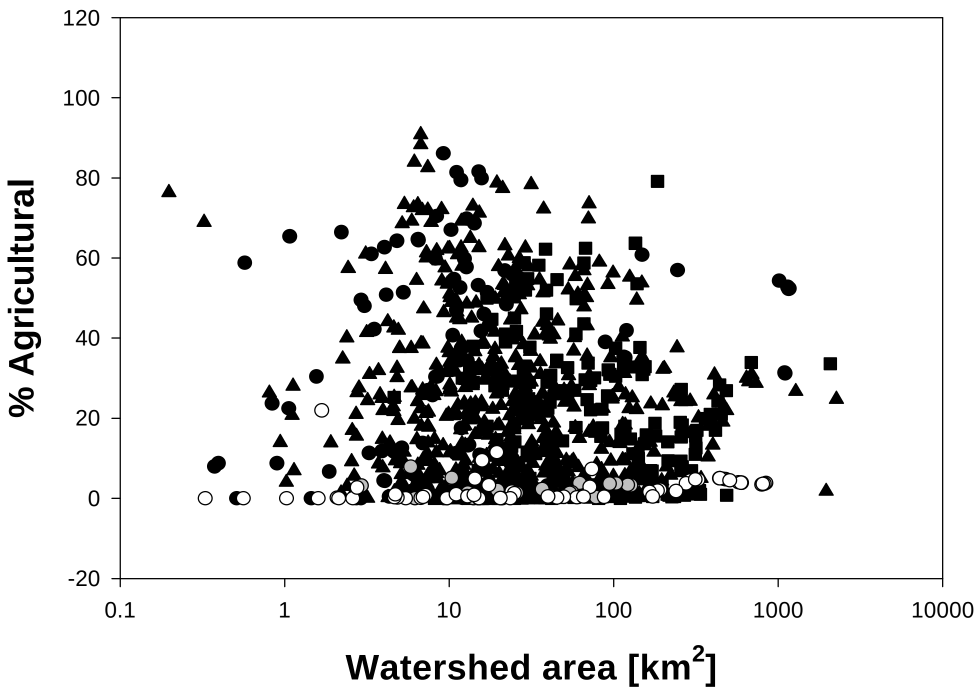


Figure 3.1.3. Relationship between %agriculture and watershed area for fish and algae samples (top) and all samples (bottom). Sites are coded by source: AMNET (closed triangles), ANS (closed circles), EPA, NJ FIBI, and NAWQA (closed squares), PWD (gray circles), and TNC (open circles).

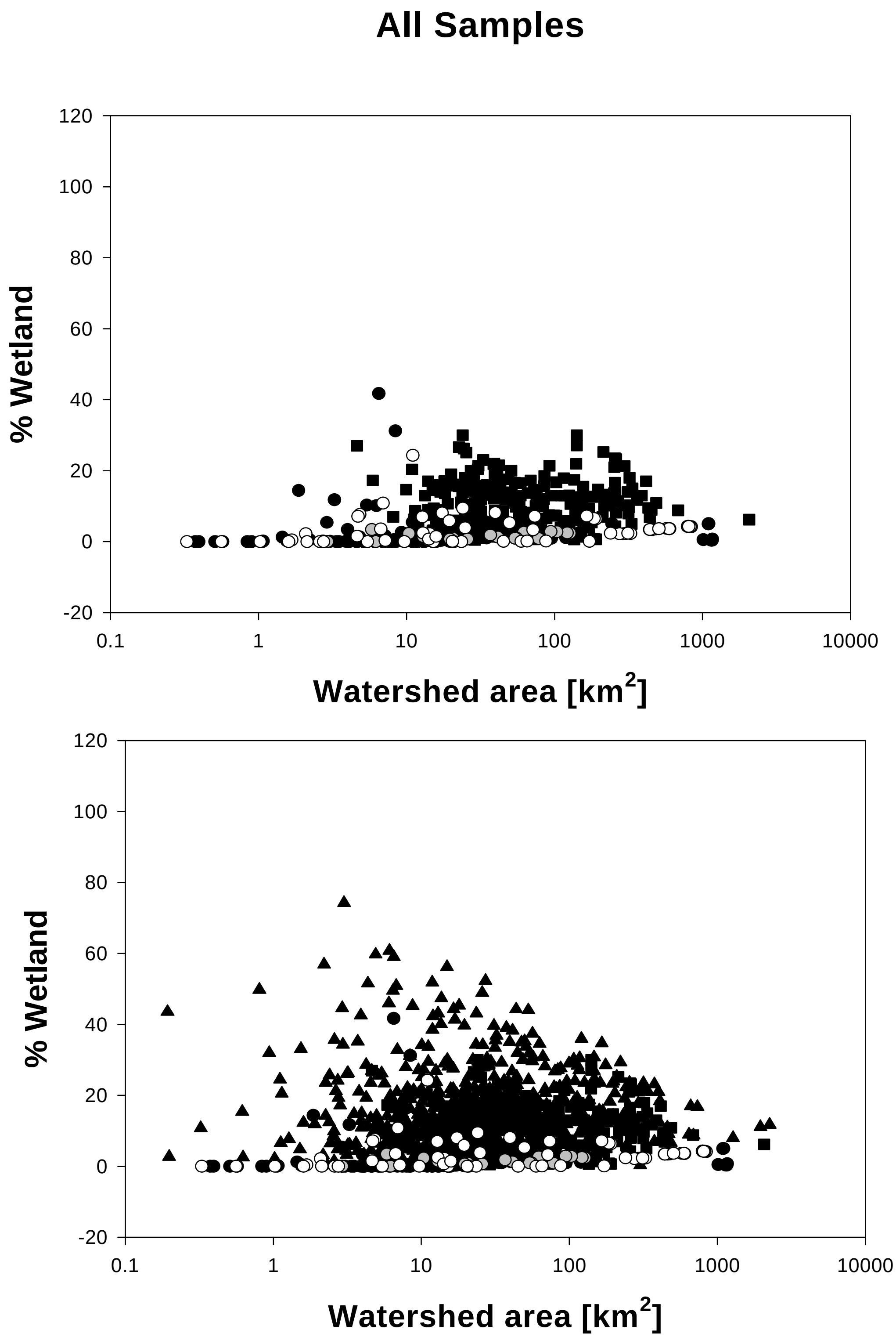


Figure 3.1.4. Relationship between %wetland and watershed area for fish and algae samples (top) and all samples (bottom). Sites are coded by source: AMNET (closed triangles), ANS (closed circles), EPA, NJ FIBI, and NAWQA (closed squares), PWD (gray circles), and TNC (open circles).

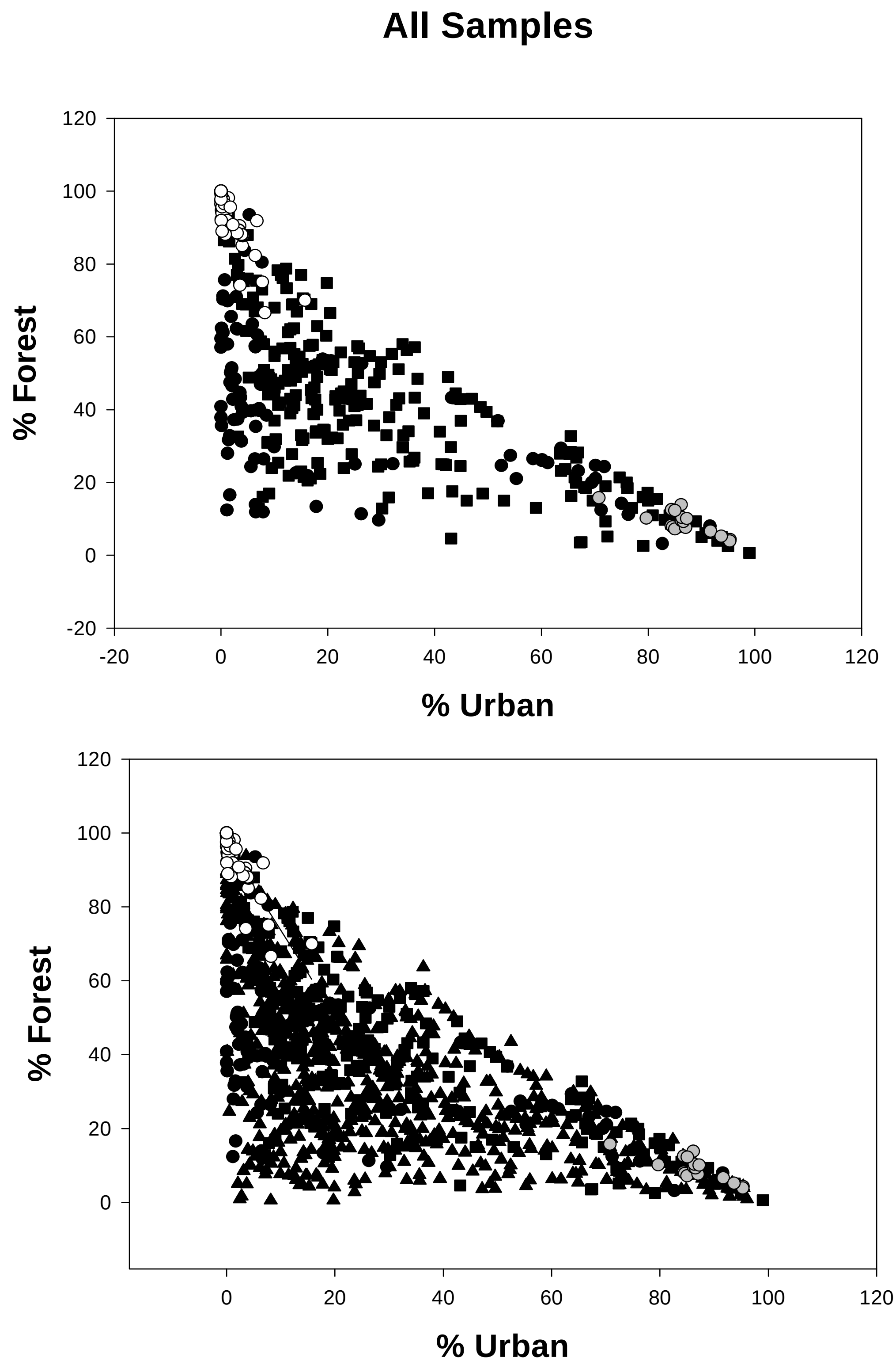


Figure 3.1.5. Relationship between %urban and %forest for fish and algae samples (top) and all samples (bottom). Sites are coded by source: AMNET (closed triangles), ANS (closed circles), EPA, NJ FIBI, and NAWQA (closed squares), PWD (gray circles), and TNC (open circles).

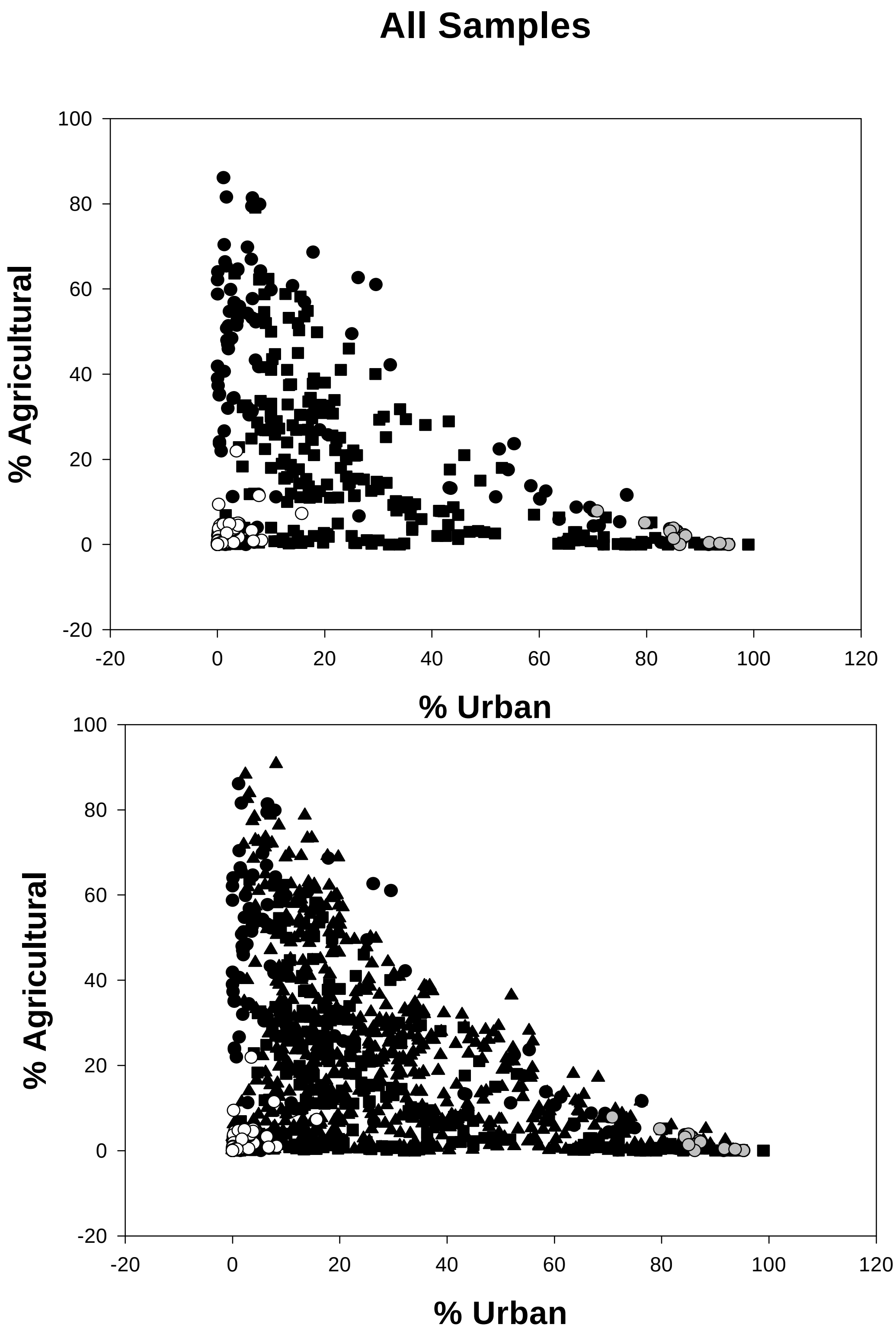


Figure 3.1.6. Relationship between %urban and %agriculture for fish and algae samples (top) and all samples (bottom). Sites are coded by source: AMNET (closed triangles), ANS (closed circles), EPA, NJ FIBI, and NAWQA (closed squares), PWD (gray circles), and TNC (open circles).

3.2 Macroinvertebrate Metrics

3.2.1 Subset Reliability

Results show that raw data from all data sources were well represented by 100 individual sub-samples for %CDF, %EPT, and FBI metrics (r^2=0.862-0.995) (Figs. 3.2.1-3.2.3). Not surprisingly, the relationship between raw and sub-sampled data was weaker for the Family Richness and #EPT taxa with the sub-sampled data having fewer families present than the raw data (r^2=0.328-0.833) (Figs. 3.2.4, 3.2.5).

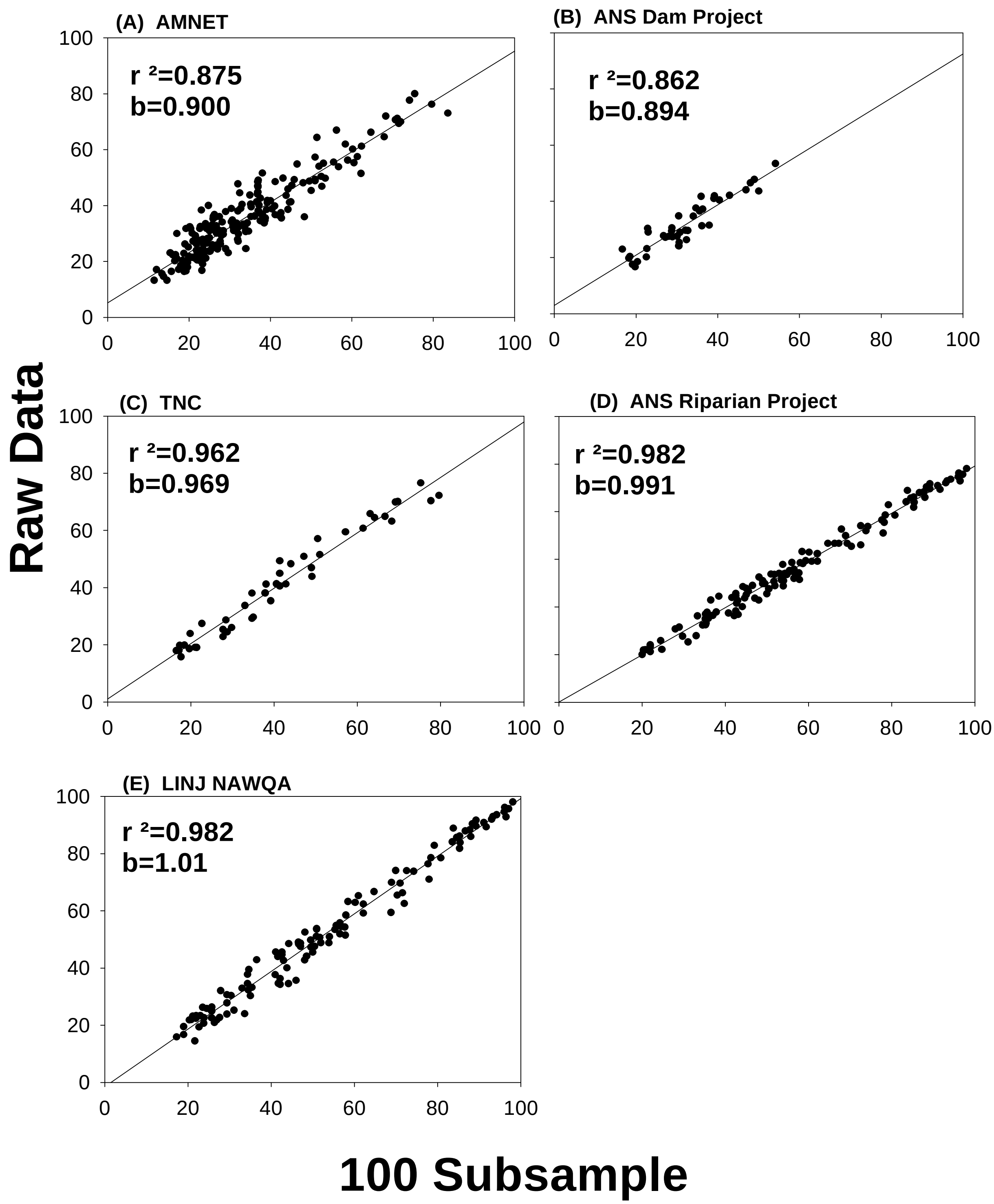


Figure 3.2.1. Relationship between raw data and 100-individual sub-sampled data for % Dominant Family metric for (A) AMNET, (B) ANS Dam Project, (C) TNC, (D) ANS Riparian Project, and (E) LINJ NAWQA data.

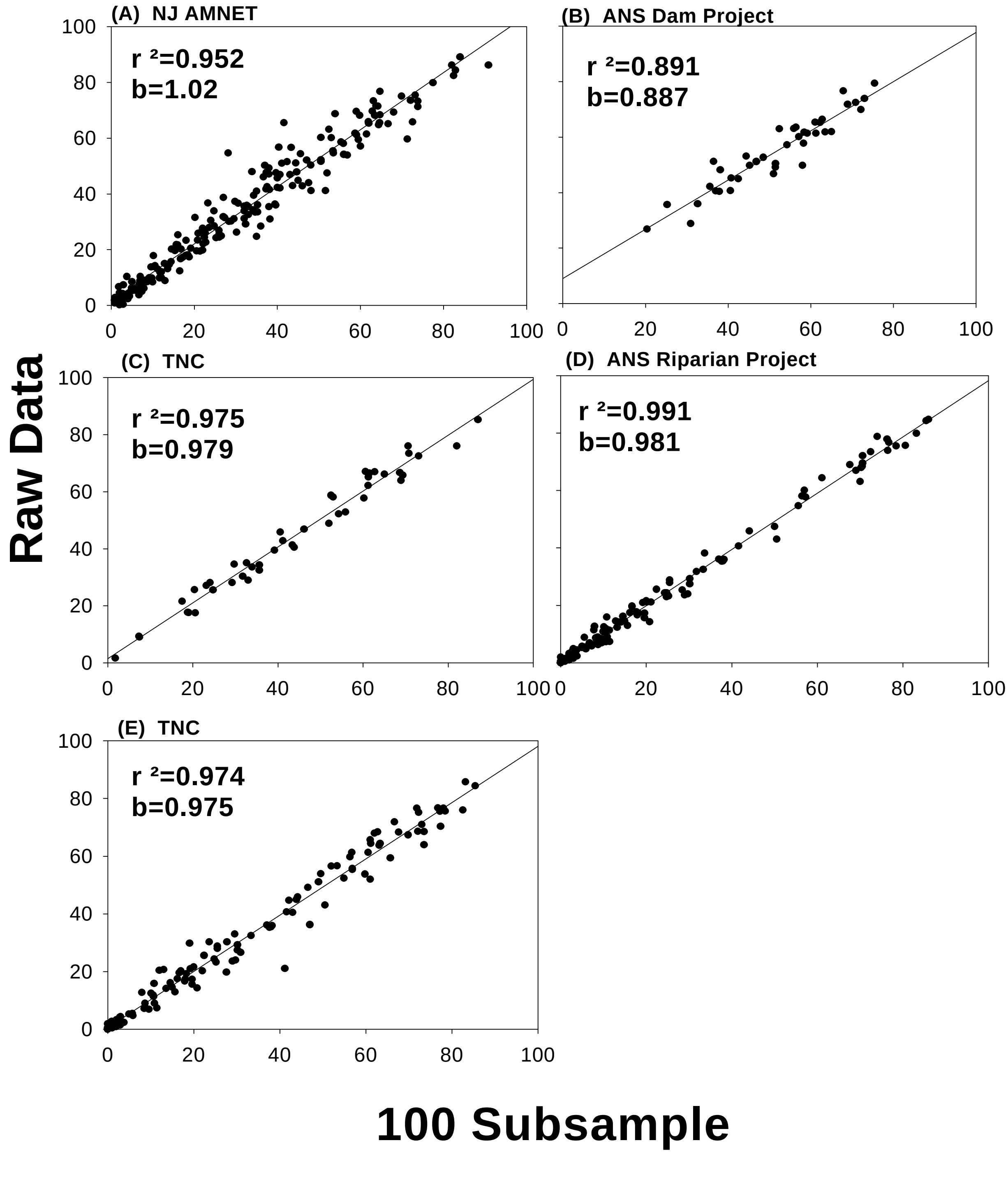


Figure 3.2.2. Relationship between raw data and 100-individual sub-sampled data for % EPT metric for (A) AMNET, (B) ANS Dam Project, (C) TNC, (D) ANS Riparian Project, and (E) LINJ NAWQA data.

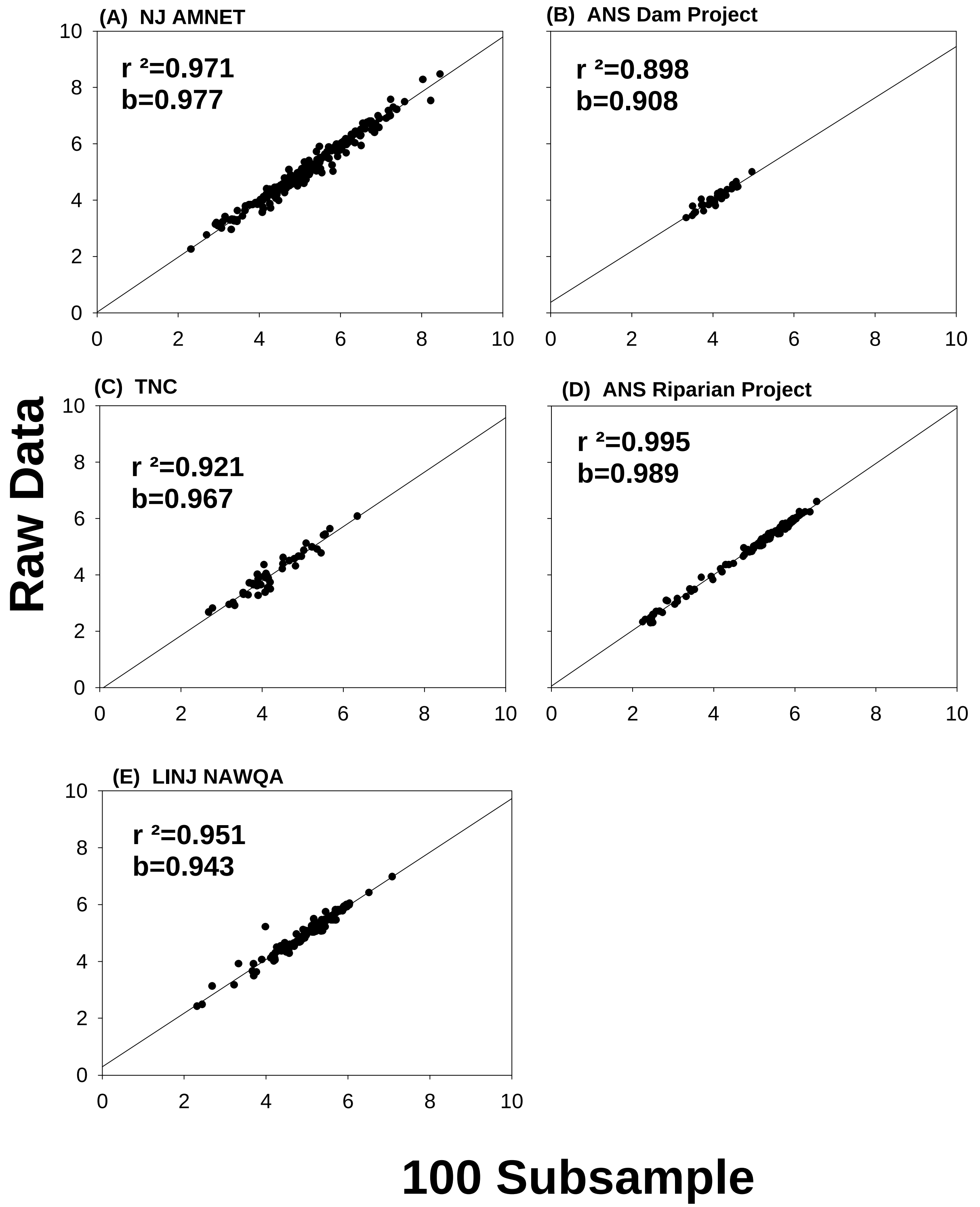


Figure 3.2.3. Relationship between raw data and 100-individual sub-sampled data for FBI metric for (A) AMNET, (B) ANS Dam Project, (C) TNC, (D) ANS Riparian Project, and (E) LINJ NAWQA data.

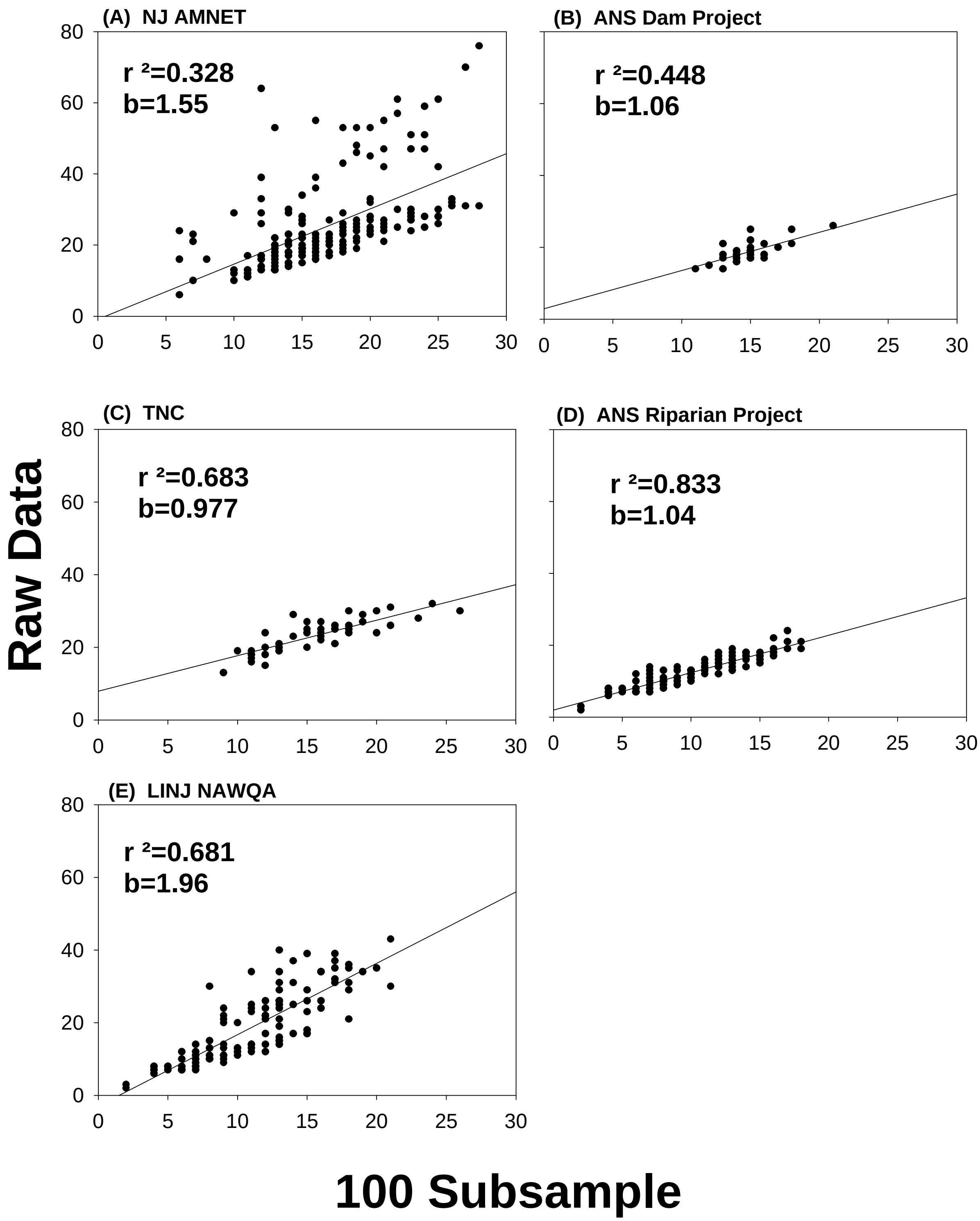


Figure 3.2.4. Relationship between raw data and 100-individual sub-sampled data for Family Richness metric for (A) AMNET, (B) ANS Dam Project, (C) TNC, (D) ANS Riparian Project, and (E) LINJ NAWQA data.

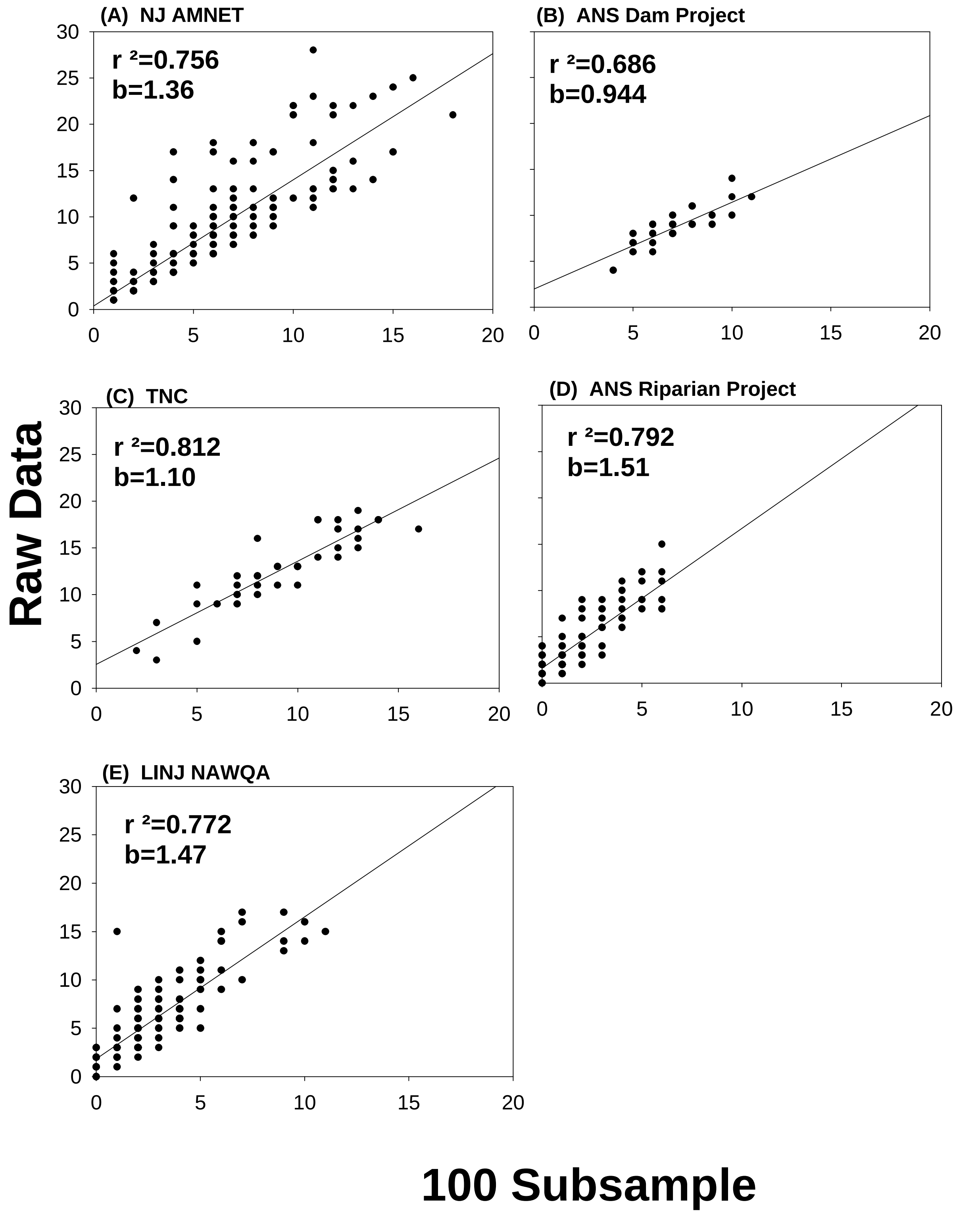


Figure 3.2.5. Relationship between raw data and 100-individual sub-sampled data for # EPT Families metric for (A) AMNET, (B) ANS Dam Project, (C) TNC, (D) ANS Riparian Project, and (E) LINJ NAWQA data.

3.2.2 Correlations among Macroinvertebrate Metrics and Relationships with Watershed Characteristics

Initial correlation analyses show strong relationships among metrics (Table 3.2.1), although the direction (positive or negative) strength of relationships differed when each dataset was examined separately (Tables 3.2.1-3.2.7). In all cases, there were strong positive correlations between metrics calculated at the family and genus level (e.g., # EPT families and #EPT Genera) and other related metrics (e.g., # EPT Genera and # Plecopteran, Ephemeropteran, and Trichopteran taxa). Percent dominant family and genera were typically negatively associated with taxa richness and taxa diversity, whereas the % Intolerant taxa was negatively correlated with FBI.

Of the metrics developed, 16 were selected for analyses based on their ability to describe diversity (Family Richness, Simpson's diversity, %Dominant Family), trophic structure (functional feeding groups: %collector-filterer, %predator, %scraper, %shredder, %collector-gatherer), and relationship to water quality (%Chironomidae, %Hyropsychidae, %Baetidae, #EPT families, % EPT taxa, Average Tolerance, %non-insect, and Chironomidae:EPT). Metrics used by NJBFBM's AMNET program (Family Richness, %Dominant Family, #EPT families, %EPT taxa, and Average Tolerance(FBI)) were maintained in the selected metrics despite some redundancy with other metrics.

Most of the macroinvertebrate metrics show expected relationships with land use, e.g., decreased abundance of specialized trophic groups (Fig 3.2.6-3.2.13) and increased tolerance, dominance and %chironomids (Fig. 3.6.9 and Fig 3.6.11) with increased urbanization. The relationships are best for EPT family richness, Average Tolerance, %Scrapers, and %Shredders (although many sites along the gradient had no shredders in the samples). Several of the metrics (e.g., %Predators) had decreasing values with increasing urbanization, but with a number of urban sites with high values. Many of these high values are from urban sites in Philadelphia (PWD data).

Because of the intercorrelation among individual metrics, principal component analyses (PCA) were done to define major axes of variation in macroinvertebrate assemblages and to relate these to watershed characteristics and to fish and algal assemblages. The PCA was performed on the normalized transformations of metrics (square root for % metrics, and ln of the chironomid/EPT ratio).The macroinvertebrate metrics showed strong loadings of most of the metrics on the first few components (Table 3.2.8). The first four components accounted for 70% of the total variance.

The first component (hereafter Mpca1) was highly positively correlated with proportion of dominant family and average tolerance, and highly negatively correlated with proportion EPT,EPT richness, Simpson's diversity, family richness, and moderately correlated with metrics related to trophic structure and proportion of noninsects and chironomids (Table 3.2.8). This component is interpretable as an impairment gradient (Figure 3.6.14). Sites with high values of Mpca1 have relatively high proportions of amphipods (*Gammarus* and *Asellus*), worms (the tubificids *Devos nivea, Nais* spp., and *Limnodrilus* spp.), molluscs (e.g., *Physella* and *Sphaerium),* bryozoans (*Plumatella*), chironomids and other tolerant taxa. Mpca1 is significantly positively related ($p<0.000001$) with the sqrt(proportion urban) and ln(Watershed area) ($p<0.03$). Other land uses are not significant in regression models containing sqrt(proportion urban).

The second component (hereafter mpca2) is strongly negatively correlated with proportion of collector-filterers, proportion of hydropsychids, and proportion of predators. Mpca2 (Fig 3.2.15) is significantly, positively correlated ($p<0.000001$) with the sqrt(proportion of wetland), but not with watershed area or other land uses (Fig. 3.2.15). Sites with low values of Mpca2 have relatively high abundance of a number of insect taxa, including caddisflies (*Hydropsyche, Cheumatopsyche),* mayflies (*Caenis, Seratella, Ephemerella*), blackflies (*Simulium, Prosimulium*), midges (*Phaenopsectra, Micropsectra*), and a tipulid (*Hexastoma*). This relationship reflects importance of macroinvertebrates which filter particulates (mainly phytoplankton) derived from upstream lakes, ponds and reservoirs.

The third component (hereafter Mpca3) is strongly negatively correlated with proportion of chironomids, highly positively correlated with proportion of noninsects, and moderately correlated with richness, EPT and trophic metrics. The sites with the highest values of Mpca3 had relatively high proportions of non-insect taxa such as worms (including Tubificids, Naids and Lumbriculids), snails (*Physella* and *Helisoma*), amphipods (*Gammarus*), flatworms (*Dugesia),* and bryozoans (*Plumatella).* Many of these taxa also contribute to metrics loading on Mpca1. However, several chironomids are frequent in the high Mpca1 sites, but not in the Mpca3 sites. The loadings of metrics and the occurrence of taxa at sites along the Mpca3 gradient suggest that this is also an enrichment/impairment gradient, but is different from the primary gradient represented by Mpca1. However, Mpca3 is significantly positively related to sqrt(proportion wetland) (Fig. 3.2.15), and other land uses are not significant in regressions of Mpca3 which contain sqrt(%wetland).

The fourth component (hereafter Mpca4) is highly positively correlated with proportion of collector-gatherers. Sites with low values of Mpca4 have relatively high numbers of a variety of mayflies and stoneflies typical of cool, unimpaired streams, while high values are associated with a few taxa (e.g., *Caenis, Gammarus, Limnodrilus,* and *Neocloeon).* Mpca4 is correlated with ln(watershed area) and with the square root of the proportion of the four land use types (Fig. 3.2.16). However, the slopes of the Mpca4-land use relationships are all negative, suggesting that Mpca4 is related to interactions between land use and watershed area.

Table 3.2.1. Pearson correlation coefficients of invertebrate metrics based on data from NJ DEP, USGS (NAWQA), TNC, ANS Riparian Study, and ANS Dam Removal Study. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	%Collector-Filterer	% Scraper	%Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	-0.129	1													
%Diptera	-0.126	0.93	1												
% Dominant Familiy	-0.252	0.692	0.636	1											
% Dominant Genera	-0.229	0.633	0.587	0.85	1										
%Hydropsychidae	0	-0.36	-0.373	-0.078	-0.201	1									
%Intolerant	0.049	-0.255	-0.254	-0.281	-0.269	-0.164	1								
% non-insect	-0.211	-0.314	-0.353	-0.088	-0.016	-0.185	-0.366	1							
%Tolerant	-0.147	-0.195	-0.14	-0.147	-0.11	-0.175	-0.272	0.63	1						
%EPT	0.32	-0.53	-0.543	-0.422	-0.464	0.554	0.599	-0.496	-0.393	1					
%Collector-Filterer	0.06	-0.186	-0.207	-0.081	-0.155	0.506	-0.021	-0.233	-0.201	0.386	1				
%Scraper	0.162	-0.309	-0.326	-0.33	-0.242	-0.012	0.38	-0.341	-0.279	0.39	0.252	1			
%Collector-Gatherer	0.274	-0.097	-0.119	-0.211	-0.287	-0.118	0.3	-0.179	-0.042	0.21	0.093	0.458	1		
%Predator	-0.022	-0.233	-0.265	-0.135	-0.195	0.62	-0.037	-0.189	-0.168	0.382	0.601	0.186	0.005	1	
%Shredder	0.007	-0.005	0.027	-0.064	-0.17	-0.029	0.263	-0.17	-0.03	0.14	-0.01	0.035	0.326	-0.043	1
# Dipteran Genera	0.12	0.041	0.118	-0.199	-0.423	-0.02	0.004	-0.049	0.105	-0.012	-0.155	-0.298	0.137	-0.132	0.263
# Ephem Families	0.37	-0.33	-0.34	-0.443	-0.49	0.048	0.431	-0.36	-0.3	0.569	0.171	0.333	0.259	0.1	0.053
# Ephem. Genera	0.442	-0.28	-0.279	-0.406	-0.449	-0.019	0.47	-0.349	-0.275	0.559	0.081	0.239	0.235	-0.006	0.108
# non-insect	-0.147	-0.271	-0.274	-0.249	-0.226	-0.117	-0.358	0.754	0.605	-0.395	-0.212	-0.298	-0.173	-0.145	-0.141
# Plecop. Families	0.053	-0.11	-0.093	-0.241	-0.26	-0.2	0.702	-0.304	-0.2	0.403	-0.054	0.182	0.206	-0.068	0.242
# Plecop. Genera	0.106	-0.143	-0.111	-0.273	-0.282	-0.194	0.714	-0.32	-0.205	0.431	-0.058	0.184	0.186	-0.083	0.216
# Tricop. Genera	0.235	-0.364	-0.349	-0.442	-0.483	0.268	0.409	-0.337	-0.248	0.664	0.124	0.179	0.105	0.087	0.098
# Tricop. Families	0.233	-0.302	-0.295	-0.437	-0.435	0.058	0.474	-0.342	-0.245	0.592	0.078	0.252	0.142	0.055	0.053
Ave. Tolerance (FBI)	-0.199	0.368	0.404	0.358	0.336	-0.231	-0.751	0.566	0.623	-0.807	-0.228	-0.531	-0.241	-0.236	-0.139
Chiro:EPT	-0.169	0.517	0.467	0.44	0.466	-0.278	-0.215	0.008	0.019	-0.431	-0.161	-0.205	-0.185	-0.215	-0.08
#EPT Families	0.284	-0.322	-0.317	-0.481	-0.506	-0.022	0.656	-0.422	-0.313	0.666	0.089	0.324	0.248	0.044	0.133
Family Richness	0.159	-0.501	-0.482	-0.691	-0.68	-0.048	0.331	0.042	0.08	0.353	-0.021	0.137	0.125	-0.01	0.071
Genera Richness	0.218	-0.411	-0.369	-0.622	-0.728	0.001	0.267	0.004	0.098	0.321	-0.072	-0.012	0.164	-0.072	0.179
Simpson's Diversity	0.213	-0.667	-0.613	-0.834	-0.97	0.205	0.266	0.062	0.128	0.453	0.143	0.233	0.308	0.187	0.178
Simpson's Evenness	0.19	-0.327	-0.328	-0.598	-0.719	0.068	0.14	-0.064	0.045	0.264	0.129	0.223	0.163	0.132	0.074
#EPT Genera	0.328	-0.366	-0.349	-0.501	-0.528	0.06	0.633	-0.423	-0.315	0.709	0.095	0.295	0.225	0.043	0.151

Table 3.2.1 (continued). Pearson correlation coefficients of invertebrate metrics based on data from NJ DEP, USGS (NAWQA), TNC, ANS Riparian Study, and ANS Dam Removal Study. Shaded areas represent correlation coefficients $0.7 or

	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro: EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	0.146	1														
# Ephem. Genera	0.204	0.862	1													
# non-insect	0.132	-0.221	-0.22	1												
# Plecop. Families	0.161	0.351	0.38	-0.291	1											
# Plecop. Genera	0.176	0.382	0.454	-0.3	0.957	1										
# Tricop. Genera	0.206	0.516	0.577	-0.232	0.395	0.426	1									
# Tricop. Families	0.16	0.522	0.58	-0.254	0.452	0.481	0.883	1								
Ave. Tolerance (FBI)	0.087	-0.55	-0.522	0.513	-0.514	-0.53	-0.554	-0.566	1							
Chiro:EPT	-0.129	-0.31	-0.271	0.025	-0.165	-0.17	-0.332	-0.283	0.328	1						
#EPT Families	0.195	0.788	0.771	-0.318	0.717	0.729	0.785	0.865	-0.684	-0.324	1					
Family Richness	0.352	0.609	0.583	0.329	0.428	0.447	0.562	0.607	-0.327	-0.319	0.698	1				
Genera Richness	0.689	0.565	0.605	0.285	0.379	0.419	0.588	0.555	-0.239	-0.326	0.637	0.878	1			
Simpson's Diversity	0.423	0.469	0.431	0.252	0.248	0.265	0.461	0.413	-0.33	-0.507	0.482	0.678	0.723	1		
Simpson's Evenness	0.2	0.323	0.309	0.097	0.137	0.17	0.286	0.251	-0.223	-0.203	0.304	0.354	0.385	0.604	1	
#EPT Genera	0.212	0.776	0.832	-0.315	0.655	0.701	0.846	0.841	-0.688	-0.35	0.961	0.679	0.674	0.505	0.321	1

Table 3.2.2. Pearson correlation coefficients of invertebrate metrics based on data from ANS Riparian Study. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	%Collector-Filterer	% Scraper	%Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	-0.03	1													
%Diptera	-0.049	0.952	1												
% Dominant Familiy	-0.109	0.865	0.769	1											
% Dominant Genera	-0.079	0.934	0.855	0.968	1										
%Hydropsychidae	-0.089	-0.31	-0.305	-0.101	-0.224	1									
%Intolerant	-0.099	-0.687	-0.702	-0.464	-0.522	-0.09	1								
% non-insect	-0.008	-0.02	-0.087	-0.194	-0.136	-0.099	-0.338	1							
%Tolerant	-0.073	-0.016	-0.081	-0.133	-0.092	-0.104	-0.23	0.755	1						
%EPT	0.08	-0.81	-0.821	-0.553	-0.652	0.365	0.841	-0.346	-0.269	1					
%Collector-Filterer	-0.08	-0.334	-0.315	-0.104	-0.255	0.771	-0.016	-0.085	-0.147	0.344	1				
%Scraper	0.143	-0.743	-0.762	-0.64	-0.653	-0.089	0.661	-0.111	-0.07	0.599	-0.009	1			
%Collector-Gatherer	0.314	-0.699	-0.693	-0.615	-0.605	-0.053	0.66	-0.119	-0.084	0.64	0.027	0.807	1		
%Predator	-0.085	-0.366	-0.351	-0.184	-0.3	0.944	-0.057	-0.013	-0.089	0.372	0.733	-0.062	-0.027	1	
%Shredder	-0.086	-0.564	-0.583	-0.419	-0.459	-0.063	0.783	-0.226	-0.128	0.643	0.014	0.398	0.365	-0.035	1
# Dipteran Genera	-0.08	-0.379	-0.308	-0.435	-0.42	0.156	0.167	-0.035	0.001	0.229	0.087	0.282	0.176	0.238	0.2
# Ephem Families	0.265	-0.559	-0.533	-0.497	-0.52	-0.198	0.633	-0.265	-0.169	0.578	-0.058	0.64	0.637	-0.178	0.436
# Ephem. Genera	0.35	-0.507	-0.501	-0.453	-0.491	-0.186	0.633	-0.279	-0.215	0.604	-0.042	0.499	0.552	-0.17	0.505
# non-insect	-0.057	0.005	-0.044	-0.164	-0.111	-0.013	-0.389	0.799	0.677	-0.364	-0.073	-0.12	-0.123	0.037	-0.252
# Plecop. Families	-0.108	-0.594	-0.59	-0.452	-0.489	-0.08	0.894	-0.321	-0.221	0.75	0.008	0.547	0.559	-0.053	0.772
# Plecop. Genera	-0.099	-0.592	-0.58	-0.463	-0.491	-0.089	0.882	-0.319	-0.216	0.739	0	0.545	0.555	-0.067	0.788
# Tricop. Genera	-0.052	-0.507	-0.434	-0.492	-0.51	0.146	0.439	-0.224	-0.198	0.557	0.253	0.401	0.293	0.18	0.292
# Tricop. Families	-0.043	-0.499	-0.448	-0.479	-0.499	0.067	0.473	-0.224	-0.197	0.568	0.142	0.423	0.303	0.114	0.309
Ave. Tolerance (FBI)	0.021	0.818	0.816	0.583	0.667	-0.141	-0.936	0.387	0.35	-0.923	-0.197	-0.734	-0.711	-0.17	-0.722
Chiro:EPT	-0.135	0.502	0.457	0.484	0.501	-0.252	-0.304	0.107	0.156	-0.461	-0.203	-0.341	-0.367	-0.257	-0.243
#EPT Families	0.022	-0.673	-0.637	-0.586	-0.618	-0.063	0.804	-0.329	-0.243	0.776	0.056	0.643	0.586	-0.022	0.608
Family Richness	-0.08	-0.737	-0.699	-0.786	-0.779	-0.026	0.51	0.178	0.211	0.504	0.048	0.603	0.527	0.053	0.434
Genera Richness	0.003	-0.781	-0.732	-0.813	-0.824	0.016	0.56	0.067	0.119	0.585	0.132	0.639	0.553	0.08	0.49
Simpson's Diversity	0.11	-0.913	-0.832	-0.961	-0.975	0.211	0.475	0.2	0.136	0.605	0.239	0.634	0.616	0.289	0.414
Simpson's Evenness	0.09	-0.582	-0.555	-0.601	-0.65	0.146	0.344	-0.009	-0.082	0.456	0.181	0.417	0.342	0.207	0.331
#EPT Genera	0.076	-0.679	-0.639	-0.6	-0.637	-0.024	0.796	-0.331	-0.26	0.794	0.115	0.599	0.572	0.009	0.633

Table 3.2.2 (continued). Pearson correlation coefficients of invertebrate metrics based on data from ANS Riparian Study. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro:EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	0.083	1														
# Ephem. Genera	0.018	0.923	1													
# non-insect	0.013	-0.314	-0.358	1												
# Plecop. Families	0.221	0.556	0.578	-0.386	1											
# Plecop. Genera	0.24	0.574	0.579	-0.377	0.975	1										
# Tricop. Genera	0.315	0.393	0.354	-0.244	0.464	0.445	1									
# Tricop. Families	0.255	0.434	0.408	-0.233	0.463	0.447	0.955	1								
Ave. Tolerance (FBI)	-0.275	-0.632	-0.627	0.407	-0.836	-0.82	-0.537	-0.552	1							
Chiro:EPT	-0.301	-0.3	-0.305	0.173	-0.304	-0.298	-0.343	-0.284	0.426	1						
#EPT Families	0.244	0.77	0.74	-0.375	0.816	0.804	0.797	0.833	-0.821	-0.363	1					
Family Richness	0.513	0.524	0.453	0.278	0.537	0.534	0.614	0.636	-0.554	-0.339	0.708	1				
Genera Richness	0.518	0.605	0.553	0.133	0.586	0.597	0.674	0.656	-0.632	-0.395	0.764	0.958	1			
Simpson's Diversity	0.435	0.492	0.45	0.178	0.45	0.449	0.484	0.461	-0.616	-0.527	0.574	0.792	0.823	1		
Simpson's Evenness	0.021	0.284	0.32	-0.142	0.276	0.27	0.262	0.31	-0.47	-0.274	0.36	0.244	0.301	0.53	1	
#EPT Genera	0.255	0.762	0.775	-0.393	0.815	0.81	0.798	0.796	-0.825	-0.404	0.978	0.683	0.778	0.594	0.358	

Table 3.2.3. Pearson correlation coefficients of invertebrate metrics based on data from NJ DEP AMNET. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	% Collector-Filterer	% Scraper	% Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	0.002	1													
%Diptera	0.029	0.814	1												
% Dominant Familiy	-0.234	0.244	0.234	1											
% Dominant Genera	-0.18	-0.173	-0.145	0.735	1										
%Hydropsychidae	0.038	-0.166	-0.212	-0.05	-0.079	1									
%Intolerant	0.104	-0.173	-0.137	-0.308	-0.282	-0.067	1								
% non-insect	-0.346	-0.334	-0.419	0.251	0.494	-0.328	-0.495	1							
%Tolerant	-0.225	-0.131	-0.002	-0.032	0.074	-0.265	-0.327	0.494	1						
%EPT	0.359	-0.244	-0.252	-0.367	-0.349	0.553	0.665	-0.711	-0.501	1					
%Collector-Filterer	0.153	-0.018	-0.025	-0.155	-0.123	-0.018	0.182	-0.15	-0.02	0.177	1				
%Scraper	0.134	-0.121	-0.141	-0.296	-0.277	-0.117	0.409	-0.291	-0.199	0.343	0.256	1			
%Collector-Gatherer	0.337	0.044	-0.027	-0.152	-0.166	-0.163	0.141	-0.111	0.104	0.119	0.187	0.394	1		
%Predator	0.061	-0.08	-0.135	-0.076	-0.049	0.606	-0.122	-0.17	-0.032	0.269	0.126	-0.092	0.012	1	
%Shredder	0.087	0.272	0.236	0.075	-0.062	-0.155	0.243	-0.221	-0.077	0.071	0.11	0.106	0.415	-0.101	1
# Dipteran Genera	0.134	0.543	0.601	-0.143	-0.407	-0.035	0.014	-0.431	-0.104	-0.002	0.056	-0.038	0.068	-0.042	0.156
# Ephem Families	0.335	0.052	0.052	-0.277	-0.379	0.008	0.507	-0.633	-0.46	0.568	0.122	0.301	0.266	-0.033	0.184
# Ephem. Genera	0.343	0.019	0.041	-0.296	-0.367	-0.053	0.643	-0.601	-0.432	0.59	0.163	0.378	0.265	-0.07	0.28
# non-insect	-0.256	-0.094	-0.138	-0.077	-0.008	-0.282	-0.444	0.657	0.524	-0.598	-0.138	-0.196	-0.005	-0.069	-0.176
# Plecop. Families	0.1	-0.091	-0.003	-0.29	-0.298	-0.138	0.759	-0.458	-0.277	0.529	0.239	0.307	0.1	-0.179	0.167
# Plecop. Genera	0.123	-0.107	0.007	-0.291	-0.304	-0.125	0.769	-0.465	-0.282	0.529	0.266	0.286	0.063	-0.166	0.159
# Tricop. Genera	0.192	-0.189	-0.162	-0.412	-0.381	0.284	0.553	-0.549	-0.373	0.739	0.109	0.386	-0.009	0.013	0.035
# Tricop. Families	0.211	-0.196	-0.154	-0.422	-0.347	0.093	0.577	-0.498	-0.322	0.662	0.123	0.387	-0.019	-0.102	0.018
Ave. Tolerance (FBI)	-0.268	0.186	0.239	0.331	0.27	-0.336	-0.694	0.647	0.731	-0.82	-0.135	-0.406	-0.006	-0.073	-0.096
Chiro:EPT	-0.161	0.372	0.276	0.207	0.122	-0.256	-0.183	0.158	0.08	-0.365	-0.068	-0.171	-0.137	-0.132	0.004
#EPT Families	0.271	-0.106	-0.057	-0.414	-0.42	0.005	0.734	-0.649	-0.434	0.727	0.187	0.412	0.13	-0.121	0.136
Family Richness	0.095	-0.09	-0.098	-0.547	-0.519	-0.17	0.44	-0.319	-0.126	0.314	0.163	0.326	0.175	-0.091	0.092
Genera Richness	0.156	0.13	0.146	-0.524	-0.627	-0.112	0.416	-0.481	-0.17	0.319	0.134	0.301	0.186	-0.094	0.152
Simpson's Diversity	0.184	0.205	0.167	-0.721	-0.966	0.073	0.285	-0.511	-0.08	0.351	0.119	0.289	0.197	0.056	0.093
Simpson's Evenness	0.105	0.178	0.138	-0.53	-0.809	0.013	0.15	-0.321	-0.067	0.192	0.062	0.209	0.101	-0.006	0.046
#EPT Genera	0.278	-0.124	-0.068	-0.42	-0.431	0.076	0.759	-0.667	-0.454	0.765	0.168	0.431	0.131	-0.081	0.167

Table 3.2.3 (continued). Pearson correlation coefficients of invertebrate metrics based on data from NJ DEP AMNET. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro: EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	0.194	1														
# Ephem. Genera	0.158	0.859	1													
# non-insect	-0.115	-0.439	-0.436	1												
# Plecop. Families	0.126	0.448	0.498	-0.376	1											
# Plecop. Genera	0.122	0.428	0.497	-0.385	0.964	1										
# Tricop. Genera	0.071	0.501	0.561	-0.43	0.499	0.482	1									
# Tricop. Families	0.061	0.499	0.56	-0.386	0.547	0.529	0.902	1								
Ave. Tolerance (FBI)	-0.014	-0.589	-0.612	0.59	-0.523	-0.531	-0.659	-0.62	1							
Chiro:EPT	0.041	-0.272	-0.241	0.234	-0.154	-0.152	-0.283	-0.242	0.304	1						
#EPT Families	0.148	0.793	0.783	-0.489	0.773	0.744	0.809	0.873	-0.712	-0.279	1					
Family Richness	0.217	0.593	0.543	0.117	0.526	0.493	0.481	0.584	-0.388	-0.14	0.697	1				
Genera Richness	0.601	0.588	0.578	0.002	0.492	0.475	0.52	0.549	-0.372	-0.131	0.667	0.86	1			
Simpson's Diversity	0.422	0.393	0.392	0.03	0.296	0.297	0.378	0.342	-0.286	-0.105	0.422	0.533	0.652	1		
Simpson's Evenness	0.256	0.221	0.239	0.008	0.183	0.206	0.239	0.179	-0.127	-0.068	0.237	0.264	0.346	0.719	1	
#EPT Genera	0.136	0.763	0.823	-0.52	0.737	0.734	0.861	0.849	-0.748	-0.299	0.965	0.633	0.658	0.441	0.256	

Table 3.2.4. Pearson correlation coefficients of invertebrate metrics based on data from TNC. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	%Collector-Filterer	% Scraper	%Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	-0.543	1													
%Diptera	-0.533	0.984	1												
% Dominant Familiy	-0.535	0.908	0.874	1											
% Dominant Genera	-0.334	0.509	0.479	0.572	1										
%Hydropsychidae	-0.012	-0.461	-0.449	-0.255	-0.063	1									
%Intolerant	0.213	-0.279	-0.311	-0.315	-0.277	-0.397	1								
% non-insect	-0.361	0.125	0.104	0.075	0.097	-0.097	-0.087	1							
%Tolerant	-0.168	-0.028	-0.021	-0.116	0.057	-0.071	-0.223	0.716	1						
%EPT	0.656	-0.845	-0.853	-0.735	-0.42	0.392	0.409	-0.486	-0.278	1					
%Collector-Filterer	0.102	-0.308	-0.277	-0.189	-0.171	0.629	-0.294	-0.312	-0.29	0.312	1				
%Scraper	0.609	-0.67	-0.661	-0.622	-0.402	0.082	0.117	-0.35	-0.169	0.551	0.124	1			
%Collector-Gatherer	-0.14	0.496	0.457	0.449	0.449	-0.511	-0.224	0.288	0.4	-0.563	-0.412	-0.085	1		
%Predator	-0.248	-0.258	-0.251	-0.141	-0.18	0.652	-0.032	0.24	-0.056	0.162	0.332	-0.187	-0.639	1	
%Shredder	-0.261	0.322	0.321	0.242	0.09	-0.345	0.139	0.169	0.184	-0.254	-0.245	-0.447	0.055	-0.139	1
# Dipteran Genera	-0.289	0.46	0.489	0.315	-0.133	-0.372	0.052	-0.074	-0.148	-0.347	-0.072	-0.307	0.161	-0.076	0.178
# Ephem Families	0.506	-0.511	-0.534	-0.491	-0.312	0.022	0.295	-0.396	-0.213	0.563	0.094	0.607	-0.027	-0.213	-0.559
# Ephem. Genera	0.655	-0.614	-0.633	-0.565	-0.343	0.046	0.223	-0.427	-0.231	0.647	0.124	0.755	-0.107	-0.181	-0.551
# non-insect	-0.428	0.121	0.122	0.057	0.171	0.055	-0.2	0.678	0.456	-0.406	-0.174	-0.315	0.142	0.218	0.078
# Plecop. Families	0.066	0.022	-0.006	-0.123	-0.214	-0.344	0.491	-0.078	-0.054	0.074	-0.125	-0.008	-0.042	-0.135	0.185
# Plecop. Genera	0.302	-0.299	-0.322	-0.418	-0.399	-0.284	0.584	-0.194	-0.108	0.347	-0.109	0.326	-0.151	-0.134	-0.02
# Tricop. Genera	0.305	-0.487	-0.498	-0.437	-0.233	0.088	0.42	-0.434	-0.301	0.605	0.257	0.407	-0.322	0.011	-0.282
# Tricop. Families	0.32	-0.414	-0.422	-0.375	-0.204	0.034	0.312	-0.51	-0.33	0.544	0.226	0.391	-0.278	-0.092	-0.347
Ave. Tolerance (FBI)	-0.509	0.689	0.709	0.626	0.437	-0.083	-0.75	0.461	0.483	-0.822	-0.165	-0.483	0.519	-0.07	0.139
Chiro:EPT	-0.301	0.409	0.429	0.388	0.284	-0.256	-0.329	0.557	0.507	-0.606	-0.232	-0.347	0.476	-0.228	0.419
#EPT Families	0.417	-0.423	-0.451	-0.47	-0.352	-0.155	0.55	-0.479	-0.294	0.565	0.095	0.457	-0.183	-0.209	-0.32
Family Richness	0.354	-0.584	-0.574	-0.632	-0.377	0.128	0.347	-0.312	-0.13	0.57	0.267	0.436	-0.256	-0.005	-0.437
Genera Richness	0.255	-0.42	-0.411	-0.49	-0.481	-0.075	0.352	-0.358	-0.218	0.419	0.144	0.45	-0.176	-0.066	-0.368
Simpson's Diversity	0.321	-0.497	-0.462	-0.551	-0.921	-0.062	0.403	-0.183	-0.134	0.405	0.183	0.48	-0.354	0.071	-0.1
Simpson's Evenness	0.411	-0.415	-0.378	-0.516	-0.88	-0.062	0.21	-0.118	-0.055	0.334	0.064	0.45	-0.334	0	-0.05
#EPT Genera	0.534	-0.613	-0.63	-0.622	-0.419	-0.035	0.477	-0.468	-0.263	0.674	0.116	0.635	-0.232	-0.152	-0.426

Table 3.2.4 (continued). Pearson correlation coefficients of invertebrate metrics based on data from TNC. Shaded areas represent correlation coefficients \$0.7 or #-0.7.

	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro: EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	-0.127	1														
# Ephem. Genera	-0.268	0.876	1													
# non-insect	0.015	-0.379	-0.373	1												
# Plecop. Families	0.307	0.095	-0.01	-0.264	1											
# Plecop. Genera	0.127	0.402	0.383	-0.318	0.879	1										
# Tricop. Genera	-0.193	0.422	0.535	-0.328	0.127	0.342	1									
# Tricop. Families	-0.137	0.455	0.582	-0.397	0.026	0.227	0.866	1								
Ave. Tolerance (FBI)	0.142	-0.543	-0.542	0.406	-0.342	-0.556	-0.598	-0.535	1							
Chiro:EPT	-0.019	-0.418	-0.397	0.343	-0.124	-0.251	-0.534	-0.52	0.615	1						
#EPT Families	0.036	0.708	0.67	-0.51	0.583	0.754	0.708	0.74	-0.693	-0.518	1					
Family Richness	0.01	0.663	0.653	-0.1	0.327	0.57	0.655	0.671	-0.591	-0.464	0.809	1				
Genera Richness	0.349	0.662	0.658	-0.138	0.335	0.573	0.538	0.57	-0.523	-0.446	0.758	0.857	1			
Simpson's Diversity	0.205	0.38	0.416	-0.21	0.256	0.47	0.324	0.307	-0.532	-0.277	0.458	0.479	0.611	1		
Simpson's Evenness	-0.003	0.297	0.39	-0.227	0.129	0.339	0.197	0.181	-0.336	-0.097	0.288	0.258	0.346	0.832	1	
#EPT Genera	-0.105	0.796	0.853	-0.416	0.365	0.667	0.747	0.741	-0.719	-0.518	0.921	0.846	0.831	0.523	0.376	

Table 3.2.5. Pearson correlation coefficients of invertebrate metrics based on data from LINJ NAWQA. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	%Collector-Filterer	% Scraper	%Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	-0.052	1													
%Diptera	-0.072	0.793	1												
% Dominant Familiy	-0.28	-0.097	-0.166	1											
% Dominant Genera	-0.347	-0.205	-0.175	0.755	1										
%Hydropsychidae	-0.12	-0.264	-0.345	0.834	0.514	1									
%Intolerant	0.117	-0.303	-0.341	-0.197	0.048	-0.292	1								
% non-insect	-0.257	0.125	-0.042	-0.277	-0.202	-0.334	-0.233	1							
%Tolerant	-0.157	0.275	0.172	-0.253	-0.238	-0.354	-0.235	0.639	1						
%EPT	0.236	-0.542	-0.601	0.458	0.367	0.638	0.365	-0.596	-0.61	1					
%Collector-Filterer	0.102	0.253	0.084	0.303	-0.115	0.433	-0.147	-0.132	-0.343	0.249	1				
%Scraper	0.349	-0.276	-0.119	-0.408	-0.276	-0.267	0.035	-0.2	-0.124	-0.012	-0.386	1			
%Collector-Gatherer	0.079	0.556	0.511	-0.424	-0.424	-0.421	-0.358	0.397	0.436	-0.684	-0.052	0.075	1		
%Predator	-0.224	-0.034	0.012	0.352	0.208	0.402	-0.33	0.115	0.066	0.011	0.086	-0.217	-0.163	1	
%Shredder	-0.075	0.211	0.405	-0.14	-0.062	-0.144	-0.023	-0.193	0.211	-0.25	-0.239	0.142	0.356	0.039	1
# Dipteran Genera	0.023	0.544	0.684	-0.188	-0.223	-0.136	-0.311	-0.12	0.168	-0.406	0.029	-0.038	0.394	0.15	0.51
# Ephem Families	0.539	-0.194	-0.204	-0.198	-0.187	-0.101	0.503	-0.354	-0.338	0.472	0.094	0.213	-0.3	-0.139	-0.203
# Ephem. Genera	0.699	-0.219	-0.224	-0.258	-0.218	-0.161	0.442	-0.321	-0.332	0.479	0.053	0.315	-0.294	-0.226	-0.253
# non-insect	-0.117	-0.055	-0.094	-0.49	-0.36	-0.468	-0.091	0.725	0.444	-0.465	-0.203	0.087	0.227	0.038	-0.211
# Plecop. Families	-0.18	-0.152	-0.067	-0.185	-0.174	-0.124	0.322	-0.097	-0.04	0.113	-0.099	0.095	-0.149	-0.01	0.321
# Plecop. Genera	-0.148	-0.146	-0.066	-0.193	-0.178	-0.12	0.325	-0.105	-0.013	0.125	-0.09	0.111	-0.156	-0.035	0.289
# Tricop. Genera	0.48	-0.418	-0.387	-0.014	-0.176	0.228	0.395	-0.45	-0.346	0.658	0.195	0.136	-0.399	-0.185	-0.129
# Tricop. Families	0.329	-0.276	-0.27	-0.27	-0.192	-0.082	0.397	-0.39	-0.267	0.453	-0.093	0.265	-0.282	-0.336	-0.045
Ave. Tolerance (FBI)	-0.092	0.681	0.687	-0.076	-0.184	-0.159	-0.647	0.493	0.509	-0.642	0.1	-0.105	0.614	0.227	0.185
Chiro:EPT	-0.148	0.47	0.272	-0.005	0.003	-0.381	-0.138	0.33	0.43	-0.513	-0.171	-0.269	0.22	-0.137	-0.117
#EPT Families	0.434	-0.288	-0.273	-0.295	-0.245	-0.127	0.563	-0.426	-0.339	0.529	-0.02	0.28	-0.35	-0.258	-0.065
Family Richness	0.341	-0.302	-0.217	-0.684	-0.577	-0.458	0.387	0.015	-0.069	0.06	-0.188	0.426	-0.046	-0.163	-0.053
Genera Richness	0.383	0.041	0.185	-0.616	-0.569	-0.381	0.179	-0.172	-0.025	-0.068	-0.07	0.365	0.107	-0.089	0.264
Simpson's Diversity	0.337	0.152	0.149	-0.795	-0.949	-0.566	0.055	0.148	0.169	-0.334	0.028	0.296	0.324	-0.29	0.095
Simpson's Evenness	0.073	0.175	0.07	-0.468	-0.624	-0.465	-0.117	0.295	0.404	-0.506	-0.05	0.036	0.28	-0.151	-0.075
#EPT Genera	0.542	-0.351	-0.293	-0.239	-0.24	-0.04	0.535	-0.473	-0.367	0.598	0.021	0.298	-0.36	-0.243	-0.069

Table 3.2.5 (continued). Pearson correlation coefficients of invertebrate metrics based on data from LINJ NAWQA. Shaded areas represent correlation coefficients $0.7 or #-0.7.

	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro: EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	-0.102	1														
# Ephem. Genera	-0.178	0.903	1													
# non-insect	-0.176	-0.207	-0.085	1												
# Plecop. Families	-0.047	0.097	0.057	-0.142	1											
# Plecop. Genera	-0.066	0.112	0.086	-0.16	0.967	1										
# Tricop. Genera	-0.18	0.616	0.582	-0.377	0.191	0.228	1									
# Tricop. Families	-0.05	0.514	0.535	-0.268	0.124	0.175	0.74	1								
Ave. Tolerance (FBI)	0.449	-0.343	-0.315	0.248	-0.19	-0.199	-0.473	-0.46	1							
Chiro:EPT	-0.034	-0.192	-0.164	0.153	-0.078	-0.07	-0.472	-0.32	0.246	1						
#EPT Families	-0.094	0.851	0.8	-0.29	0.343	0.372	0.779	0.846	-0.478	-0.295	1					
Family Richness	0.013	0.608	0.606	0.34	0.273	0.29	0.478	0.547	-0.268	-0.228	0.69	1				
Genera Richness	0.546	0.49	0.456	0.096	0.223	0.207	0.399	0.455	0.01	-0.313	0.564	0.78	1			
Simpson's Diversity	0.284	0.246	0.271	0.35	0.201	0.192	0.211	0.284	0.073	-0.007	0.333	0.665	0.678	1		
Simpson's Evenness	0.025	-0.091	-0.066	0.371	0.026	0.04	-0.194	-0.09	0.023	0.561	-0.092	0.214	0.076	0.603	1	
#EPT Genera	-0.137	0.841	0.835	-0.301	0.315	0.348	0.858	0.774	-0.476	-0.382	0.95	0.665	0.567	0.312	-0.147	

Table 3.2.6. Pearson correlation coefficients of invertebrate metrics based on data from ANS Dam Project (GG2). Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	%Collector-Filterer	% Scraper	%Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	-0.055	1													
%Diptera	0.05	0.939	1												
% Dominant Familiy	-0.135	-0.078	-0.111	1											
% Dominant Genera	-0.307	0.15	0.053	0.768	1										
%Hydropsychidae	-0.239	-0.228	-0.244	0.754	0.569	1									
%Intolerant	-0.245	-0.229	-0.293	-0.356	-0.153	-0.201	1								
% non-insect	-0.2	0.083	0.026	-0.086	0.043	0.078	-0.171	1							
%Tolerant	-0.011	0.25	0.232	-0.229	-0.1	-0.197	-0.006	0.496	1						
%EPT	0.147	-0.463	-0.52	0.096	0.061	0.505	0.389	-0.222	-0.207	1					
%Collector-Filterer	-0.071	-0.152	-0.192	0.501	0.5	0.687	-0.094	0.086	-0.221	0.521	1				
%Scraper	0.26	-0.404	-0.369	-0.159	-0.308	-0.444	-0.157	-0.304	-0.14	-0.265	-0.385	1			
%Collector-Gatherer	0.542	-0.196	-0.145	-0.354	-0.429	-0.575	0.059	-0.368	-0.067	-0.078	-0.306	0.587	1		
%Predator	-0.266	-0.145	-0.144	0.66	0.505	0.929	-0.299	0.263	-0.201	0.362	0.646	-0.49	-0.615	1	
%Shredder	-0.261	-0.016	-0.008	-0.207	-0.079	-0.1	0.529	-0.012	0.306	0.075	-0.215	-0.151	-0.046	-0.173	1
# Dipteran Genera	0.056	0.307	0.533	-0.13	-0.125	0.005	-0.221	-0.032	0.23	-0.209	-0.012	-0.242	-0.108	0.037	0.032
# Ephem Families	0.031	-0.279	-0.281	-0.248	-0.302	-0.224	0.258	-0.129	-0.211	0.133	0.199	0.158	0.361	-0.233	-0.235
# Ephem. Genera	0.288	-0.339	-0.346	-0.213	-0.311	-0.154	0.215	-0.24	-0.084	0.358	0.201	0.175	0.48	-0.208	-0.25
# non-insect	-0.216	0.112	0.07	0.291	0.381	0.317	-0.456	0.54	0.401	-0.135	0.368	-0.13	-0.418	0.319	-0.025
# Plecop. Families	-0.202	-0.163	-0.18	-0.173	-0.257	0.038	0.161	0.109	0.034	0.17	-0.09	-0.095	-0.233	0.022	0.453
# Plecop. Genera	-0.202	-0.163	-0.18	-0.173	-0.257	0.038	0.161	0.109	0.034	0.17	-0.09	-0.095	-0.233	0.022	0.453
# Tricop. Genera	0.151	-0.175	-0.181	-0.424	-0.474	-0.176	0.362	-0.191	0.149	0.358	-0.257	0.125	0.198	-0.203	0.225
# Tricop. Families	0.144	-0.12	-0.149	-0.406	-0.411	-0.17	0.378	-0.259	0.092	0.394	-0.222	0.062	0.234	-0.202	0.27
Ave. Tolerance (FBI)	0.178	0.627	0.675	0.188	0.165	0.034	-0.809	0.346	0.376	-0.531	-0.015	-0.104	-0.141	0.175	-0.28
Chiro:EPT	-0.138	0.916	0.876	-0.029	0.187	-0.314	-0.271	0.131	0.31	-0.666	-0.278	-0.221	-0.122	-0.209	0.025
#EPT Families	0.054	-0.295	-0.326	-0.515	-0.578	-0.23	0.492	-0.23	-0.014	0.449	-0.118	0.089	0.274	-0.267	0.28
Family Richness	0.05	-0.117	-0.101	-0.363	-0.367	-0.107	0.096	-0.093	0.291	0.254	0.043	0.058	0.097	-0.18	0.207
Genera Richness	0.095	-0.241	-0.157	-0.409	-0.475	-0.179	0.127	-0.149	0.293	0.215	-0.001	0.158	0.229	-0.244	0.177
Simpson's Diversity	0.269	-0.226	-0.146	-0.779	-0.953	-0.569	0.228	0.018	0.184	0.002	-0.398	0.289	0.445	-0.516	0.075
Simpson's Evenness	0.332	-0.141	-0.099	-0.712	-0.886	-0.611	0.153	0.036	0.041	-0.121	-0.52	0.372	0.458	-0.533	-0.089
#EPT Genera	0.161	-0.363	-0.374	-0.444	-0.56	-0.197	0.457	-0.274	-0.004	0.482	-0.099	0.134	0.333	-0.255	0.176

Table 3.2.6 (continued). Pearson correlation coefficients of invertebrate metrics based on data from ANS Dam Project (GG2). Shaded areas represent correlation coefficients $0.7 or #-0.7.

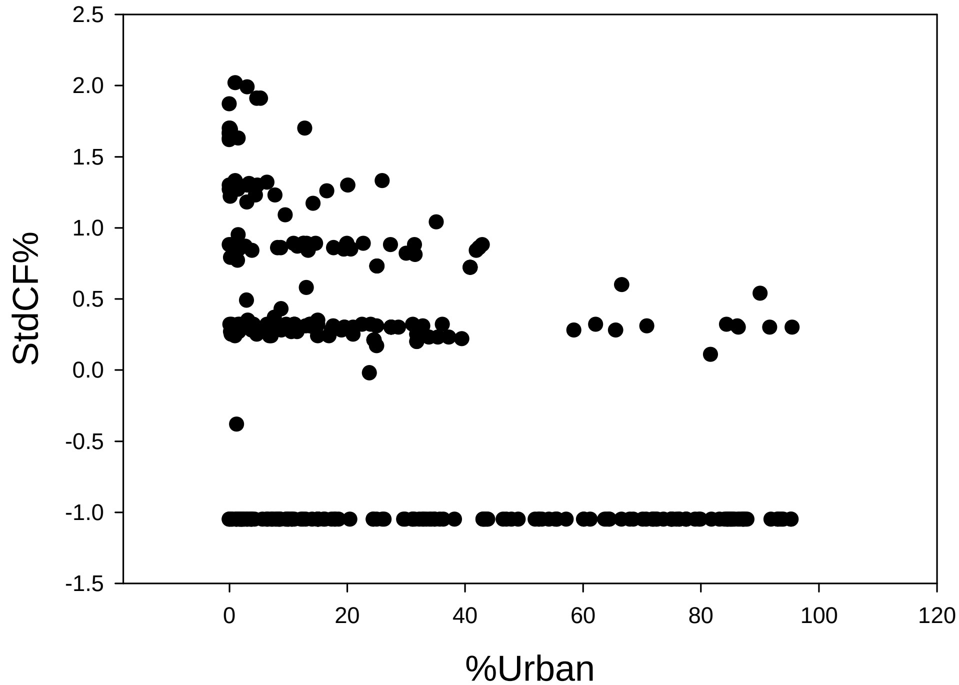
	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro: EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	-0.174	1														
# Ephem. Genera	-0.101	0.772	1													
# non-insect	0.164	-0.196	-0.203	1												
# Plecop. Families	-0.153	-0.093	-0.085	0.005	1											
# Plecop. Genera	-0.153	-0.093	-0.085	0.005	1	1										
# Tricop. Genera	0.01	0.017	0.216	-0.385	0.053	0.053	1									
# Tricop. Families	0.009	0.017	0.251	-0.345	-0.024	-0.024	0.918	1								
Ave. Tolerance (FBI)	0.375	-0.39	-0.344	0.493	-0.232	-0.232	-0.293	-0.294	1							
Chiro:EPT	0.273	-0.327	-0.389	0.137	-0.187	-0.187	-0.234	-0.198	0.65	1						
#EPT Families	-0.136	0.46	0.543	-0.372	0.322	0.322	0.772	0.808	-0.516	-0.39	1					
Family Richness	0.32	0.211	0.399	0.246	0.127	0.127	0.515	0.594	-0.063	-0.177	0.631	1				
Genera Richness	0.365	0.347	0.537	0.048	0.206	0.206	0.555	0.504	-0.143	-0.253	0.655	0.873	1			
Simpson's Diversity	0.093	0.422	0.39	-0.341	0.227	0.227	0.483	0.403	-0.25	-0.244	0.618	0.424	0.555	1		
Simpson's Evenness	-0.088	0.313	0.218	-0.444	0.08	0.08	0.363	0.3	-0.196	-0.144	0.424	0.104	0.186	0.886	1	
#EPT Genera	-0.124	0.441	0.647	-0.448	0.311	0.311	0.793	0.729	-0.524	-0.436	0.922	0.55	0.701	0.604	0.403	

Table 3.2.7. Pearson correlation coefficients of invertebrate metrics based on data from ANS Dam Project (GG3). Shaded areas represent correlation coefficients $0.7 or #-0.7.

	% Baetidae	% Chironomidae	% Diptera	% Dominant Familiy	% Dominant Genera	% Hydropsychidae	% Intolerant	% non-insect	% Tolerant	%EPT	%Collector-Filterer	% Scraper	%Collector-Gatherer	% Predator	% Shredder
%Baetidae	1														
%Chironomidae	-0.224	1													
%Diptera	-0.254	0.855	1												
% Dominant Familiy	-0.288	0.523	0.528	1											
% Dominant Genera	-0.297	0.557	0.566	0.93	1										
%Hydropsychidae	0.101	-0.086	-0.17	-0.071	-0.275	1									
%Intolerant	0.281	-0.277	-0.34	-0.608	-0.577	-0.074	1								
% non-insect	-0.473	0.14	0.184	0.542	0.577	-0.304	-0.458	1							
%Tolerant	-0.377	0.168	0.251	0.646	0.632	-0.29	-0.362	0.739	1						
%EPT	0.528	-0.424	-0.541	-0.717	-0.763	0.49	0.633	-0.761	-0.664	1					
%Collector-Filterer	0.22	0.155	0.115	0.043	-0.051	0.669	-0.194	-0.239	-0.068	0.357	1				
%Scraper	0.39	-0.651	-0.707	-0.461	-0.483	-0.07	0.214	-0.527	-0.448	0.474	-0.193	1			
%Collector-Gatherer	0.373	-0.467	-0.508	-0.178	-0.207	-0.065	0.018	-0.103	-0.376	0.154	-0.228	0.627	1		
%Predator	0.013	0.003	-0.094	-0.311	-0.403	0.721	0.111	-0.301	-0.382	0.481	0.478	-0.177	-0.314	1	
%Shredder	0.041	-0.228	-0.266	-0.251	-0.262	-0.02	0.681	-0.254	-0.145	0.365	-0.178	0.118	-0.078	-0.082	1
# Dipteran Genera	-0.13	0.287	0.328	-0.1	-0.104	0.125	-0.144	-0.047	-0.074	-0.072	0.304	-0.267	-0.108	0.129	-0.231
# Ephem Families	0.416	-0.169	-0.255	-0.631	-0.597	-0.043	0.651	-0.524	-0.57	0.653	-0.024	0.392	0.234	0.13	0.142
# Ephem. Genera	0.449	-0.148	-0.209	-0.593	-0.562	-0.059	0.603	-0.526	-0.579	0.651	0.022	0.367	0.202	0.142	0.156
# non-insect	-0.111	-0.186	-0.1	-0.307	-0.225	-0.026	-0.103	0.071	-0.034	-0.015	-0.104	0.021	-0.07	0.281	-0.25
# Plecop. Families	0.148	0.13	0.056	-0.313	-0.237	-0.176	0.644	-0.306	-0.242	0.322	-0.184	-0.066	-0.119	-0.016	0.491
# Plecop. Genera	0.148	0.13	0.056	-0.313	-0.237	-0.176	0.644	-0.306	-0.242	0.322	-0.184	-0.066	-0.119	-0.016	0.491
# Tricop. Genera	0.214	-0.21	-0.281	-0.726	-0.668	0.06	0.525	-0.454	-0.595	0.559	-0.128	0.193	0.073	0.294	0.245
# Tricop. Families	0.292	-0.148	-0.172	-0.662	-0.618	-0.063	0.467	-0.474	-0.538	0.494	-0.092	0.214	0.069	0.241	0.166
Ave. Tolerance (FBI)	-0.467	0.524	0.668	0.776	0.768	-0.201	-0.686	0.696	0.797	-0.826	0.052	-0.618	-0.438	-0.244	-0.446
Chiro:EPT	-0.2	0.717	0.752	0.569	0.593	-0.278	-0.289	0.145	0.213	-0.538	-0.158	-0.412	-0.271	-0.272	-0.14
#EPT Families	0.342	-0.065	-0.14	-0.644	-0.579	-0.12	0.724	-0.525	-0.539	0.59	-0.128	0.205	0.064	0.138	0.341
Family Richness	0.225	-0.201	-0.22	-0.805	-0.697	-0.108	0.599	-0.468	-0.552	0.537	-0.109	0.206	0.034	0.3	0.201
Genera Richness	0.225	-0.24	-0.276	-0.849	-0.745	-0.074	0.619	-0.491	-0.63	0.592	-0.12	0.252	0.107	0.268	0.212
Simpson's Diversity	0.29	-0.544	-0.548	-0.955	-0.976	0.178	0.595	-0.58	-0.68	0.753	0.006	0.496	0.252	0.375	0.239
Simpson's Evenness	0.275	-0.433	-0.499	-0.393	-0.586	0.293	0.367	-0.165	0.003	0.464	0.041	0.221	-0.022	0.272	0.1
#EPT Genera	0.297	-0.093	-0.18	-0.683	-0.61	-0.075	0.717	-0.523	-0.593	0.623	-0.129	0.204	0.054	0.186	0.355

Table 3.2.7 (continued). Pearson correlation coefficients of invertebrate metrics based on data from ANS Dam Project (GG3). Shaded areas represent correlation coefficients $0.7 or #-0.7.

	# Dipteran Genera	# Ephem. Families	# Ephem. Genera	# non-insect	# Plecop. Families	# Plecop. Genera	# Tricop. Genera	# Tricop. Families	Ave. Tolerance (FBI)	Chiro: EPT	#EPT Families	Family Richness	Genera Richness	Simpson's Diversity	Simpson's Evenness	#EPT Genera
# Dipteran Genera	1															
# Ephem Families	0.084	1														
# Ephem. Genera	0.076	0.96	1													
# non-insect	-0.026	-0.048	-0.099	1												
# Plecop. Families	0.05	0.364	0.355	-0.24	1											
# Plecop. Genera	0.05	0.364	0.355	-0.24	1	1										
# Tricop. Genera	0.026	0.502	0.451	0.134	0.601	0.601	1									
# Tricop. Families	0.082	0.561	0.533	0.158	0.566	0.566	0.921	1								
Ave. Tolerance (FBI)	0.084	-0.623	-0.606	0.059	-0.388	-0.388	-0.658	-0.559	1							
Chiro:EPT	-0.038	-0.281	-0.277	0.014	-0.131	-0.131	-0.336	-0.251	0.544	1						
#EPT Families	0.087	0.769	0.739	-0.065	0.812	0.812	0.825	0.863	-0.634	-0.266	1					
Family Richness	0.229	0.62	0.571	0.368	0.595	0.595	0.835	0.866	-0.612	-0.326	0.846	1				
Genera Richness	0.257	0.689	0.657	0.236	0.588	0.588	0.882	0.865	-0.699	-0.384	0.869	0.961	1			
Simpson's Diversity	0.141	0.667	0.642	0.223	0.318	0.318	0.718	0.682	-0.797	-0.614	0.668	0.783	0.833	1		
Simpson's Evenness	-0.188	0.321	0.29	0.085	-0.03	-0.03	0.207	0.193	-0.315	-0.416	0.186	0.15	0.172	0.469	1	
#EPT Genera	0.069	0.759	0.746	-0.079	0.792	0.792	0.866	0.851	-0.68	-0.31	0.983	0.836	0.892	0.699	0.181	



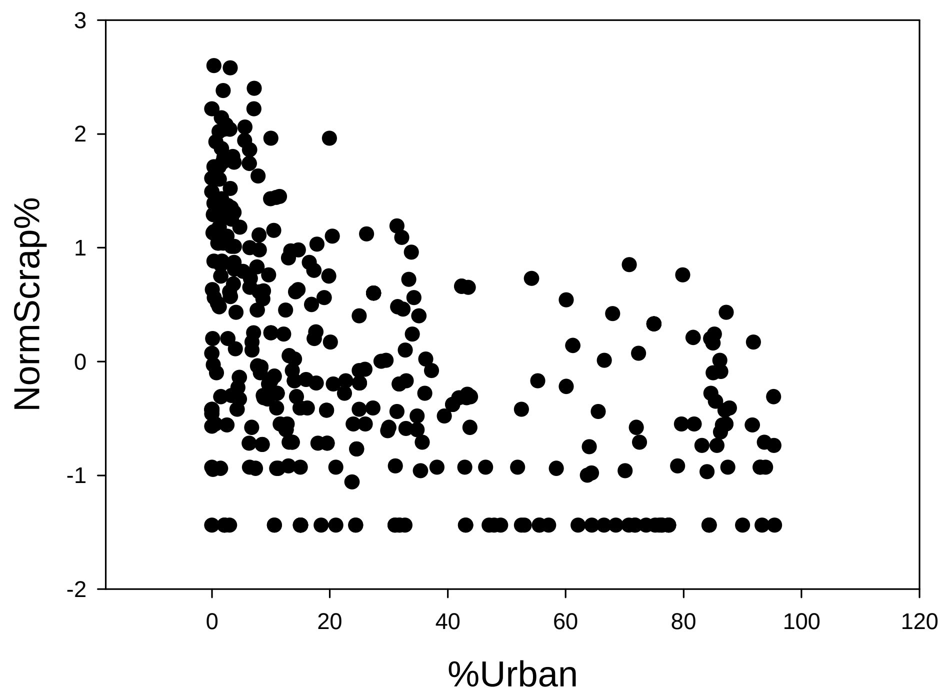


Figure 3.2.6. Relationship between % urban and normalized, transformed %CF (percentage collector-filterer macroinvertebrates) (top) and %Scrap (percentage scraper macroinvertebrates) (bottom) for joint fish-macroinvertebrate sites.

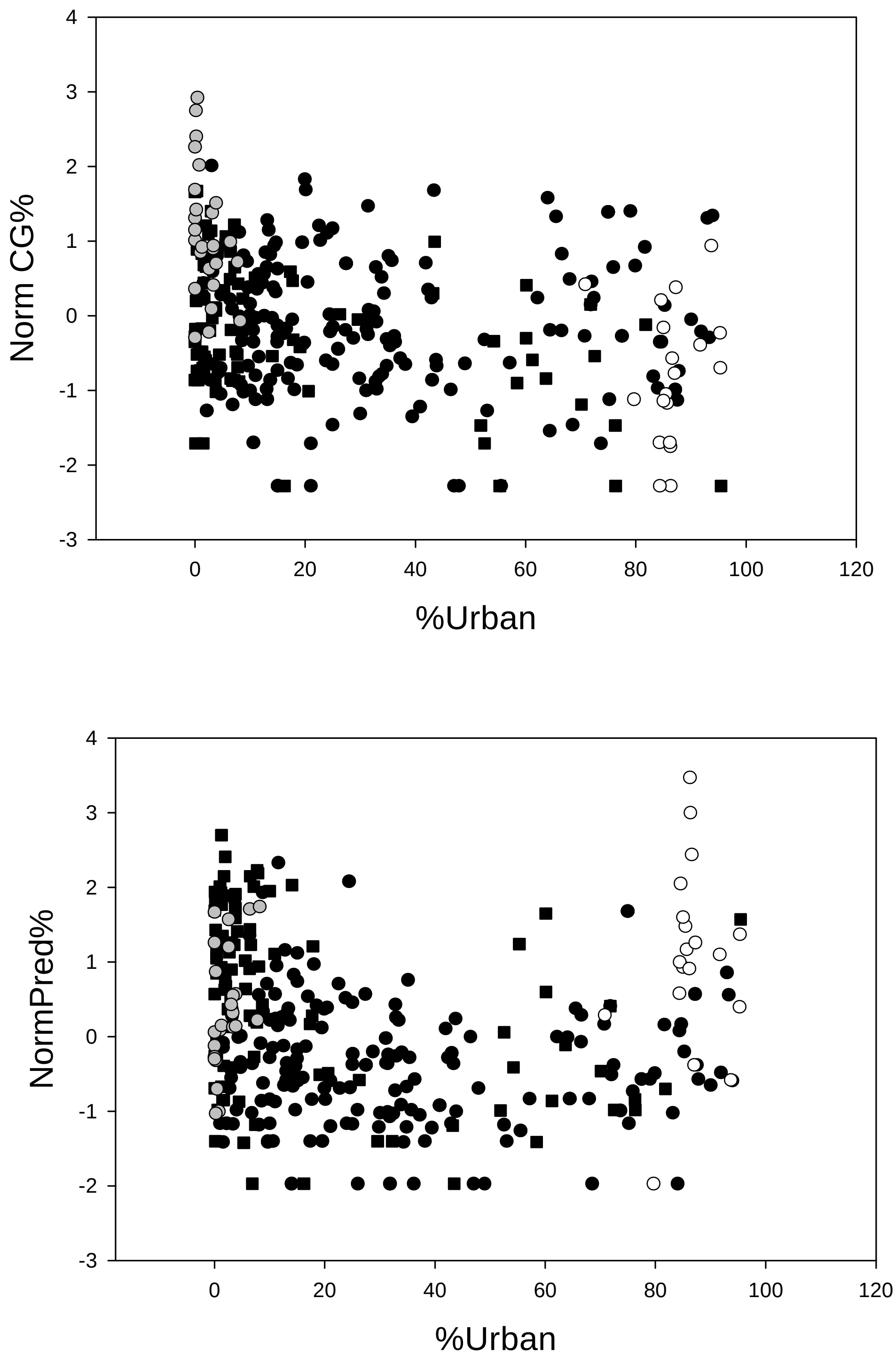


Figure 3.2.7. Relationship between % urban and normalized, transformed %CG (percentage collector-gatherer macroinvertebrates) (top) and %Pred (percentage predator macroinvertebrates) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

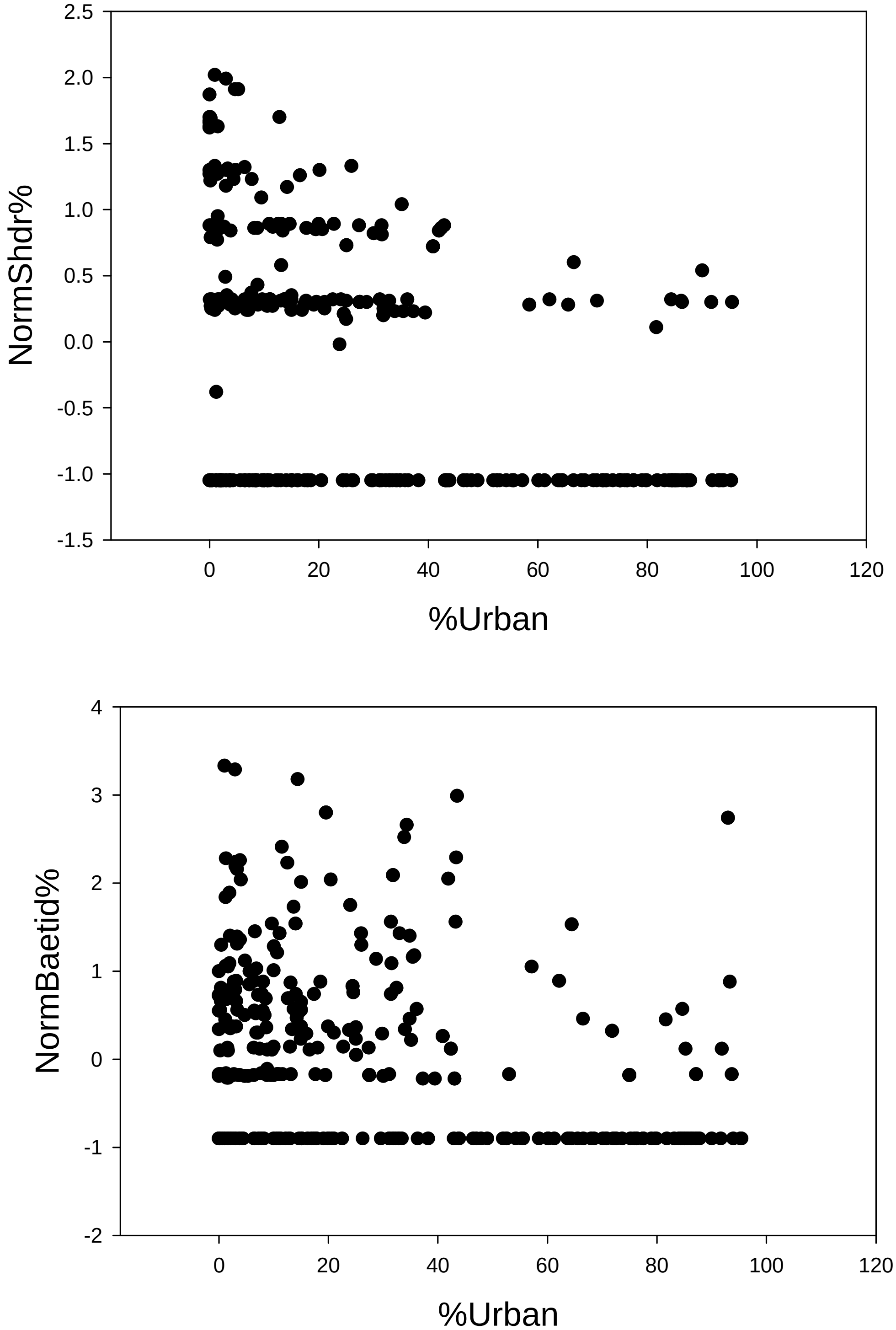
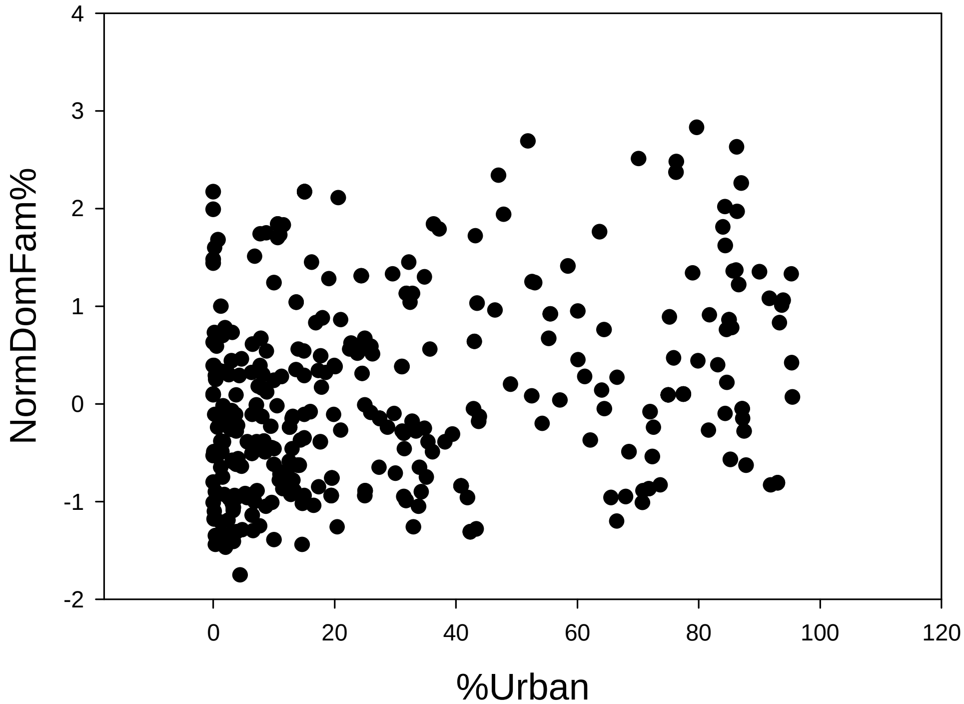


Figure 3.2.8. Relationship between % urban and normalized, transformed %Shdr (percentage shredder macroinvertebrates) (top) and %baetid (percentage baetid mayflies) (bottom) for joint fish-macroinvertebrate sites.



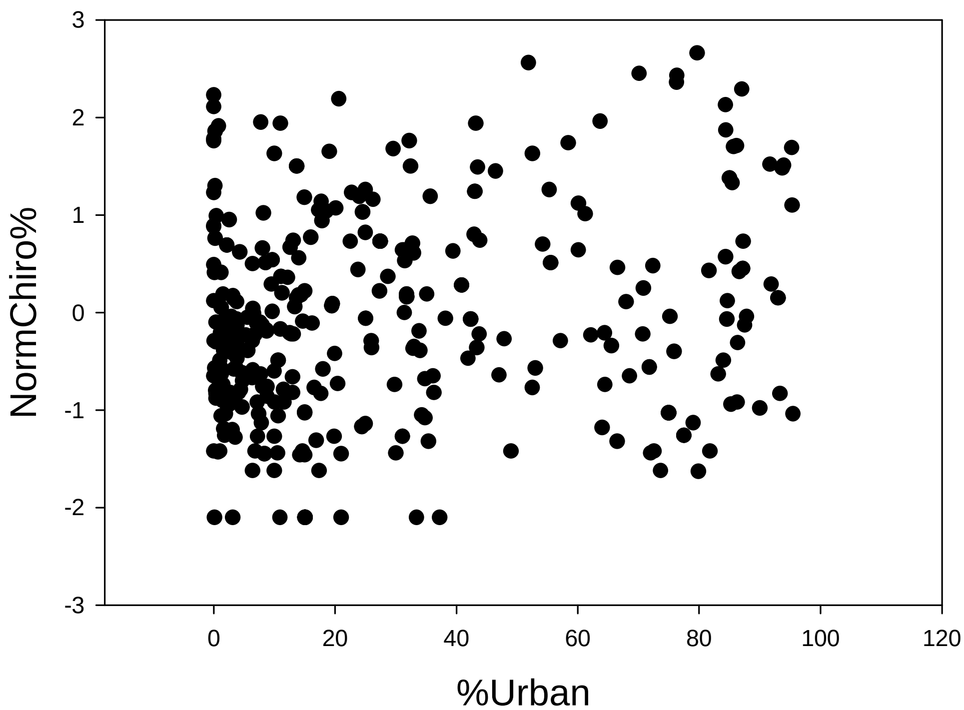
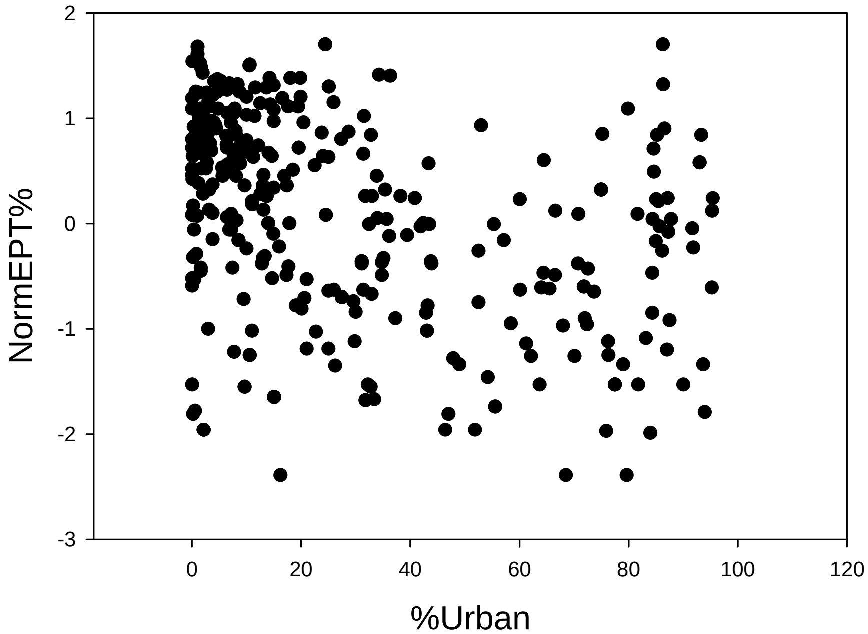


Figure 3.2.9. Relationship between % urban and normalized, transformed %DomFam (percentage macroinvertebrate dominant family) (top) and %Chiro (percentage chironomid midge) (bottom) for joint fish-macroinvertebrate sites.

2D Graph 14



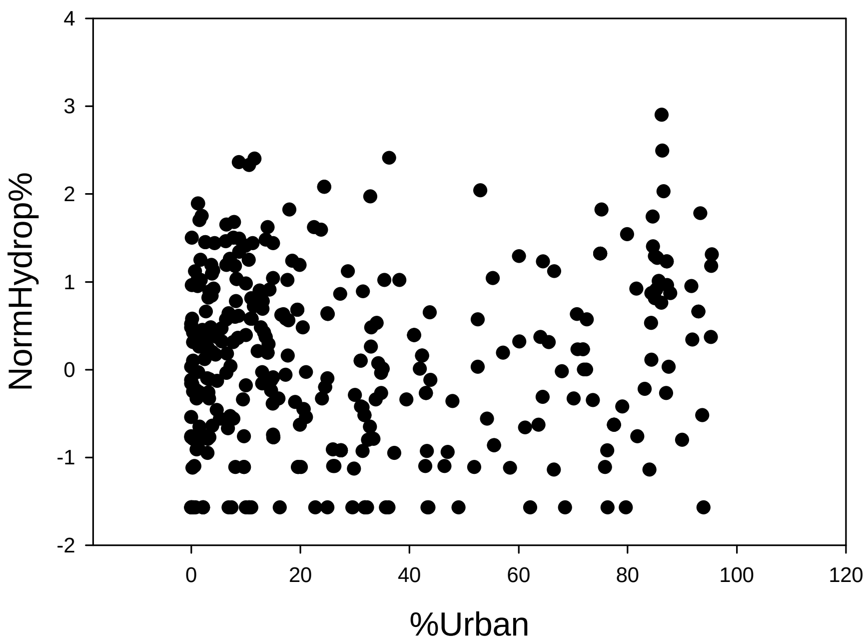
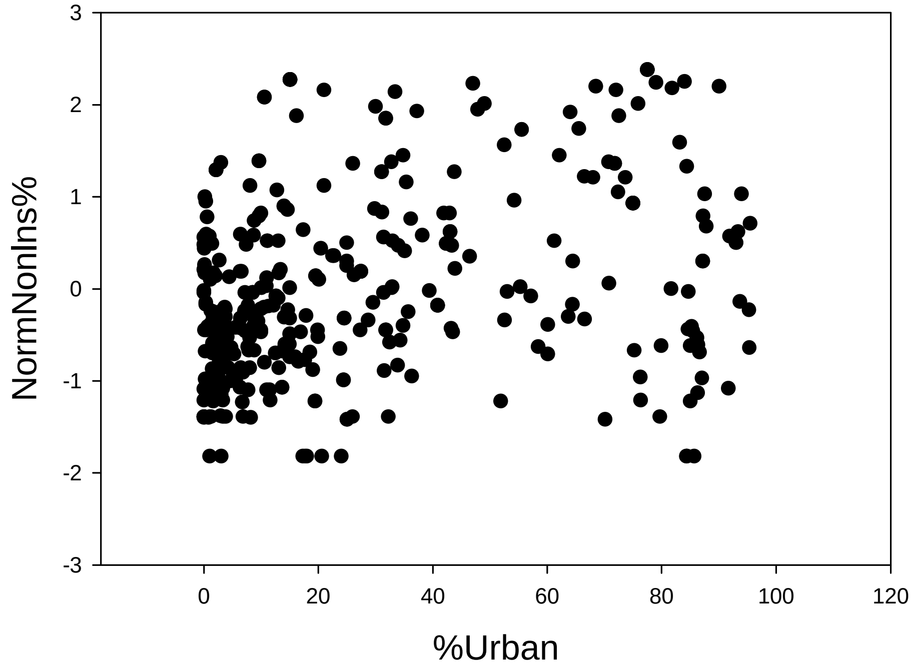


Figure 3.2.10. Relationship between % urban and normalized, transformed %EPT (percentage EPT macroinvertebrates) (top) and %Hydrop (percentage hydropsychid caddisflies) (bottom) for joint fish-macroinvertebrate sites.



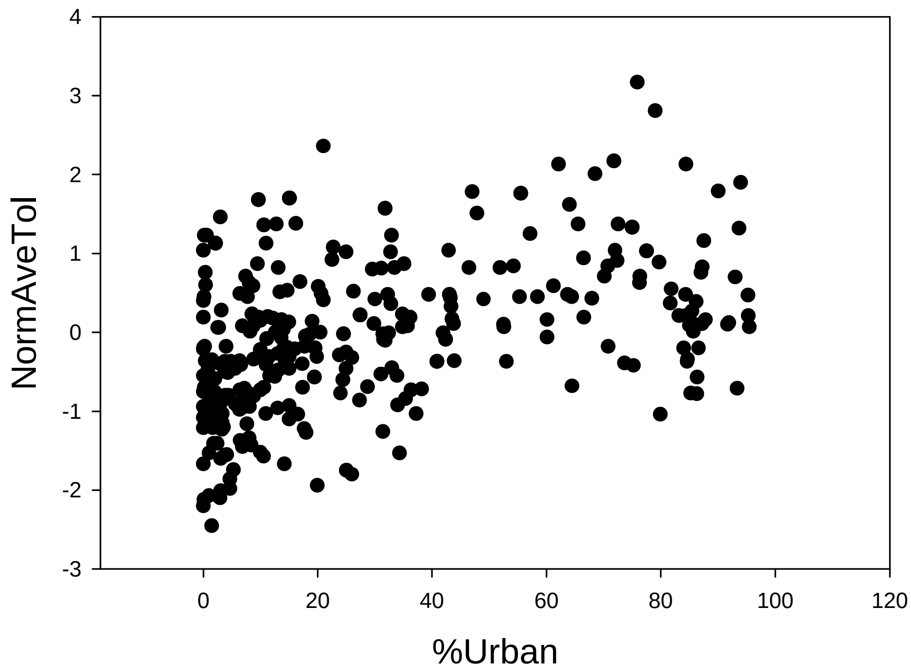
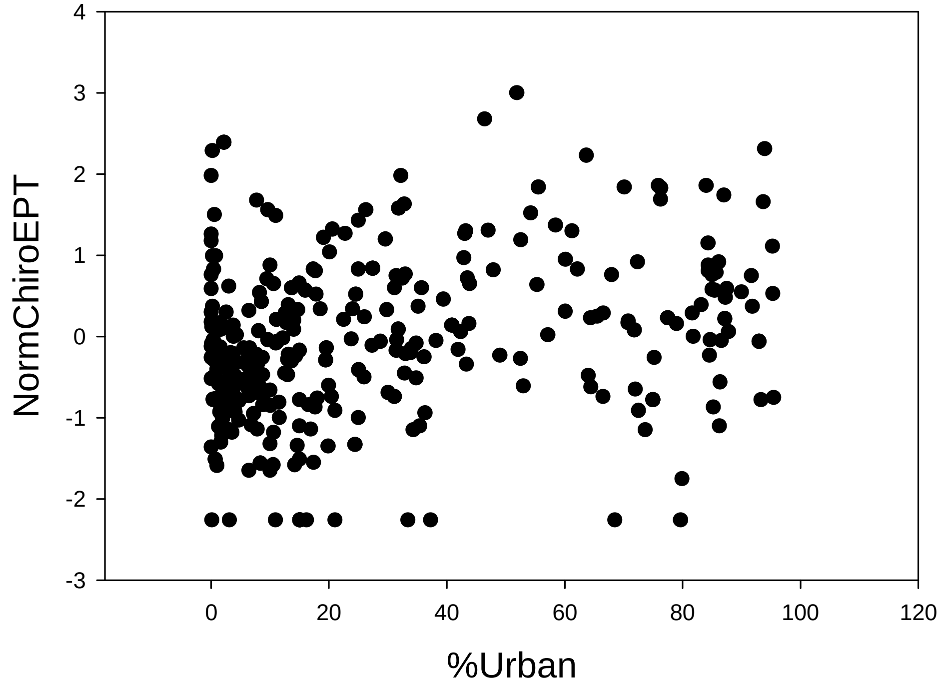


Figure 3.2.11. Relationship between % urban and normalized, transformed %NonIns (percentage non-insect macroinvertebrates) (top) and AveTol (average tolerance of macroinvertebrates) (bottom) for joint fish-macroinvertebrate sites.



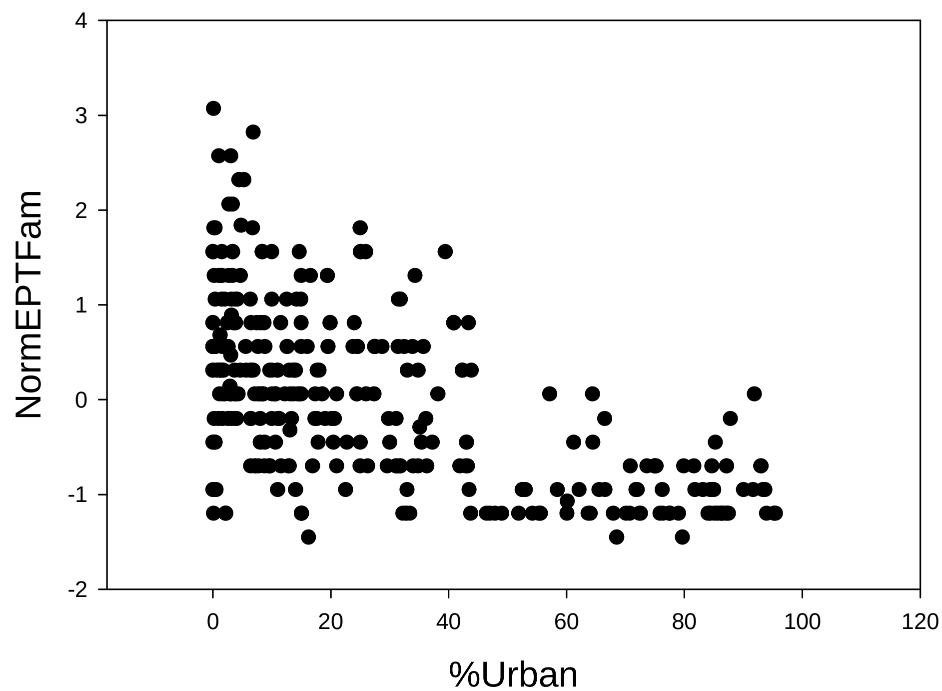
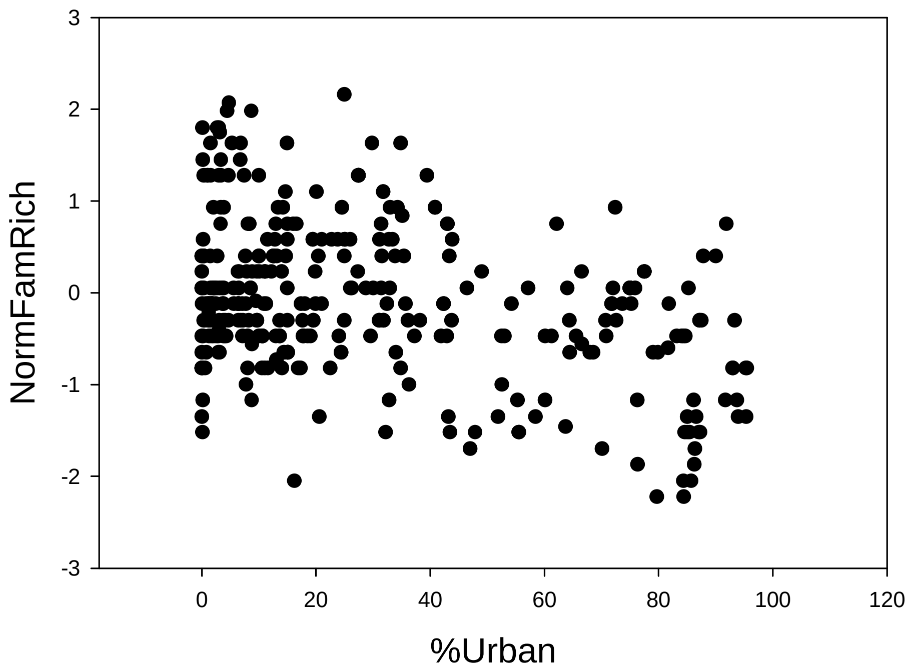


Figure 3.2.12. Relationship between % urban and normalized, transformed %ChiroEPT (ratio chironomid to EPT) (top) and EPTFamRich (family richness of EPTs) (bottom) for joint fish-macroinvertebrate sites.



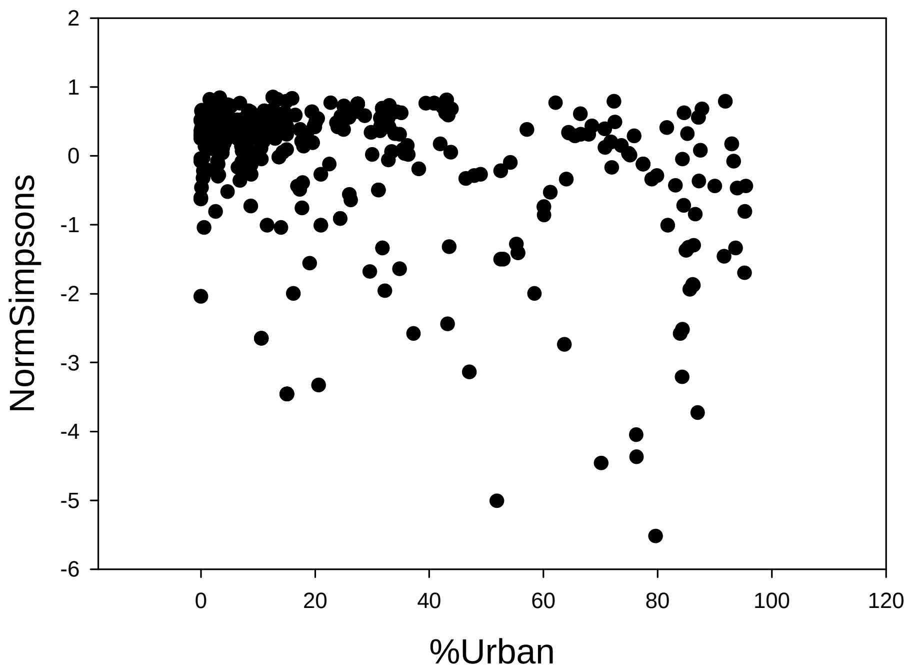
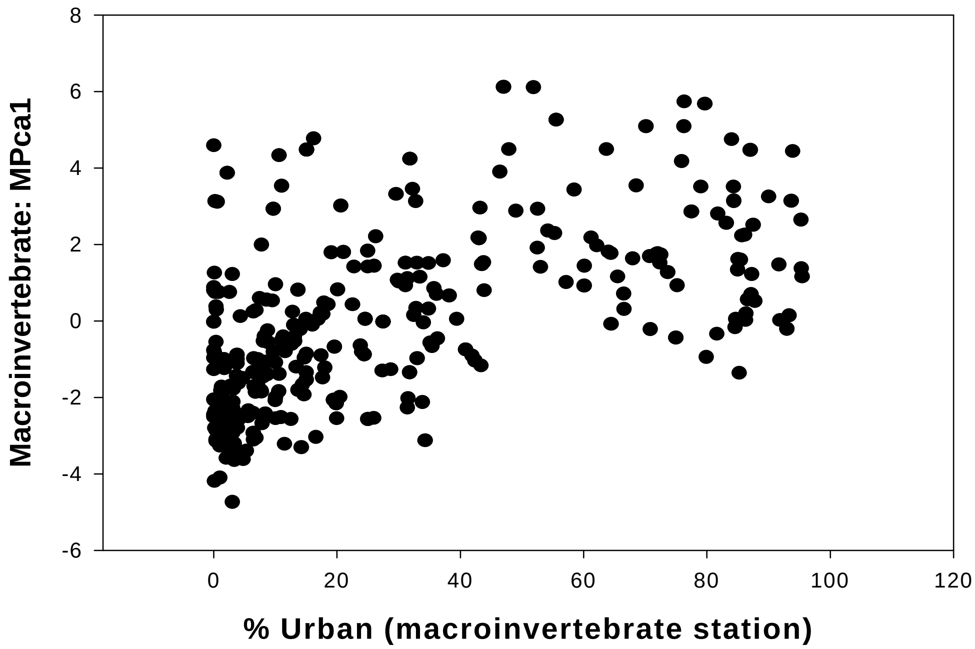


Figure 3.2.13. Relationship between % urban and normalized, transformed %FamRich (family richness of macroinvertebrates) (top) and Simpsons (Simpsons diversity index of macroinvertebrates) (bottom) for joint fish-macroinvertebrate sites.

Table 3.2.8. Correlations of metrics with the first 9 principal components in PCA of variation over 520 macroinvertebrate samples. The cumulative % variance associated with each component is presented under each title. Correlations greater than 0.5 are bolded for emphasis.

	MFpca1	**MFpca2**	**MFpca3**	**MFpca4**	**MFpca5**	**MFpca6**	**MFpca7**	**MFpca8**	**MFpca9**
	32.3	**48.9**	**61.8**	**70.1**	**76.4**	**81.9**	**85.9**	**89.2**	**91.5**
Metric									
NrmspCF	-0.35	**-0.61**	0.23	0.38	0.10	0.04	-0.19	-0.33	0.34
NSQpScrap	**-0.69**	-0.12	0.14	0.18	0.41	0.16	-0.39	0.09	-0.08
NSQpC-G	-0.39	0.21	0.45	0.43	0.45	0.09	0.30	0.19	-0.06
NSQpPreds	-0.36	**-0.54**	-0.09	**0.54**	-0.29	-0.01	0.01	-0.11	-0.40
NSQpSHRD	-0.39	0.35	0.41	0.11	-0.31	0.50	0.31	-0.18	0.09
NSQpBaetid	**-0.54**	0.10	0.15	-0.20	0.33	**-0.57**	0.25	-0.33	-0.07
NSQpChiro	0.41	-0.13	**0.79**	0.01	-0.21	-0.23	-0.04	0.13	-0.02
NSQpDomFam	**0.65**	-0.48	0.22	-0.25	0.08	0.19	0.19	-0.09	-0.09
NSQpEPT	**-0.84**	-0.35	-0.07	-0.21	-0.12	-0.08	0.11	0.09	0.08
NSQpHydrop	-0.36	**-0.67**	-0.37	0.12	-0.28	-0.17	0.19	0.11	0.11
NSQpNonins	0.47	0.46	**-0.58**	0.29	0.04	-0.03	-0.05	-0.12	0.05
NAveTol	**0.75**	0.28	-0.04	0.40	0.02	-0.11	0.15	-0.17	0.03
NLChiroEPT	**0.54**	0.09	**0.61**	0.16	-0.25	-0.29	-0.20	0.05	0.04
NEPTFams	**-0.78**	0.27	0.22	-0.35	-0.16	0.05	-0.09	-0.10	-0.03
NFamRich	**-0.57**	**0.63**	0.00	0.03	-0.28	-0.05	-0.19	-0.19	-0.11
NSimpson	**-0.63**	0.46	-0.10	0.32	-0.16	-0.22	0.10	0.28	0.15

Joint fish-macroinvertebrate samples



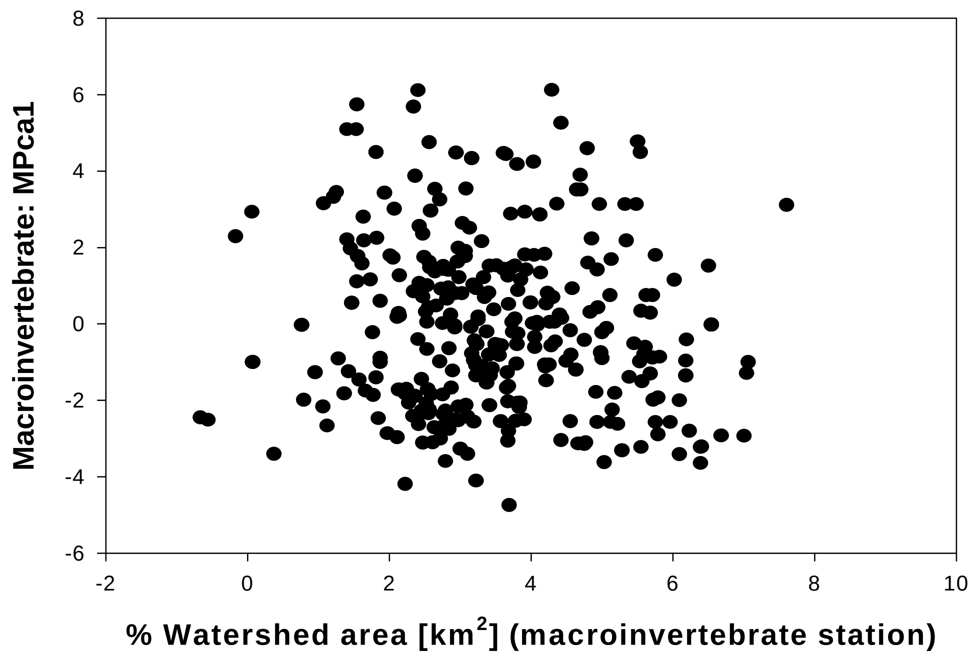
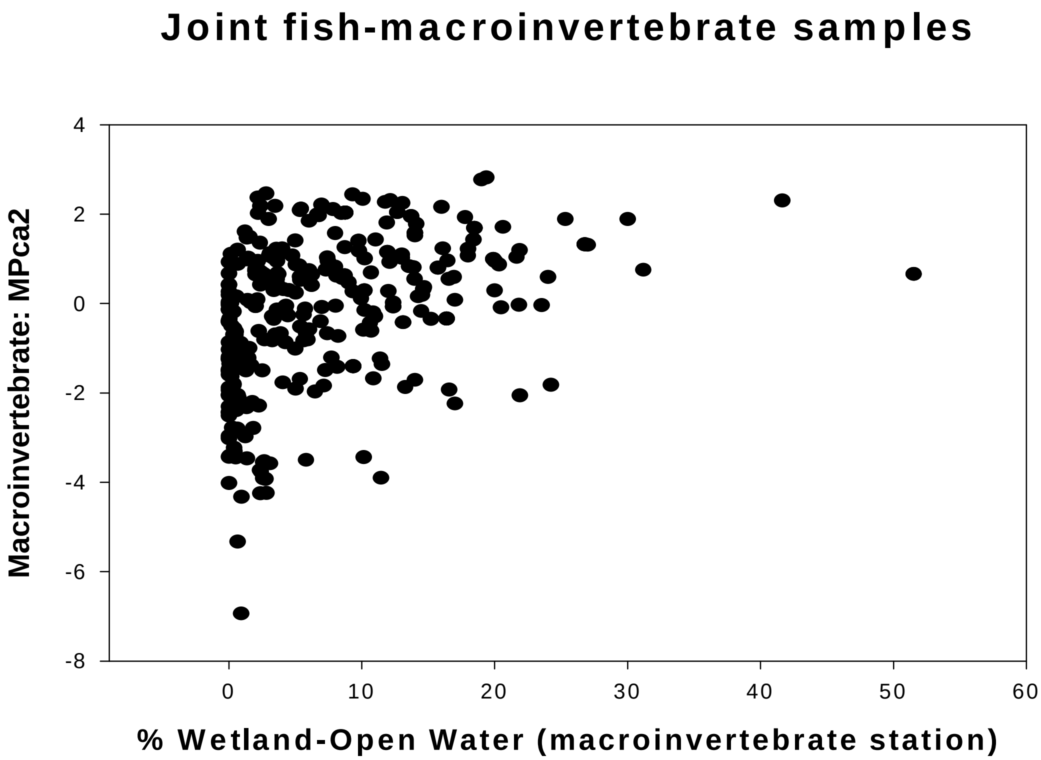


Figure 3.2.14. Relationship between mpca1 (first principal component for macroinvertebrates) and % urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).



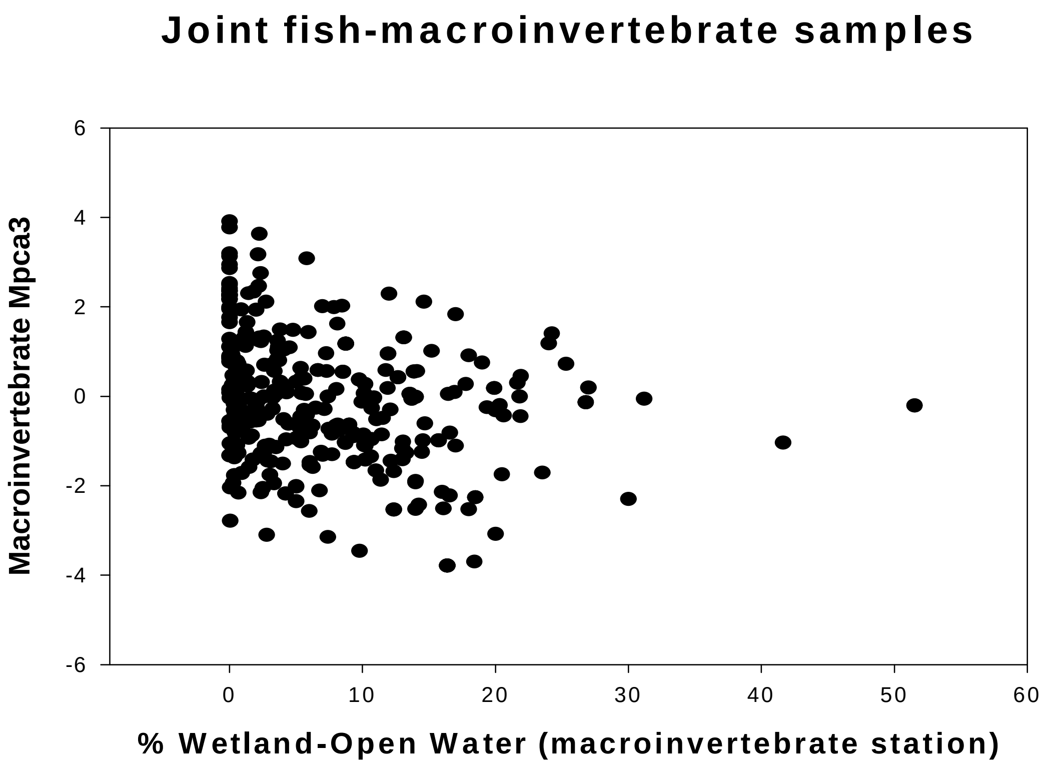
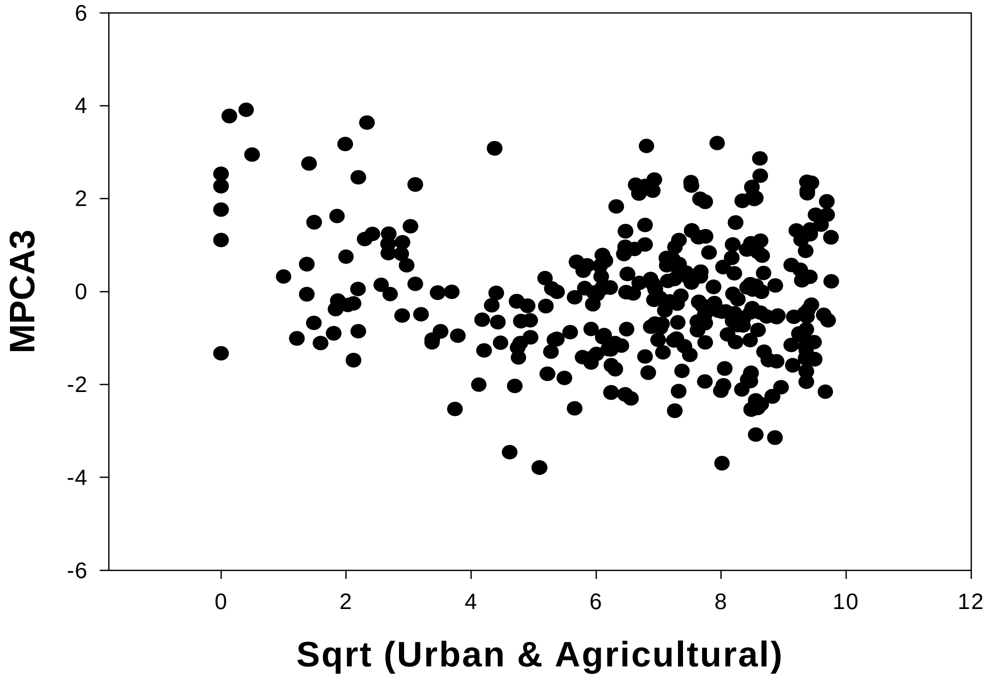


Figure 3.2.15. Relationship between %wetland and mpca2 (second principal component for macroinvertebrates) (top) and mpca3 (bottom) for joint fish-macroinvertebrate sites.



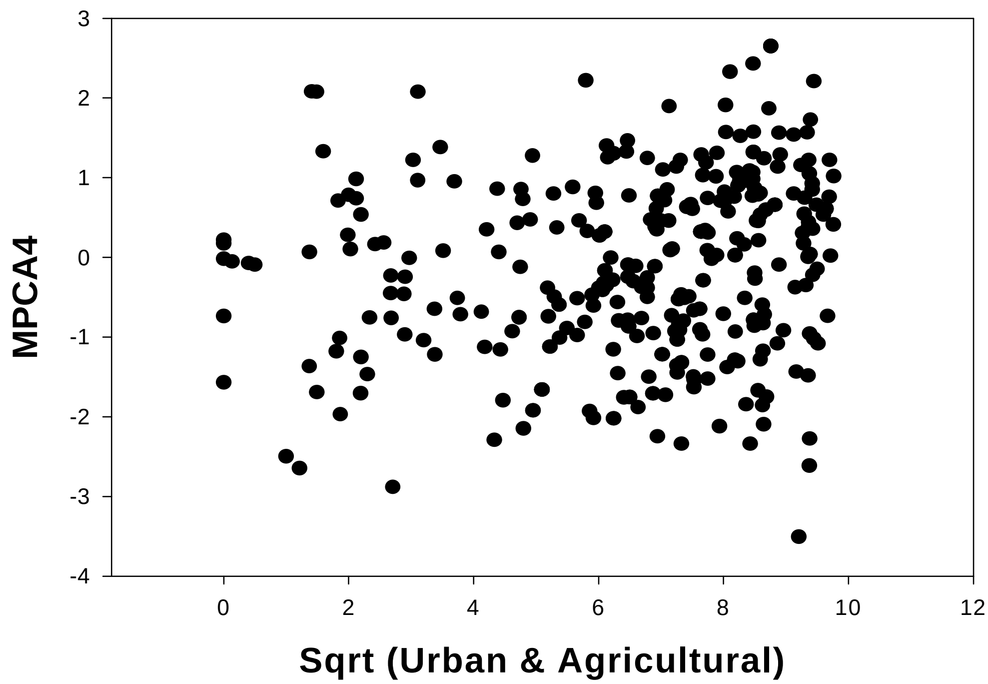


Figure 3.2.16. Relationship between square root of developed area (%urban and %agriculture) and Mpca3 (third macroinvertebrate principal component) (top) and Mpca4 (fourth macronvertebrate principal component) (bottom) for joint fish-macroinvertebrate sites.

3.3 Fish Assemblages and Metrics

3.3.1 Relationship Between Fish Species Occurrence, Watershed Size and Urbanization

The purpose of fish metrics is to subsume attributes of various species into aggregate measures which reflect community-level ecological responses. Understanding patterns of species occurrence is useful in interpreting patterns of fish metrics. Fish species distribution was related to stream size classes and urbanization classes. Six stream size classes were defined on the basis of ln(watershed size) (Table 3.3.1). The first two classes include small streams, which are below the threshold for the NJ FIBI. Five urbanization classes were defined on the basis of % urbanization of the watershed above the sampling site (Table 3.3.1). These classes and fish abundance were compiled for 438 sites, which included 24 of the 30 possible urbanization-stream size classes. There were no large streams for the most urban classes (Table 3.3.1), since undeveloped headwaters of large streams lead to intermediate overall urbanization for the largest streams. Average proportions of species were calculated for 438 sampling sites, using first pass data only (Tables 3.3.2-3.3.6). In addition, catch per unit effort was calculated as the number of fish caught (using first pass data only) per 100 m of shoreline sampled. Since catch rates are apt to be lognormally distributed, the geometric mean catch rate over all samples at a station was used as the index of abundance. Graphs of these average catch rates are presented in Appendix A (Figures A.1.1-A.1.23). These two measures (proportions and catch rates) are complementary. Changes in relative abundance may reflect increases in abundance of a species or decreases in abundance of other species; catch rates can aid in interpreting such differences. The catch rates were not adjusted for differences in stream width (since width data were not uniformly available). As a result, sites with similar linear densities may differ greatly in areal densities, with lower areal densities in streams in larger watersheds and, to a lesser extent, in more urban streams.

The average proportions of species in different urbanization and stream size classes demonstrate several patterns of distribution and response to urbanization (Tables 3.3.2-3.3.6). Several species (Table 3.3.2) may be considered headwater-intolerant species, since these species are most common in small streams and are rare or absent in urban streams. This group includes two coldwater species, the brook trout and slimy sculpin. In addition, it includes two other species of sculpins, the least brook lamprey and a minnow, the rosyside dace. These four species are not found in northern New Jersey, and were represented in the database from streams in Pennsylvania, Delaware and Maryland.

A second group of species (Table 3.3.3) also show decreases in relative abundance with urbanization, but are not headwater species, i.e., they are more common in larger streams or do not show a clear stream size pattern. This group includes some piscivorous species (smallmouth bass, chain pickerel and walleye), introduced trout (rainbow and brown trout), two species of lampreys, several species commonly found in riffles and runs (margined madtom, shield darter and Northern hogsucker), several minnows (fallfish, bluntnose minnow and cutlips minnow) and the bluespotted sunfish. These species show a range of tolerance, with some species absent from urban streams, while others (e.g., fallfish, smallmouth bass, cutlips minnow, and the trouts) are found in some very urban streams. The occurrence of trout probably reflects survival of stocked individuals. These species also show differences in stream size occurrence. Some of these are found mainly in small streams (e.g., American brook lamprey), while others occur most common in large streams.

In contrast to the first two groups, a number of species increase in relative frequency with urbanization (Table 3.3.4). Two of these (blacknose dace and creek chub) are common in nonurban small streams, but become even more prevalent in urban streams. Others occur rarely in very small, nonurban streams, but expand into small streams with increasing urbanization. These include species (e.g., white sucker and tessellated darter) which are common in undeveloped small streams, and whose relative abundance increases with urbanization as other species decline. Others (e.g., common carp, banded killifish, green sunfish and redbreast sunfish) are rare or absent in undeveloped small streams, but increase with urbanization. The headwater expansion into urban streams is seen mainly in the smallest and most urban streams. The satinfin shiner shows a somewhat different pattern. In undeveloped streams, it is found in a range of stream sizes but at low relative abundance. It becomes common in some of the most urban streams, including both small and moderate-sized streams. The abundance in larger streams mainly reflects abundance in some urban streams in Philadelphia. Although very urban, these streams, located near the Fall Line in parks, have relatively high gradient which may improve habitat and water quality.

A few species (Table 3.3.5) increase in relative abundance with urbanization, but without any clear pattern with respect to stream size. These include the pumpkinseed and bluegill, which are widespread in urban streams. The mummichog is very common in a few very urban streams. The Eastern mudminnow is more common in streams of intermediate urbanization, but absent from most urban streams. The Eastern silvery minnow and spottail shiner are rare or absent in nonurban streams, but increases in relative abundance in larger urban streams. These two species are common in large rivers (e.g., the Delaware River) which are not represented in the dataset, and their occurrence in the wadeable stream represents upstream expansion from these large rivers into large streams.

A number of species (Table 3.3.6) show no clear relationship with urbanization or stream size. Many of these (e.g., brown bullhead, black crappie, rock bass, golden shiner, yellow perch, and creek chubsucker) are common in large rivers as well as in lakes, ponds and impoundments. Their abundance in smaller streams probably reflects nearby lentic source habitats, which can be independent of stream size and urbanization. A number of species (Table 3.3.6) which were rare in the sample sites, showed no clear pattern with urbanization or stream size.

Table 3.3.1. Definitions of watershed size and urbanization classes and numbers of samples in different classes. The existing NJ FIBI is only defined for watersheds in size classes 2 through 5.

Watershed Area Classes							
	Class	0	1	2	3	4	5
	ln(watershed area)						
	lower threshold	0.00	1.26	2.56	3.75	5.00	>6.25
	upper threshold	1.26	2.60	3.75	4.99	6.25	
	Watershed area (km^2)						
	lower threshold	0	3.5	12.9	42.5	148.4	518.0
	upper threshold	3.5	13.5	42.5	146.9	517.5	
Urban Classes							
	Urban class	0	1	2	3	4	
	% Urban area	<20	20-40	40-60	60-80	>80	
Number of samples (full database, first pass only)							
		Urbancode					
	Areacode	0	1	2	3	4	Total
	0	14	3	2		2	21
	1	48	9	3	15*	4	64
	2	81	27	13	19	19	159
	3	57	21	4	8	10	100
	4	53	14	3			70
	5	9	1				10
		262	75	25	27	35	424

* 1 without passlength

Table 3.3.2. Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which are most common in small streams (watershed size classes 0 or 1) and which show decreases in relative abundance with increasing urbanization. Gray cells are those for which no samples exist.

	Watershed size group				
	0	1	2	3	4
Brook trout (*Salvelinus fontinalis*)					
0	**37.883%**	**0.370%**	0.000%		0.000%
1	**7.694%**	0.000%	0.000%	0.000%	0.000%
2	**1.926%**	**0.409%**	0.000%	**0.007%**	0.000%
3	**1.903%**	**1.593%**	0.000%	0.000%	0.000%
4	**0.666%**	**0.192%**	0.000%	0.000%	
5	0.000%	0.000%			
Rosyside dace (*Clinostomus funduloiodes*)					
0	**2.666%**	0.000%	0.000%		0.000%
1	**5.568%**	0.000%	0.000%	0.000%	0.000%
2	**1.067%**	0.000%	0.000%	0.000%	0.000%
3	**0.262%**	0.000%	0.000%	0.000%	0.000%
4	0.000%	0.000%	0.000%		
5	0.000%	0.000%			
Slimy sculpin (*Cottus cognatus*)					
0	**2.158%**	0.000%	0.000%		0.000%
1	0.000%	0.000%	0.000%	0.000%	0.000%
2	**0.035%**	**0.249%**	0.000%	0.000%	0.000%
3	**0.459%**	**1.517%**	0.000%	0.000%	0.000%
4	0.000%	0.000%	0.000%		
5	0.000%	0.000%			
Blue Ridge sculpin (*Cottus caerulomontanum*)					
0	**2.126%**	0.000%	0.000%		0.000%
1	**2.833%**	0.000%	0.000%	0.000%	0.000%
2	0.000%	0.000%	0.000%	0.000%	0.000%
3	0.000%	0.000%	0.000%	0.000%	0.000%
4	0.000%	0.000%	0.000%		
5	**4.514%**	0.000%			
Least brook lamprey (*Lampetra aepyptera*)					
0	**0.031%**	0.000%	0.000%		0.000%
1	**0.086%**	0.000%	0.000%	0.000%	0.000%
2	0.000%	0.000%	0.000%	0.000%	0.000%
3	0.000%	0.000%	0.000%	0.000%	0.000%
4	0.000%	0.000%	0.000%		
5	0.000%	0.000%			

Table 3.3.3a. Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which are not headwater specialists and show decreases in relative abundance with increasing urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Sea lamprey (*Petromyzon marinus*)					
0	**0.056%**	0.000%	0.000%		0.000%
1	**0.121%**	0.000%	0.000%	0.000%	0.000%
2	**0.453%**	0.000%	0.000%	0.000%	0.000%
3	**0.251%**	**0.076%**	0.000%	0.000%	0.000%
4	**0.194%**	0.000%	0.000%		
5	**0.387%**	0.000%			
Cutlips minnow (*Exoglossum maxillingua*)					
0	**0.031%**	0.000%	0.000%		0.000%
1	**2.409%**	**0.240%**	0.000%	0.000%	0.000%
2	**2.490%**	0.000%	0.000%	**0.014%**	0.000%
3	**5.888%**	**0.406%**	0.000%	0.000%	**0.108%**
4	**4.888%**	**1.029%**	0.000%		
5	**10.216%**	0.000%			
Bluntnose minnow (*Pimephales notatus*)					
0	0.000%	0.000%	0.000%		0.000%
1	**1.450%**	**0.438%**	**0.104%**	**0.100%**	0.000%
2	**0.063%**	0.000%	0.000%	0.000%	0.000%
3	**0.258%**	0.000%	0.000%	0.000%	0.000%
4	**0.101%**	0.000%	0.000%		
5	**0.267%**	0.000%			
Northern hogsucker (*Hypentelium nigricans*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.060%**	0.000%	0.000%	0.000%	0.000%
2	**0.022%**	0.000%	0.000%	0.000%	0.000%
3	0.000%	0.000%	0.000%	0.000%	0.000%
4	**0.042%**	0.000%	0.000%		
5	0.000%	0.000%			
Shield darter (*Percina peltata*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.172%**	0.000%	0.000%	0.000%	0.000%
2	**0.052%**	0.000%	0.000%	0.000%	0.000%
3	**0.223%**	**0.310%**	0.000%	0.000%	0.000%
4	**1.142%**	**1.008%**	0.000%		
5	**2.882%**	0.000%			
Bluespotted sunfish (*Enneacanthus gloriosus*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.080%**	0.000%	0.000%	0.000%	0.000%
2	**0.063%**	**0.128%**	**0.210%**	**0.021%**	0.000%
3	**0.051%**	**0.101%**	0.000%	0.000%	0.000%
4	**0.040%**	0.000%	0.000%		
5	0.000%	0.000%			

Table 3.3.3a.(Continued) Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which are not headwater specialists and show decreases in relative abundance with increasing urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Walleye (*Zander vitreum*)					
0	0.000%	0.000%	0.000%		0.000%
1	0.000%	0.000%	0.000%	0.000%	0.000%
2	**0.014%**	0.000%	0.000%	0.000%	0.000%
3	0.000%	0.000%	0.000%	0.000%	0.000%
4	**0.027%**	**0.078%**	0.000%		
5	**0.050%**	0.000%			
American brook lamprey (*Lampetra appendix*)					
0	**0.221%**	0.000%	0.000%		0.000%
1	0.000%	**0.126%**	0.000%	0.000%	0.000%
2	**0.022%**	0.000%	0.000%	0.000%	0.000%
3	**0.779%**	**0.694%**	0.000%	0.000%	0.000%
4	0.000%	**0.390%**	0.000%		
5	0.000%	0.000%			
Chain pickerel (*Esox niger*)					
0	**0.056%**	0.000%	0.000%		0.000%
1	**1.224%**	**1.111%**	0.000%	0.000%	0.000%
2	**0.923%**	**0.730%**	**1.073%**	0.000%	0.000%
3	**0.170%**	**0.149%**	0.000%	0.000%	0.000%
4	**0.344%**	**0.040%**	0.000%		
5	**0.364%**	0.000%			
Margined madtom (*Noturus insignis*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.081%**	0.000%	0.000%	0.000%	0.000%
2	**0.389%**	**0.671%**	**0.030%**	**0.028%**	0.000%
3	**0.973%**	**0.693%**	0.000%	0.000%	0.000%
4	**0.522%**	**1.544%**	0.000%		
5	**1.741%**	0.000%			

Table 3.3.3b Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which are not headwater specialists and show decreases in relative abundance with increasing urbanization, although they may occur in urban streams. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Fallfish (*Semotilus corporalis*)					
0	**1.486%**	0.000%	0.000%		0.000%
1	**1.310%**	**4.731%**	**0.079%**	**0.287%**	0.000%
2	**1.480%**	**2.153%**	**2.211%**	**0.054%**	0.000%
3	**3.091%**	**3.001%**	0.000%	**0.076%**	**0.004%**
4	**1.884%**	**0.148%**	**0.242%**		
5	**4.465%**	0.000%			
Rainbow trout (*Oncorhynchus mykiss*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.572%**	0.000%	0.000%	**0.058%**	0.000%
2	**0.172%**	**0.145%**	0.000%	**0.007%**	**0.096%**
3	**0.171%**	**0.055%**	0.000%	0.000%	**0.090%**
4	**0.207%**	**0.108%**	0.000%		
5	0.000%	0.000%			
Brown trout (*Salmo trutta*)					
0	**0.137%**	0.000%	0.000%		0.000%
1	**2.560%**	**1.476%**	0.000%	0.000%	0.000%
2	**2.843%**	**3.159%**	**1.057%**	0.000%	**0.415%**
3	**1.698%**	**0.536%**	0.000%	0.000%	**0.058%**
4	**4.209%**	**0.336%**	0.000%		
5	**0.441%**	0.000%			
Rock bass (*Ambloplites rupestris*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.649%**	0.000%	0.000%	0.000%	0.000%
2	**0.631%**	**0.146%**	**0.118%**	0.000%	**0.105%**
3	**1.434%**	**1.776%**	0.000%	0.000%	**0.997%**
4	**3.137%**	**3.474%**	0.000%		
5	**2.882%**	**3.401%**			
Smallmouth bass (*Micropterus dolomieu*)					
0	0.000%	**0.167%**	0.000%		0.000%
1	**0.268%**	**1.743%**	**0.565%**	0.000%	0.000%
2	**0.528%**	**0.143%**	**0.030%**	**0.050%**	0.000%
3	**0.588%**	**0.386%**	0.000%	**0.025%**	**0.288%**
4	**1.231%**	**0.303%**	0.000%		
5	**4.310%**	**3.401%**			

Table 3.3.4. Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show greater relative abundance in small streams with increasing urbanization. The blacknose dace and creek chub are common in non-urban small streams as well, while the swallowtail shiner and fathead minnow occur mainly in urban streams. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Blacknose dace (*Rhinichthys atratulus*)					
0	**33.873%**	**38.899%**	**76.136%**		**66.763%**
1	**31.076%**	**23.010%**	**18.423%**	**28.440%**	**8.459%**
2	**26.857%**	**17.811%**	**18.433%**	**15.138%**	**14.061%**
3	**15.709%**	**5.662%**	**2.298%**	**3.228%**	**10.201%**
4	**7.980%**	**5.329%**	**0.887%**		
5	**8.820%**	0.000%			
Creek chub (*Semotilus atromaculatus*)					
0	**8.229%**	**24.741%**	**23.864%**		**9.250%**
1	**6.264%**	**5.670%**	**8.475%**	**9.381%**	**5.817%**
2	**7.405%**	**4.505%**	**4.494%**	**6.684%**	**1.999%**
3	**1.995%**	**0.615%**	**0.307%**	0.000%	**0.661%**
4	**0.277%**	**1.002%**	**1.210%**		
5	**0.029%**	**0.680%**			
Swallowtail shiner (*Notropis procne*)					
0	**0.061%**	0.000%	0.000%		**5.433%**
1	**0.127%**	**0.303%**	0.000%	**0.126%**	**2.799%**
2	**0.230%**	**0.385%**	**0.820%**	**6.749%**	**2.876%**
3	**1.171%**	**5.469%**	0.000%	**0.322%**	**13.834%**
4	**0.042%**	**0.009%**	0.000%		
5	**0.347%**	0.000%			
Fathead minnow (*Pimephales promelas*)					
0	0.000%	**4.344%**	0.000%		0.000%
1	**0.011%**	**0.438%**	**1.349%**	**1.605%**	0.000%
2	**0.014%**	**0.017%**	0.000%	**0.057%**	**0.329%**
3	**0.002%**	**0.007%**	0.000%	**0.046%**	**0.059%**
4	**0.003%**	0.000%	0.000%		
5	0.000%	0.000%			

Table 3.3.4 (continued). Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show greater relative abundance in small streams with increasing urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Satinfin shiner (*Cyprinella analostana*)					
0	**0.182%**	0.000%	0.000%		**0.453%**
1	**0.036%**	**0.044%**	0.000%	**0.084%**	**7.715%**
2	**0.493%**	**0.474%**	**0.438%**	**2.113%**	**4.382%**
3	**0.634%**	**1.848%**	**3.414%**	**0.727%**	**7.929%**
4	**0.940%**	**2.348%**	0.000%		
5	**0.027%**	0.000%			
Common shiner (*Luxilus cornutus*)					
0	**0.303%**	0.000%	0.000%		**0.970%**
1	**2.914%**	**3.493%**	**11.914%**	**4.008%**	0.000%
2	**3.448%**	**2.661%**	**2.400%**	**7.363%**	**0.613%**
3	**3.990%**	**4.734%**	0.000%	**4.724%**	**1.509%**
4	**2.261%**	**0.838%**	0.000%		
5	**6.707%**	0.000%			
Longnose dace (*Rhinichthys cataractae*)					
0	**0.167%**	0.000%	0.000%		**0.259%**
1	**2.439%**	**4.838%**	0.000%	0.000%	**2.100%**
2	**6.205%**	**10.118%**	**3.068%**	**2.216%**	**2.641%**
3	**5.352%**	**6.060%**	0.000%	**4.535%**	**3.168%**
4	**3.942%**	**5.021%**	0.000%		
5	**2.707%**	**12.245%**			
White sucker (*Catostomus commersoni*)					
0	**1.493%**	**1.751%**	0.000%		**8.538%**
1	**6.908%**	**20.874%**	**11.314%**	**9.399%**	**13.941%**
2	**9.921%**	**10.914%**	**11.318%**	**10.848%**	**13.028%**
3	**11.882%**	**12.648%**	**18.272%**	**15.404%**	**12.455%**
4	**12.470%**	**7.989%**	**10.961%**		
5	**13.958%**	**8.844%**			
Banded killifish (*Fundulus diaphanus*)					
0	0.000%	**22.222%**	0.000%		**1.294%**
1	**2.035%**	**7.043%**	**26.140%**	**25.481%**	**3.292%**
2	**2.233%**	**2.946%**	**2.088%**	**8.221%**	**32.198%**
3	**2.142%**	**3.867%**	0.000%	**8.449%**	**2.973%**
4	**1.821%**	**0.572%**	**1.131%**		
5	**0.884%**	**12.925%**			

Table 3.3.4 (continued). Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show greater relative abundance in small streams with increasing urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Mummichog (*Fundulus heteroclitus*)					
0	0.000%	**1.086%**	0.000%		0.000%
1	**0.001%**	**0.741%**	**0.104%**	0.000%	**0.195%**
2	0.000%	0.000%	0.000%	**0.013%**	**5.247%**
3	0.000%	0.000%	0.000%	**2.201%**	**9.725%**
4	0.000%	0.000%	0.000%		
5	0.000%	0.000%			
Redbreast sunfish (*Lepomis auritus*)					
0	0.000%	**0.167%**	0.000%		0.000%
1	**0.098%**	**2.277%**	**5.611%**	**2.063%**	**4.214%**
2	**2.533%**	**3.195%**	**4.397%**	**6.064%**	**3.399%**
3	**4.044%**	**7.724%**	**4.554%**	**6.677%**	**5.130%**
4	**3.507%**	**7.128%**	**11.937%**		
5	**11.826%**	**17.687%**			
Green sunfish (*Lepomis cyanellus*)					
0	0.000%	**2.334%**	0.000%		**0.776%**
1	**0.438%**	**1.182%**	**3.631%**	**2.493%**	**2.674%**
2	**1.645%**	**0.782%**	**2.241%**	**2.950%**	**1.006%**
3	**0.264%**	**2.508%**	**4.151%**	**4.367%**	**0.676%**
4	**1.741%**	**0.794%**	**15.035%**		
5	**0.850%**	**6.122%**			
Tesselated darter (*Etheostoma olmstedi*)					
0	**2.680%**	0.000%	0.000%		**4.140%**
1	**6.637%**	**11.421%**	**2.223%**	**6.512%**	**37.590%**
2	**8.576%**	**13.288%**	**19.283%**	**9.314%**	**9.341%**
3	**6.675%**	**13.000%**	**6.642%**	**10.101%**	**8.165%**
4	**7.509%**	**9.586%**	**7.345%**		
5	**2.063%**	**10.204%**			

Table 3.3.5. Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show greater relative abundance with increasing urbanization, but no clear expansion into small streams. Some species show lower relative abundance at very high levels of urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
American eel (*Anguilla rostrata*)					
0	**1.128%**	0.000%	0.000%		**0.259%**
1	**1.971%**	**3.545%**	0.000%	**0.633%**	**1.483%**
2	**4.802%**	**10.067%**	**9.684%**	**8.382%**	**3.056%**
3	**7.991%**	**7.079%**	**16.667%**	**16.442%**	**7.069%**
4	**10.579%**	**14.793%**	0.000%		
5	**6.684%**	**20.408%**			
Eastern mudminnow (*Umbra pygmaea*)					
0	0.000%	0.000%	0.000%		0.000%
1	0.000%	0.000%	0.000%	0.000%	0.000%
2	**0.460%**	**4.062%**	**1.287%**	**0.915%**	0.000%
3	**0.471%**	**0.485%**	**8.491%**	0.000%	0.000%
4	**0.096%**	**0.040%**	**0.529%**		
Eastern silvery minnow (*Hybognathus regius*)					
0	0.000%	0.000%	0.000%		0.000%
1	0.000%	0.000%	0.000%	0.000%	0.000%
2	0.000%	0.000%	0.000%	0.000%	**0.012%**
3	**0.161%**	0.000%	**4.088%**	**2.565%**	**1.112%**
4	0.000%	0.000%	**13.986%**		
5	0.000%	0.000%			
Spottail shiner (*Notropis hudsonius*)					
0	**1.270%**	0.000%	0.000%		**0.194%**
1	**1.006%**	**0.214%**	0.000%	**0.414%**	**0.766%**
2	**2.795%**	**2.764%**	**1.722%**	**3.928%**	**0.785%**
3	**5.843%**	**6.368%**	**15.591%**	**9.000%**	**6.191%**
4	**3.306%**	**9.757%**	**14.046%**		
5	**0.030%**	**0.680%**			
Comely shiner (*Notropis amoenus*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.005%**	0.000%	0.000%	0.000%	0.000%
2	**0.152%**	**0.129%**	0.000%	**0.054%**	**0.006%**
3	**0.699%**	**0.122%**	0.000%	0.000%	**0.024%**
4	**0.042%**	**0.009%**	0.000%		
5	**0.000%**	**0.000%**			

Table 3.3.5. (Continued) Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show greater relative abundance with increasing urbanization, but no clear expansion into small streams. Some species show lower relative abundance at very high levels of urbanization. Gray cells are those for which no samples exist.

Urbanization Group				
0	1	2	3	4
Pumpkinseed (*Lepomis gibbosus*)				
0	**2.687%**	**1.667%**	0.000%	
1	**2.479%**	**1.813%**	**1.689%**	**2.769%**
2	**2.683%**	**2.709%**	**5.852%**	**2.720%**
3	**2.057%**	**1.683%**	**9.028%**	**6.088%**
4	**1.747%**	**4.053%**	**9.373%**	
5	**1.632%**	0.000%		
Bluegill (*Lepomis maculatus*)				
0	**0.575%**	**0.167%**	0.000%	
1	**1.587%**	**0.324%**	**0.565%**	**0.089%**
2	**1.820%**	**1.631%**	**2.255%**	**3.831%**
3	**1.645%**	**1.418%**	**2.478%**	**3.385%**
4	**1.269%**	**2.068%**	**0.485%**	
5	**3.422%**	**0.680%**		

Table 3.3.6. Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show no clear pattern of occurrence with respect to urbanization, except for decrease of some species at very high levels of urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Redfin pickerel (*Esox americanus*)					
0	0.000%	0.000%	0.000%		0.000%
1	**1.034%**	0.000%	0.000%	**0.580%**	**0.444%**
2	**1.185%**	**0.900%**	**2.584%**	**0.266%**	0.000%
3	**0.321%**	**0.763%**	**0.114%**	**0.142%**	0.000%
4	**0.276%**	**0.339%**	**0.403%**		
5	0.000%	0.000%			
Common carp (*Cyprinus carpio*)					
0	0.000%	0.000%	0.000%		0.000%
1	0.000%	0.000%	0.000%	**0.190%**	0.000%
2	0.000%	0.000%	0.000%	**0.062%**	**0.024%**
3	**0.003%**	**0.178%**	**0.883%**	**0.041%**	0.000%
4	**0.594%**	**2.554%**	**0.950%**		
5	**0.099%**	0.000%			
Golden shiner (*Notemigonus crysoleucas*)					
0	0.000%	**0.167%**	0.000%		0.000%
1	**1.225%**	**0.052%**	**0.263%**	**0.945%**	0.000%
2	**0.564%**	**0.069%**	**0.541%**	**0.369%**	**0.211%**
3	**0.086%**	**0.108%**	**0.061%**	**0.546%**	**0.008%**
4	**0.939%**	**0.234%**	0.000%		
5	**0.769%**	0.000%			
Spotfin shiner (*Cyprinella spiloptera*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.046%**	**0.236%**	**3.390%**	**0.299%**	**0.076%**
2	**0.209%**	0.000%	0.000%	0.000%	**0.160%**
3	**0.009%**	**0.597%**	0.000%	0.000%	**0.395%**
4	**0.178%**	**0.026%**	0.000%		
5	**2.680%**	**1.361%**			
Creek chubsucker (*Erimyzon oblongus*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.691%**	**0.123%**	0.000%	**0.682%**	0.000%
2	**0.681%**	**0.618%**	**0.177%**	**0.197%**	**0.006%**
3	**0.321%**	**0.072%**	0.000%	**0.052%**	0.000%
4	**0.222%**	**0.146%**	0.000%		
5	0.000%	0.000%			
Brown bullhead (*Amiurus nebulosus)*					
0	**0.384%**	**1.500%**	0.000%		0.000%
1	**1.125%**	0.000%	0.000%	**0.211%**	0.000%
2	**0.546%**	**0.419%**	**0.622%**	**0.249%**	**0.308%**
3	**0.428%**	**0.083%**	0.000%	**0.328%**	**0.186%**
4	**0.265%**	**0.476%**	0.000%		
5	0.000%	**1.361%**			

Table 3.3.6. (Continued) Average species proportions of selected species for different watershed size classes and urbanization classes. See Table 3.3.1 for explanation of classes. This table includes species which show no clear pattern of occurrence with respect to urbanization, except for decrease of some species at very high levels of urbanization. Gray cells are those for which no samples exist.

	Urbanization Group				
	0	1	2	3	4
Yellow bullhead (*Amieurus natalis*)					
0	**0.061%**	**0.417%**	0.000%		0.000%
1	**0.039%**	**1.329%**	**0.565%**	**1.273%**	**0.473%**
2	**0.308%**	**0.107%**	**0.170%**	**0.381%**	**0.079%**
3	**0.204%**	**0.305%**	**1.790%**	**0.100%**	**0.733%**
4	**0.298%**	**0.923%**	**4.831%**		
5	**0.999%**	0.000%			
Largemouth bass (*Micropterus salmoides*)					
0	**0.056%**	0.000%	0.000%		0.000%
1	**0.249%**	**0.741%**	**3.598%**	**0.447%**	**0.148%**
2	**0.406%**	**0.845%**	**1.082%**	**0.260%**	**0.063%**
3	**0.481%**	**0.462%**	**0.664%**	**0.409%**	**1.407%**
4	**1.083%**	**0.654%**	**0.727%**		
5	**2.265%**	0.000%			
Black crappie (*Pomoxis nigromaculatus*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.011%**	0.000%	0.000%	0.000%	0.000%
2	**0.014%**	**0.023%**	**0.036%**	**0.198%**	**0.027%**
3	**0.020%**	**0.028%**	**0.445%**	**0.021%**	**0.006%**
4	**0.291%**	**0.252%**	0.000%		
5	0.000%	0.000%			
Yellow perch (*Perca flavescens*)					
0	0.000%	0.000%	0.000%		0.000%
1	**0.209%**	**0.427%**	0.000%	**1.151%**	0.000%
2	**0.260%**	**0.261%**	**0.069%**	0.000%	0.000%
3	**0.015%**	**0.009%**	0.000%	0.000%	**1.919%**
4	**0.105%**	**0.009%**	0.000%		
5	**0.446%**	0.000%			
Species occurring in less than 5 cells:					
	Alewife		Western mosquitofish		
	American shad		Channel catfish		
	White catfish		Oriental weatherfish		
	Pirateperch		Striped bass		
	Stoneroller		White perch		
	Goldfish		Bridle shiner		
	Gizzard shad		Stonecat		
	Banded sunfish		Tadpole madtom		
	Northern pike		River chub		

3.3.2 Correlations among Primary Fish Metrics and Relationship to Stream Characteristics

The individual fish metrics tend to be correlated with land use and often with watershed area (Fig.3.3.1-3.3.14). The richness metrics tend to be strongly related to watershed area. Metrics related to species intolerance or tolerance and %riffle species tend to be relatively strongly correlated with %urban land in the watershed.

Correlations among metrics may be due to internal redundancy among metrics (e.g., if the same species influence several metrics) or to common response among different aspects of assemblages. The correlation structure among the 14 normalized metrics was assessed by PCA (Table 3.3.7). The first four components of the PCA accounted for 70% of total variance.

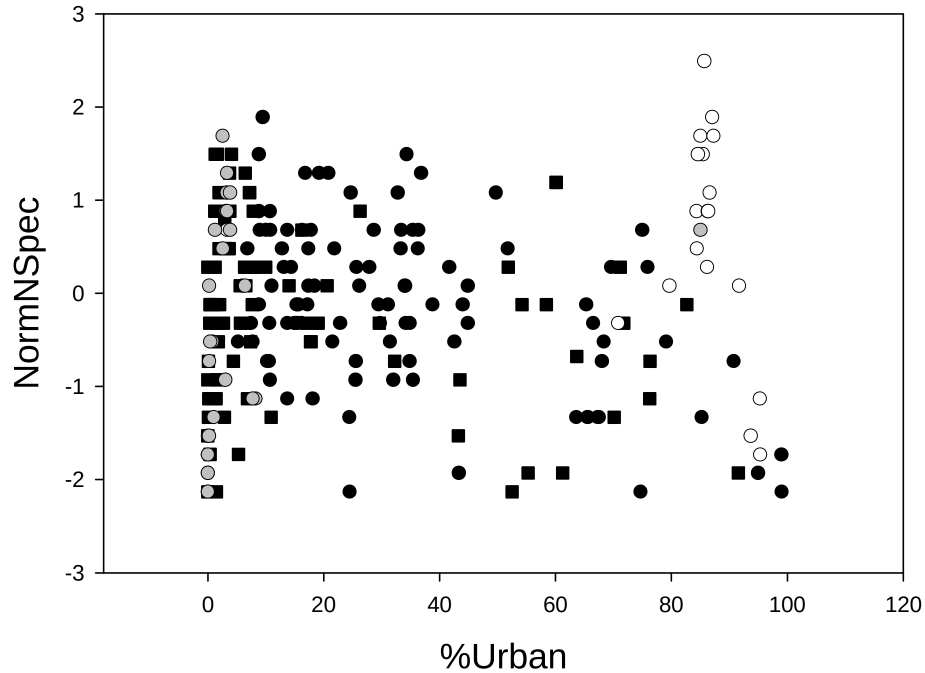
The first component (hereafter called Fpca1) was highly negatively correlated with various species richness measures (total number of species, number of native species, number of benthic invertivores), moderately negatively correlated with number of intolerant species, proportion of riffle species and proportion of white sucker, and moderately positively correlated with proportion of tolerant species and proportion of generalists. This correlation structure suggests that this component is largely a stream size gradient. Multiple linear regression of Fpca1 with watershed area (ln-transformed), land uses (square root transformation of proportions) and habitat scores indicate that Fpca1 is highly correlated ($p<0.000000$) with ln(watershed area) and with sqrt(proportion agriculture) (Figs. 3.3.15-3.3.16). Both slopes are negative, i.e., higher values of Fpca1 (associated with lower richnesses) are associated with smaller watersheds and lower proportions of agriculture. The watershed area is consistent with the common pattern of increasing species richness with increasing stream size. The agriculture relationship may represent two effects of land use. Extreme values of various land use proportions are most common in small watersheds (larger watersheds usually have a mix of land use types from headwaters downstream). Small forested watersheds often have low species richness (e.g., brook trout and a few other species). Increasing watershed development (e.g., increasing agriculture) may lead to increases in abundance of a variety of generalist species (including several minnows). In highly developed watersheds, increasing urbanization may lead to loss of species. Both patterns would lead to a negative relationship between the metrics associated with Fpca1 and proportion of agriculture.

The second component (hereafter Fpca2) was highly positively correlated with proportion of top carnivores and proportion of intolerant species, moderately positively correlated with number of intolerant species, highly negatively correlated with proportion of tolerant species and moderately negatively correlated with proportion of generalists and proportion of insectivorous cyprinids. This correlation structure suggests that this is a impairment gradient. Regression of Fpca2 with watershed area and land use proportions supports this interpretation (Figs. 3.3.17-3.3.18). There are highly significant negative relationships between Fpca2 and sqrt(urban) and sqrt(agriculture) ($p<0.000000$ for both effects). The slopes of these relationships are similar (-3.0 for sqrt(urban) and -2.6 for sqrt(agriculture)), indicating roughly similar effects of urban and agricultural land use on this component. In addition to the land use effects, there is a highly significant positive relationship ($p<0.000009$) with ln(watershed area). The positive area relationship indicates greater proportions of top carnivores and intolerant species in larger streams. The relationship between Fpca2 and ln(watershed area) consisted of an approximately linear, positive relationship for most points (Fig. 3.3.17), with a few outliers with high Fpca2 values and small-moderate stream sizes. The relationship between Fpca2 and sqrt(urban area) consists of an approximately linear, decreasing relationship (Fig. 3.3.16), with a few low-urban sites with high Fpca2 values. For the most part these outliers for the two relationships represent the same sites. These are mainly small tributaries of the Delaware River in Sussex County (a Vancampens Brook site and a Dunfield Creeek site) or of the Neversink River (e.g., Wolf Creek, Gumar Brook). These had a few species, including one-two species of trout. Some of these (e.g., the Vancampens site) may be impaired by acid precipitation. One site with a large Fpca2 value (2.7) for the size of stream, is from a tributary of Primrose Brook. This sample had only brook trout, slimy sculpin and blacknose dace.

The third component (hereafter Fpca3) was highly positively correlated with proportion of insectivorous cyprinids, moderately positively correlated with number of intolerant species, proportion of intolerant species and proportion of riffle species, highly negatively correlated with proportion of pool species, and moderately negatively correlated with number of salmonid-centrarchid species. This structure suggests that this reflects aspects of habitat quality (e.g., amount of pools versus riffles) and aspects of impairment not reflected in the second component. Multiple linear regression found significant negative relationships between Fpca3 and ln(watershed area) and sqrt(proportion wetland) and positive relationship with sqrt(proportion forest area) ($p<0.000001$ for all three effects). Habitat score was significant in models without sqrt(proportion forest), but was not significant when sqrt(proportion forest) was included. The land use effects indicate more lotic species (e.g., more riffle fish) in more forested watersheds and more lentic species (e.g., more pool fish) in developed watersheds, watersheds with lakes and wetlands (possibly providing source populations for some lentic species) and in larger watersheds.

The fourth axis was highly negatively correlated with proportion of omnivores. This axis is dominated by two samples with very high negative values (other points have low values on this axis). These are both highly urban sites, one on the Elizabeth River (NJ) and the other on Tacony Creek (PA). Both sites have very high proportions of mummichog (*Fundulus heteroclitus*), an omnivorous species, and low abundances of a few other, tolerant species. The fourth axis was significantly correlated with sqrt(proportion forest); this relationship reflects the importance of the few sites with high negative values of Fpca4.

The influence of extremely large or small streams on the correlation structure was assessed by conducting PCA on subsets of the data. One PCA was done on all sites with watershed area greater than 12.95 km^2 (i.e., greater than the 5 mi^2 lower threshold for the FIBI). Based on observed pattern between the Fpca1 and watershed area (Fig. 3.3.15), a second PCA was done on sites with watershed area between 7.4 and 54.6 km^2. Both PCAs showed a correlation structure similar to that of the full dataset, with the first component reflecting species richness of several groups, the second reflecting proportions of tolerant and intolerant species, the third reflecting pool species and cyprinids, and the fourth reflecting proportions of generalists and omnivores. There were some differences, such as importance of white sucker abundance on the fourth component for mid-sized streams.



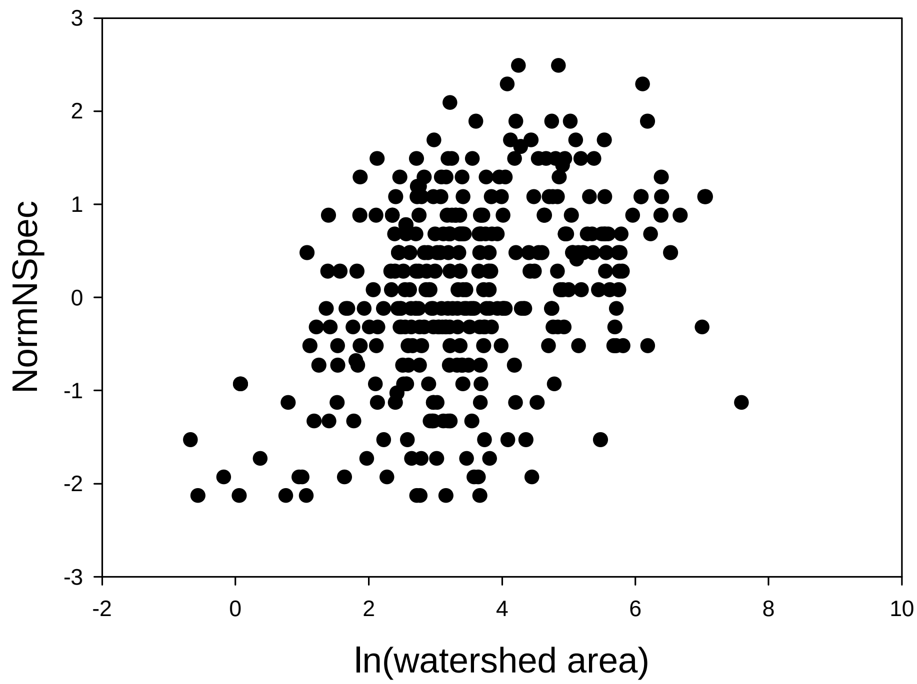


Figure 3.3.1. Relationship between normalized Nspec (total number of fish species) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

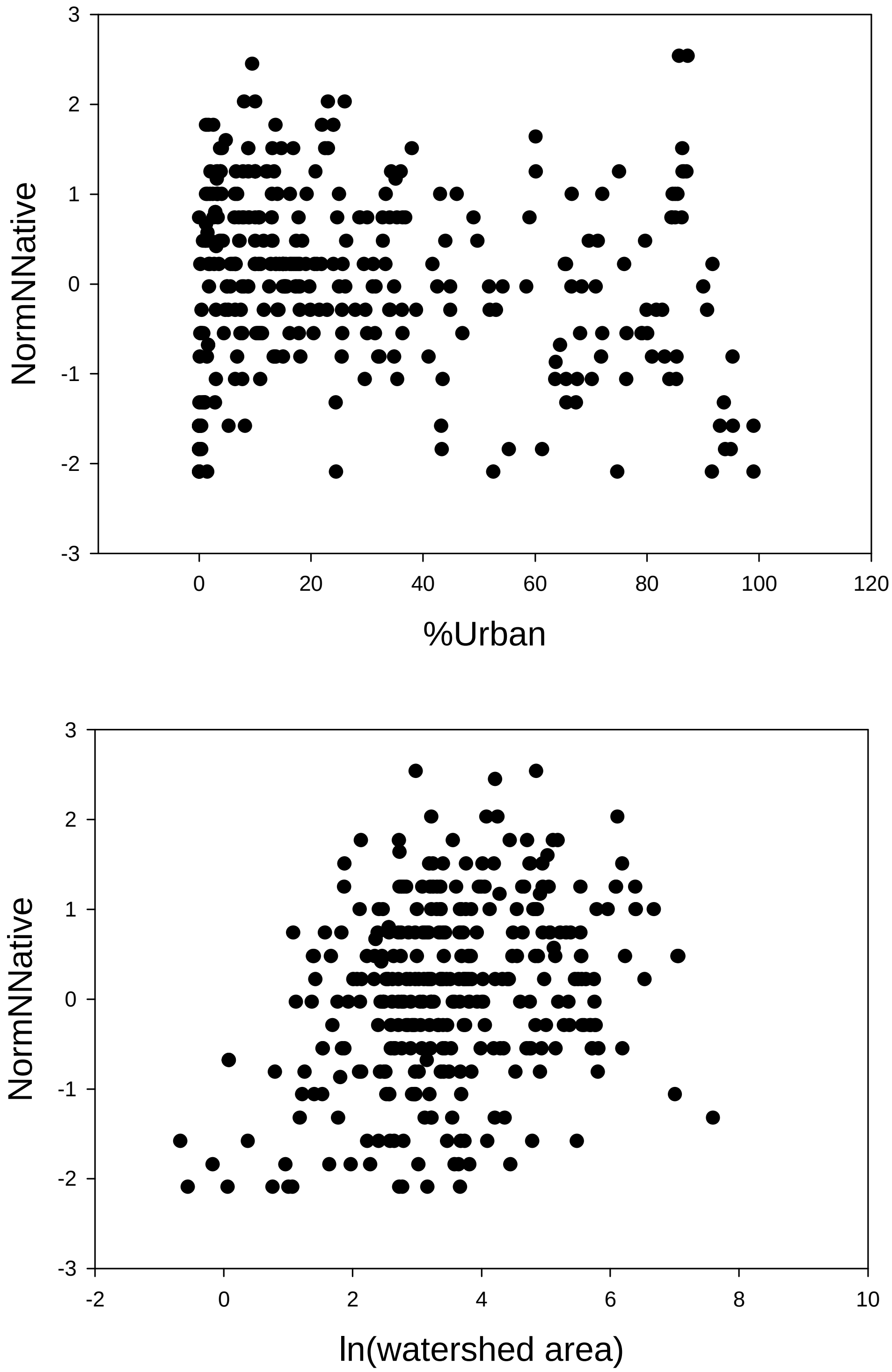
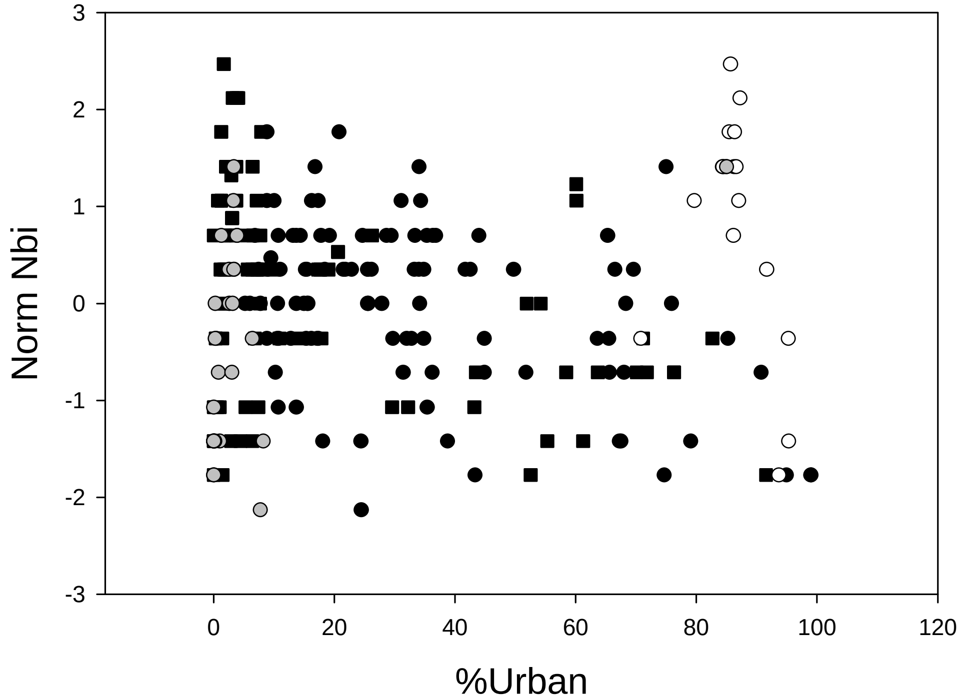


Figure 3.3.2. Relationship between normalized Nnat (total number of native fish species) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites.



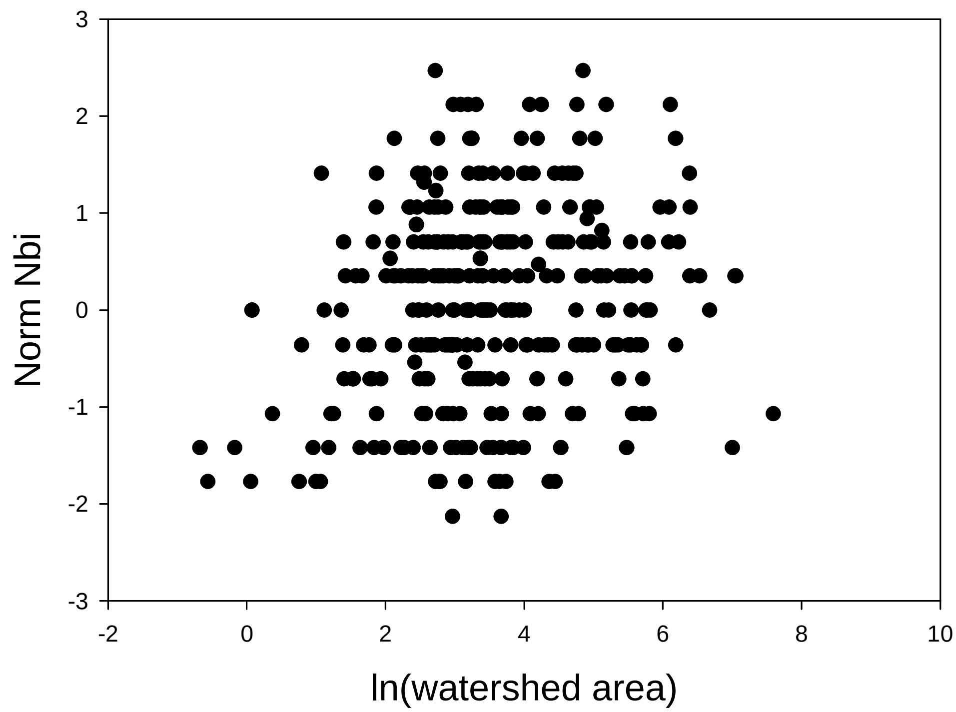
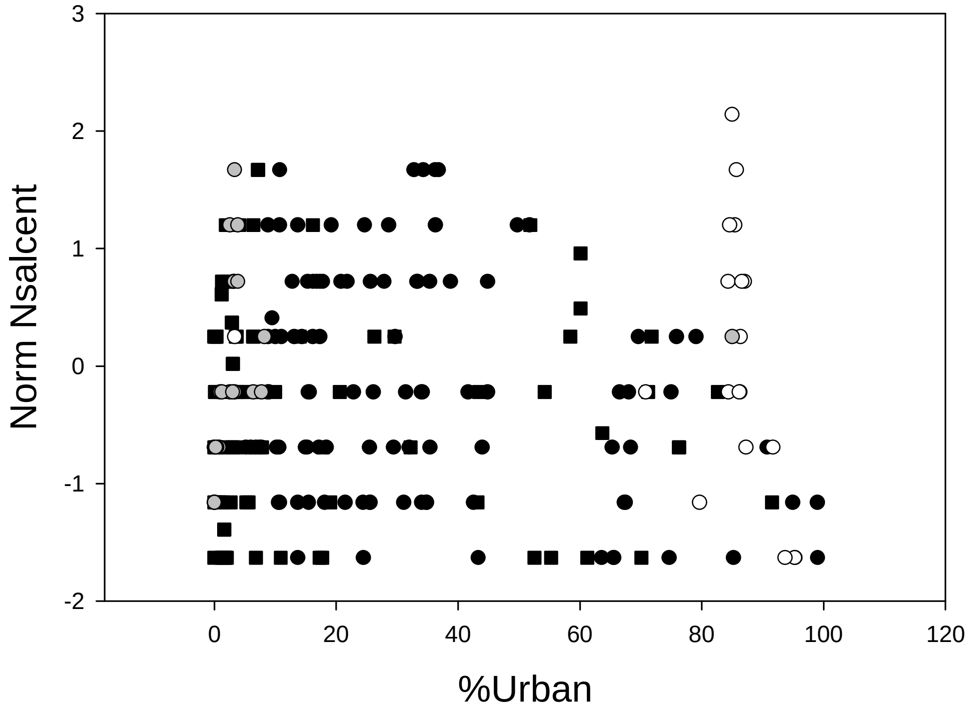


Figure 3.3.3. Relationship between normalized Nbi (number of benthic invertivore fish species) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).



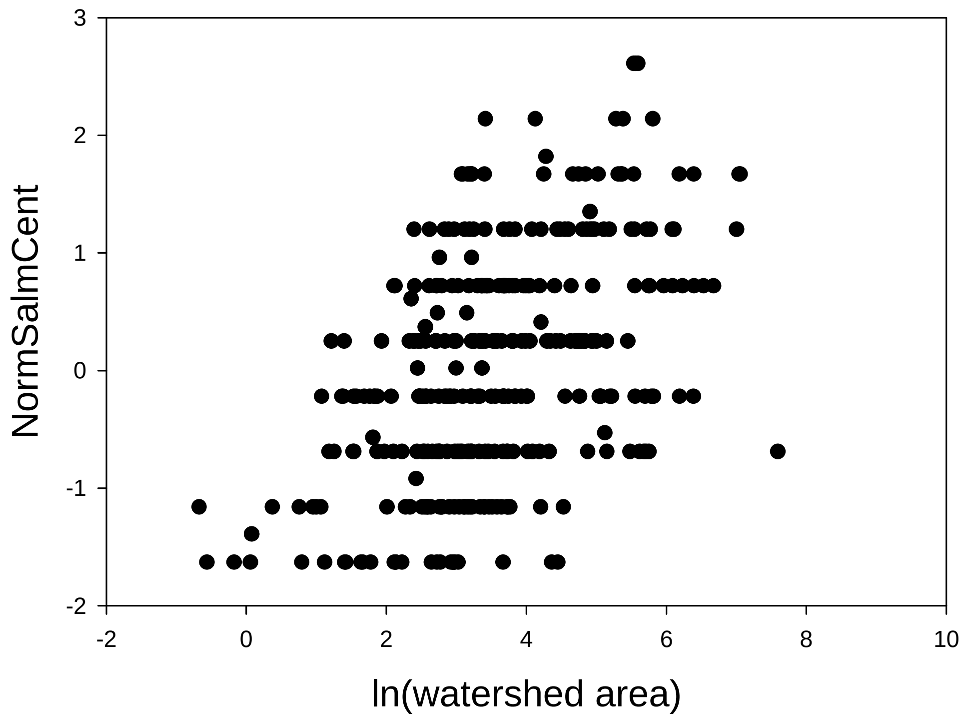
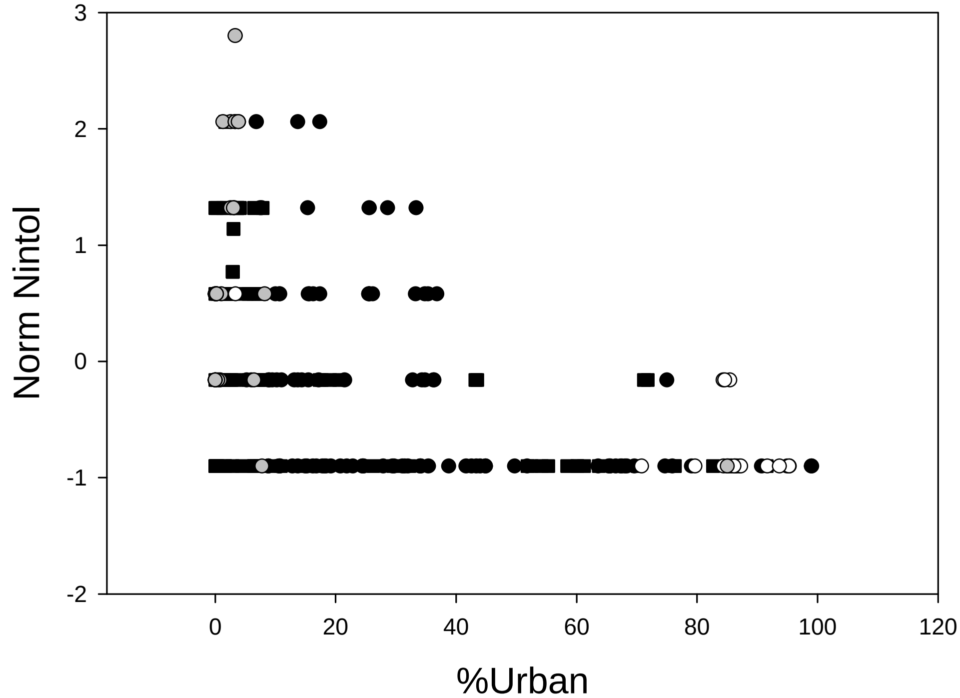


Figure 3.3.4. Relationship between normalized Nsalcent (number of salmonid and centrarchid species) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).



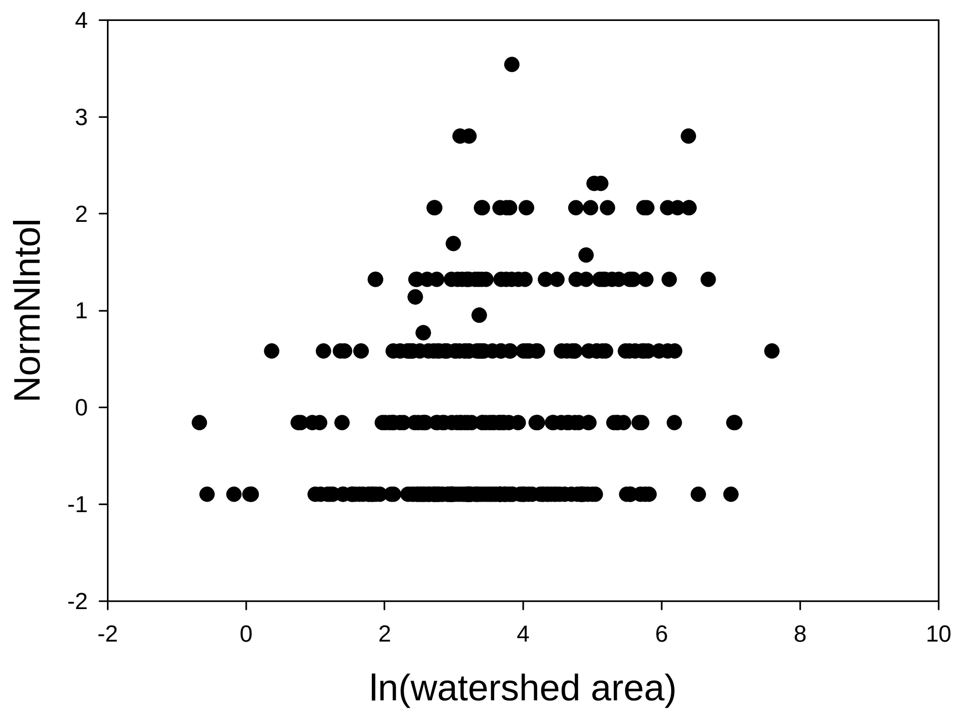
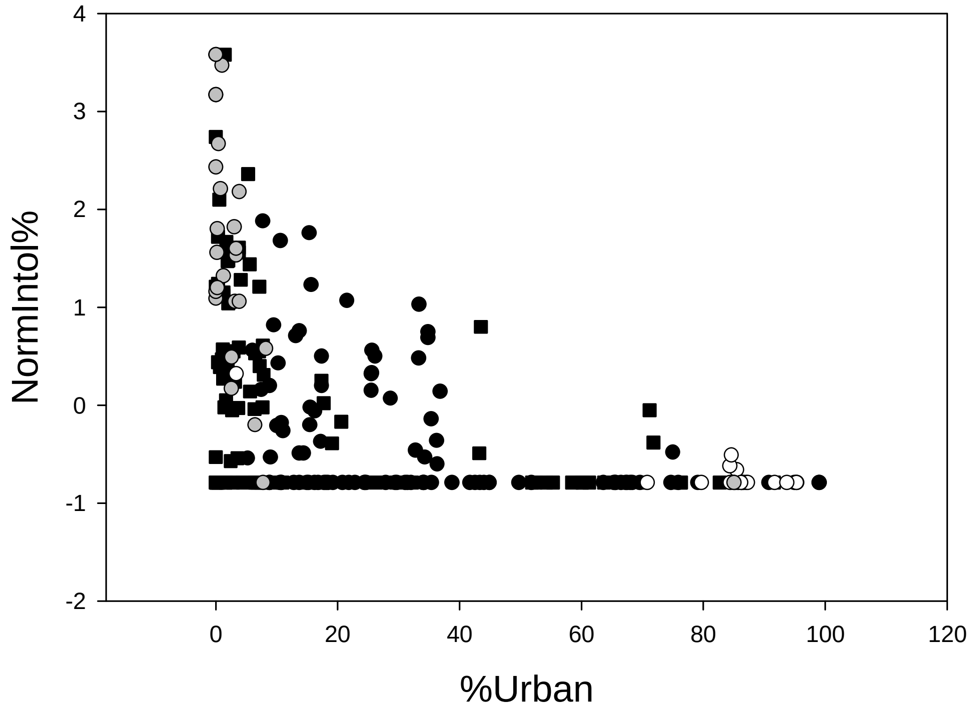


Figure 3.3.5. Relationship between normalized Nintol (number of intolerant fish species) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).



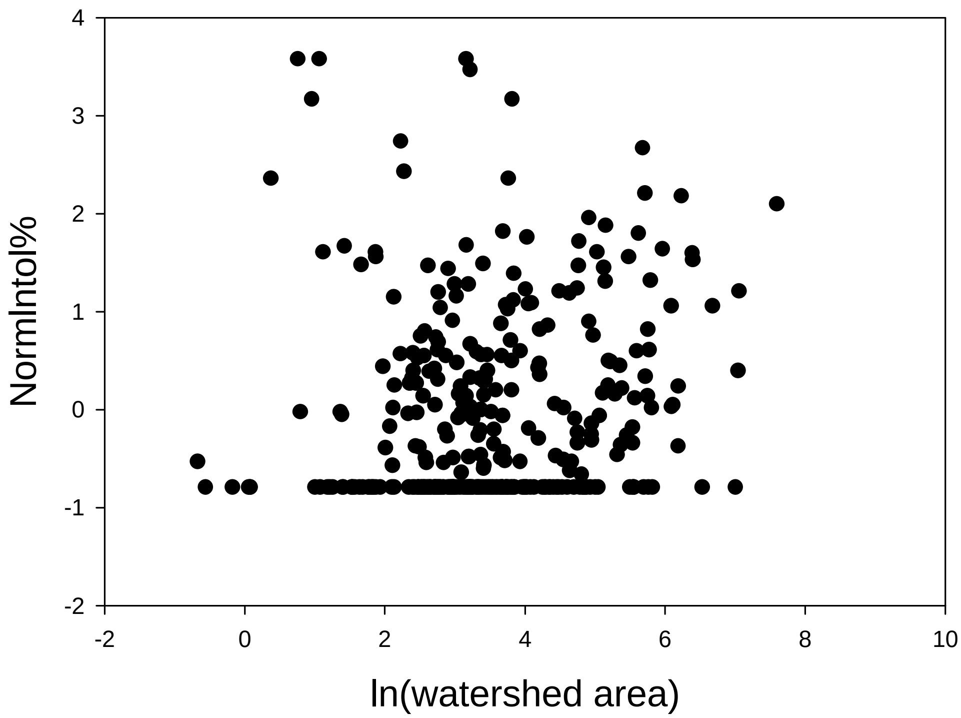


Figure 3.3.6. Relationship between normalized, transformed %Intol (proportion intolerant fish) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

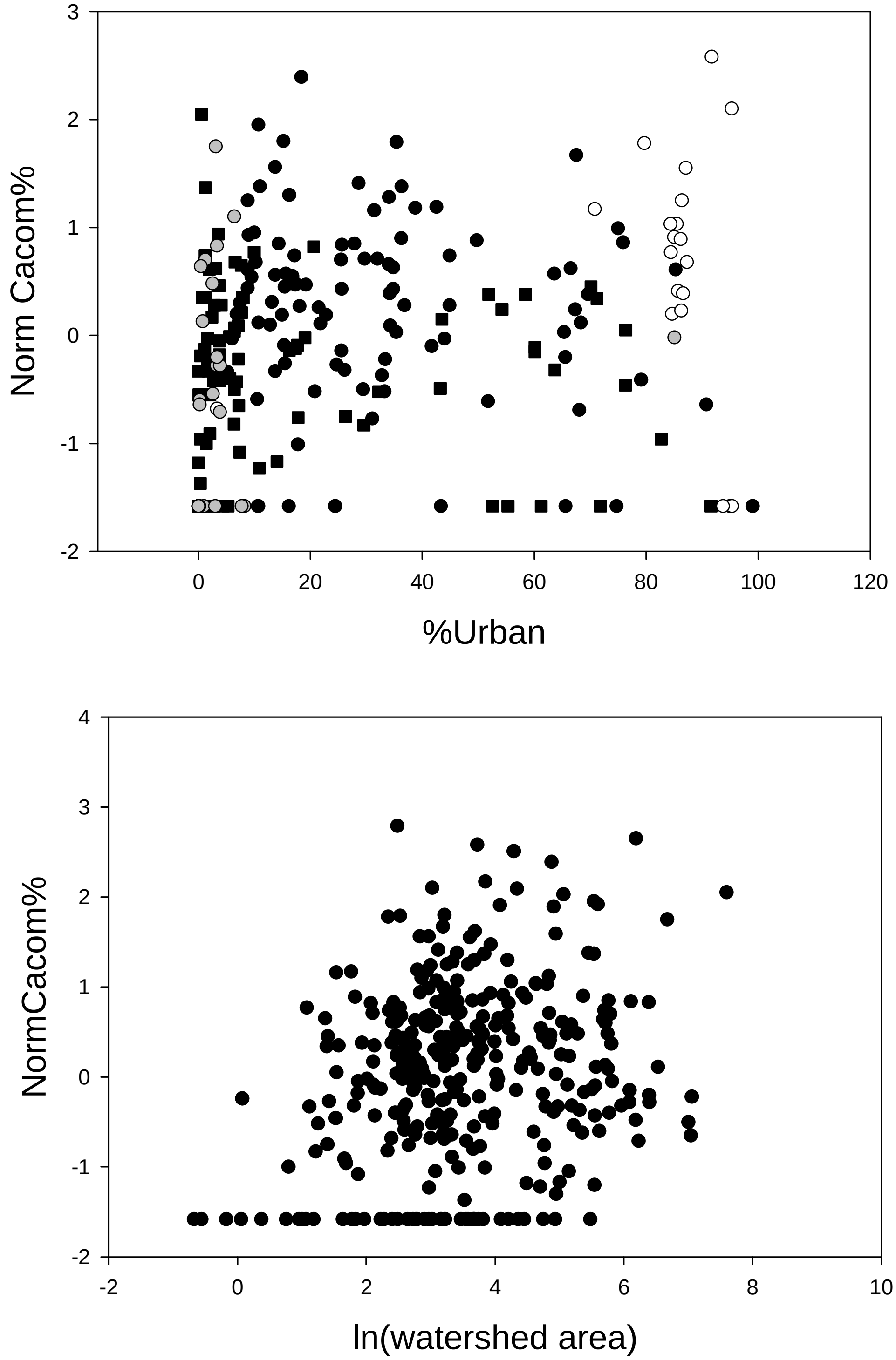


Figure 3.3.7. Relationship between normalized, transformed %Cacom (proportion white sucker) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

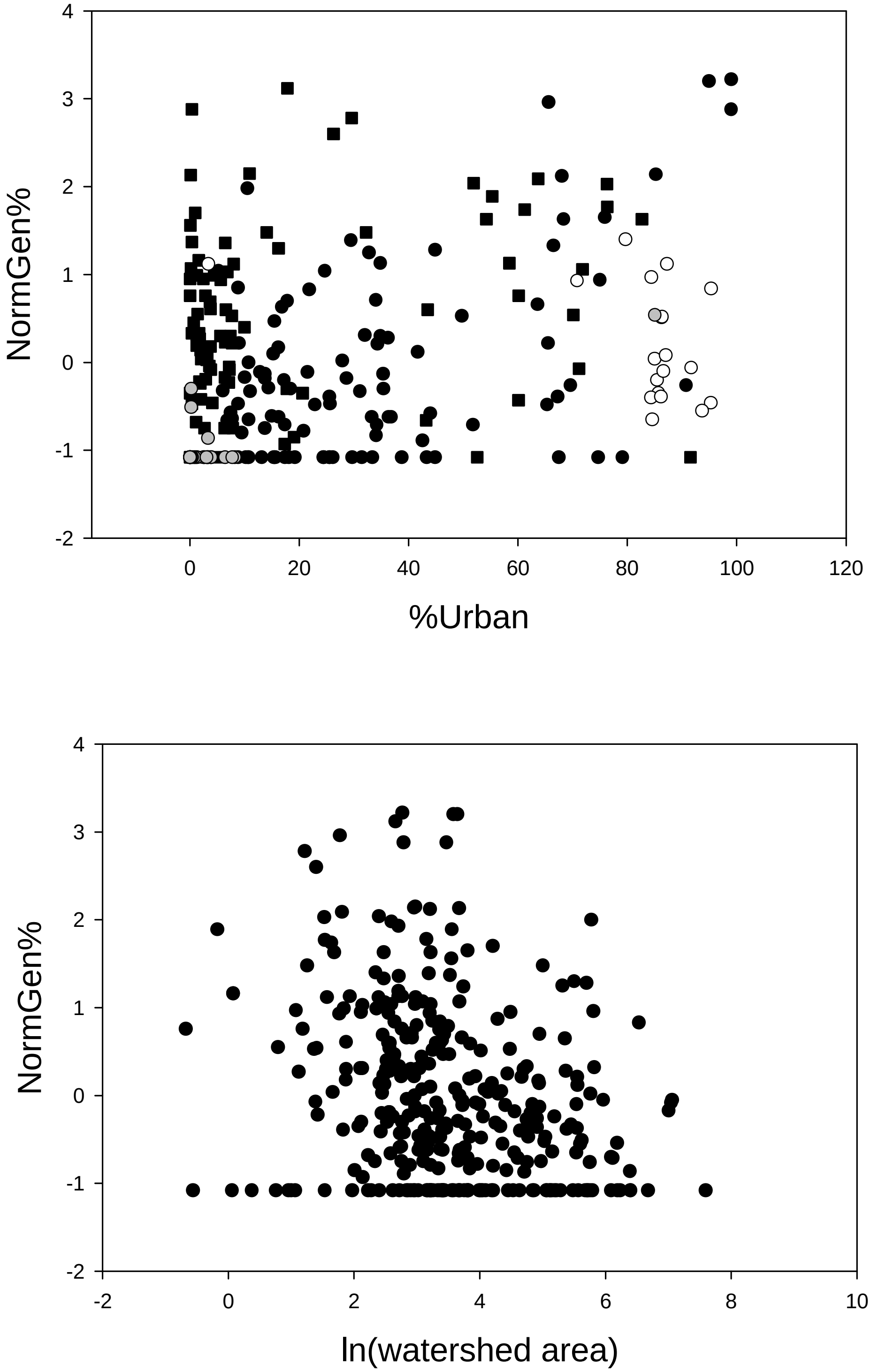
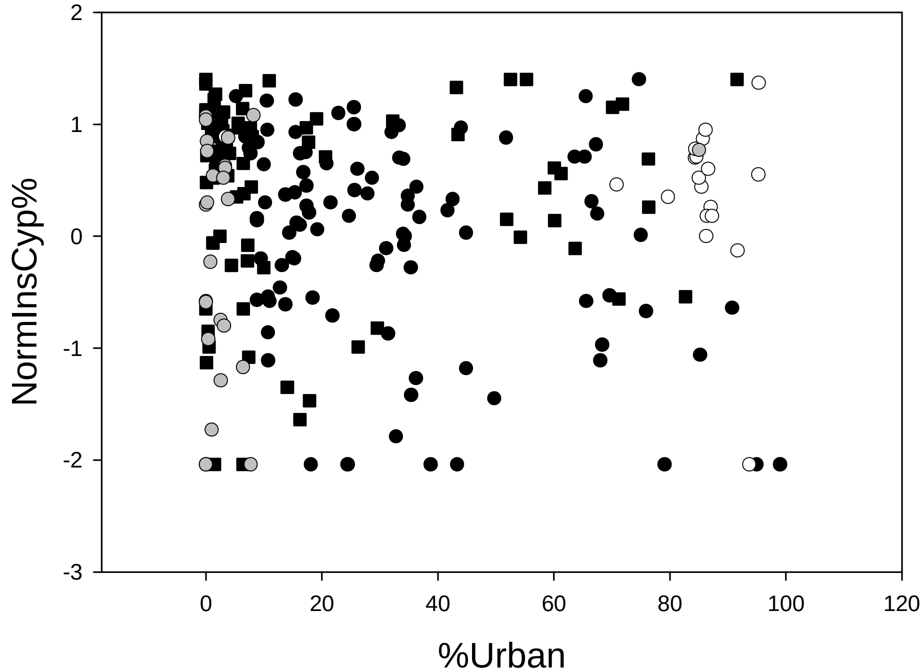


Figure 3.3.8. Relationship between normalized, transformed %Gen (proportion generalist fish) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).



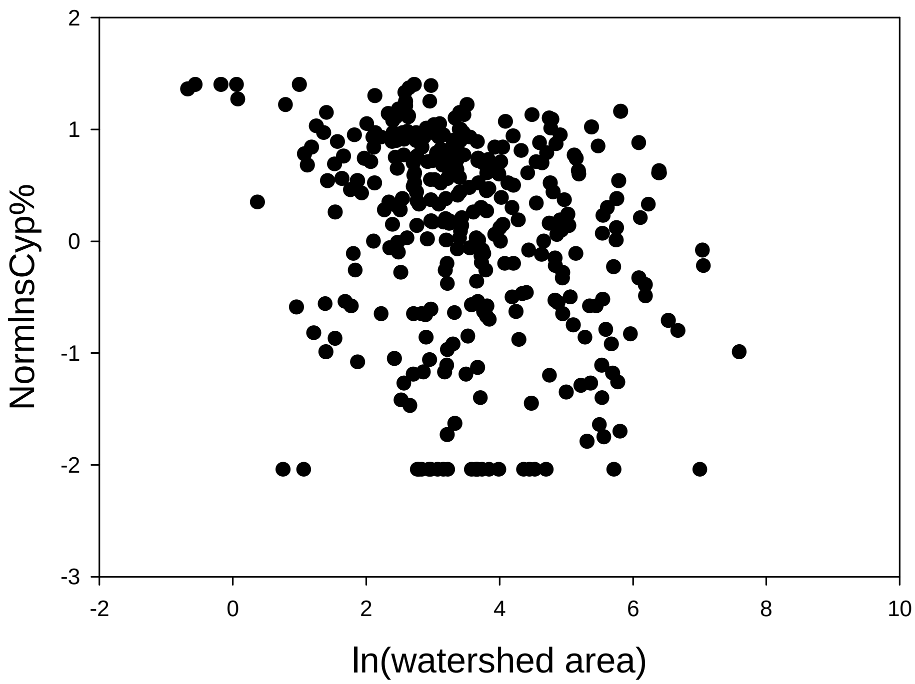


Figure 3.3.9. Relationship between normalized, transformed %Inscyp (proportion of insectivorous cyprinid fish) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

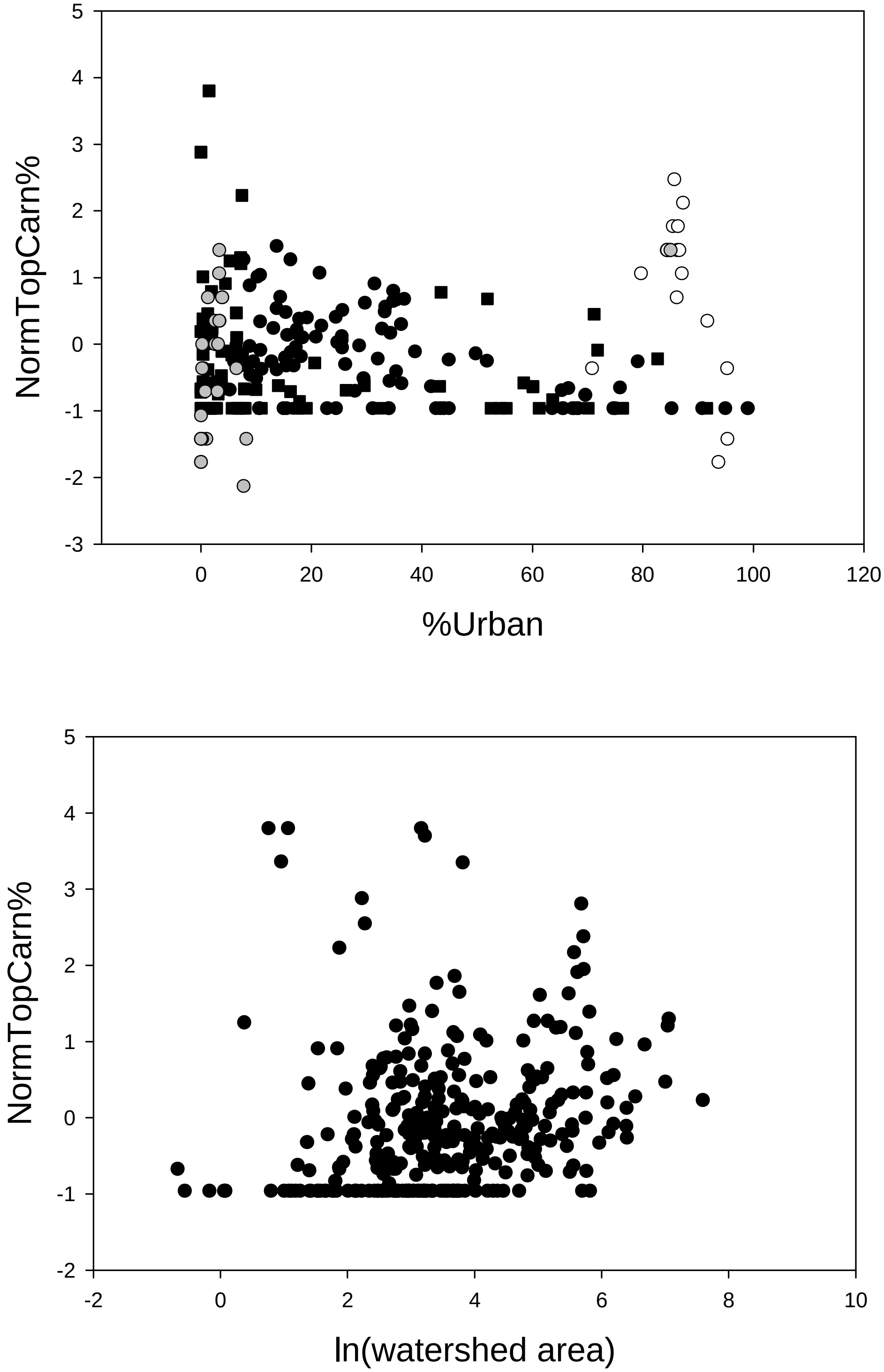


Figure 3.3.10. Relationship between normalized, transformed %Topcarn (proportion salmonids or top carnivores) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

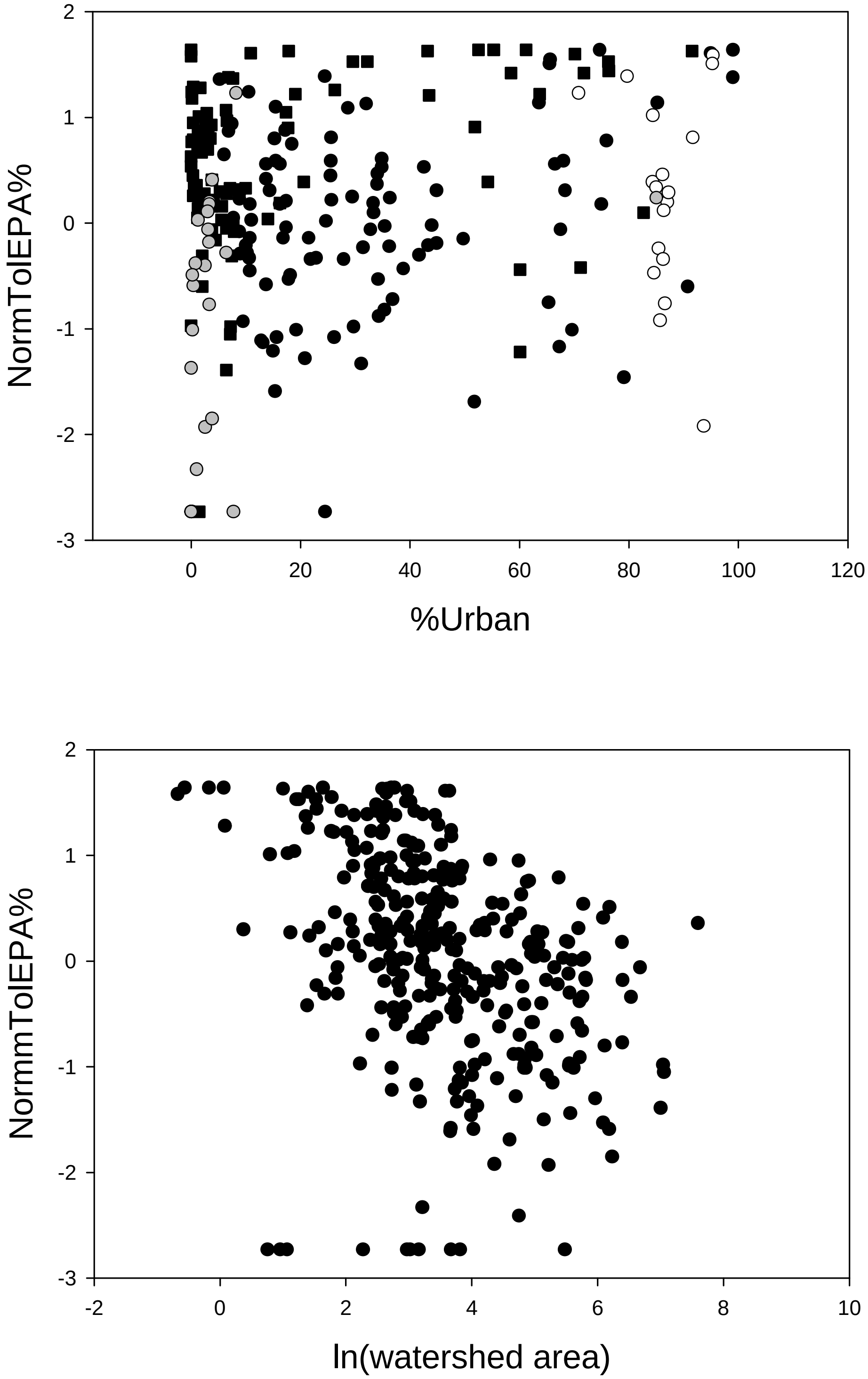


Figure 3.3.11. Relationship between normalized, transformed %Tol (proportion tolerant fish) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

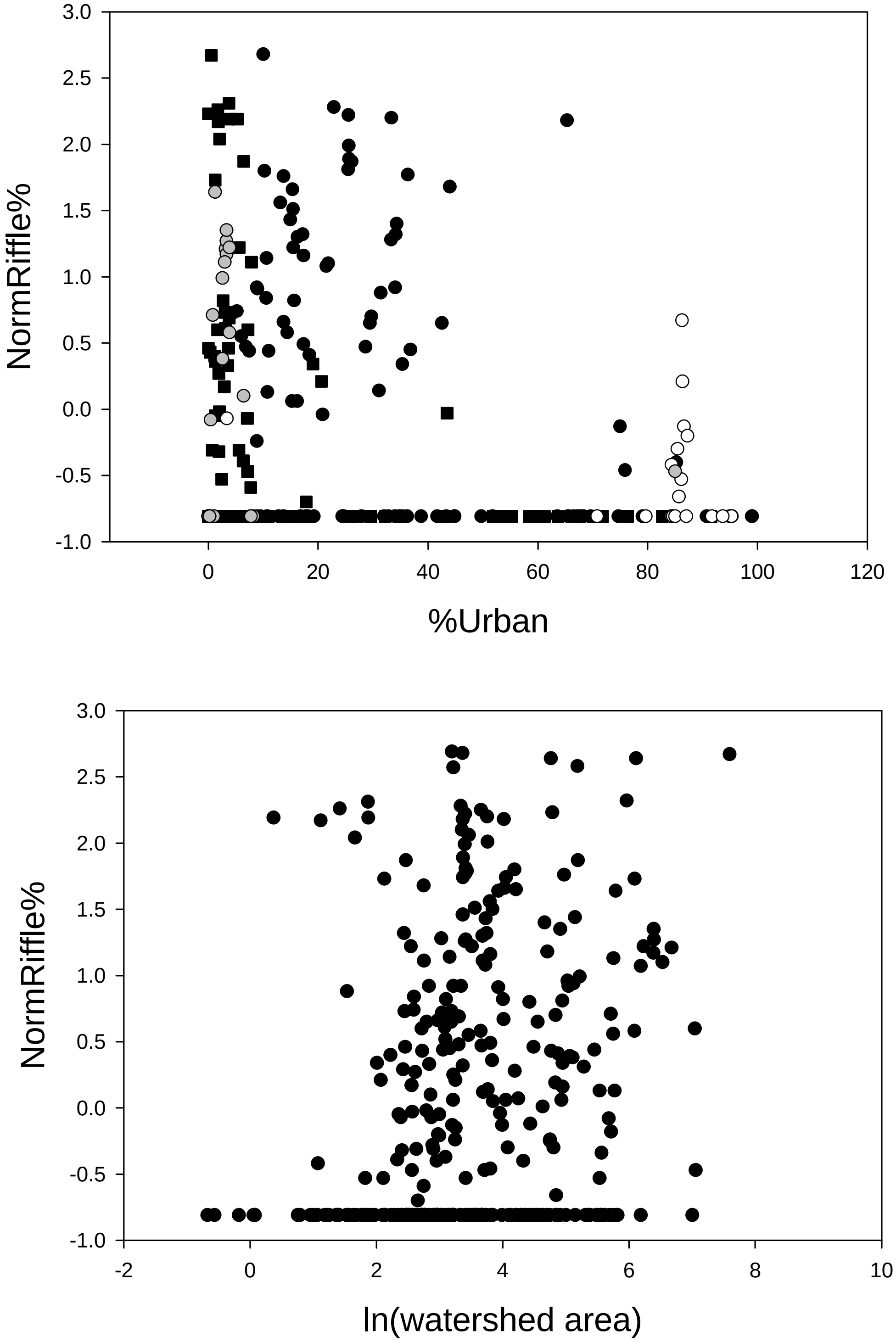


Figure 3.3.12. Relationship between normalized, transformed %riffle (proportion riffle fish) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

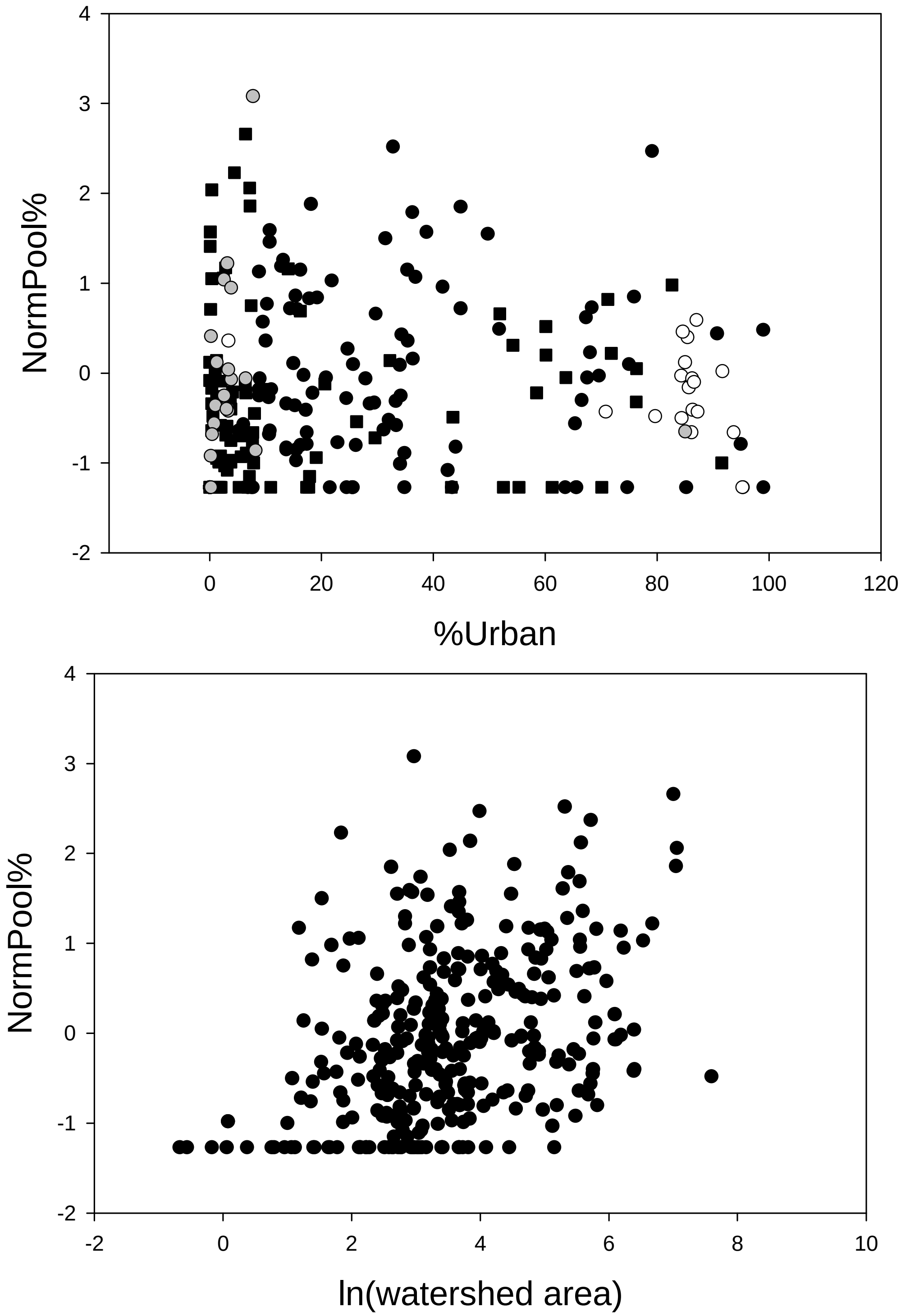
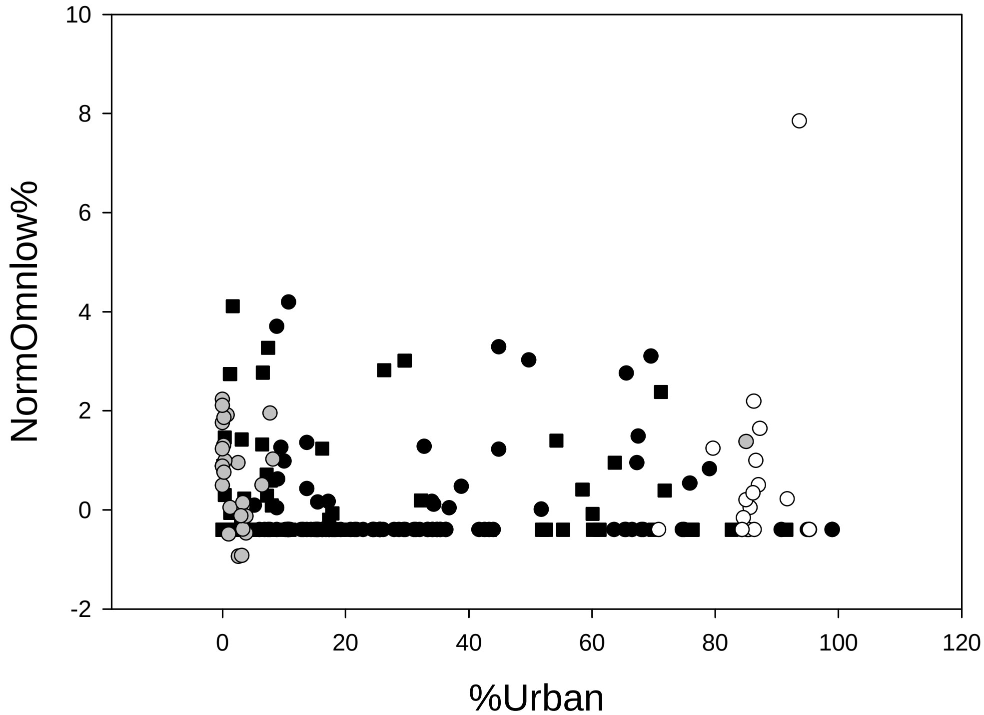


Figure 3.3.13. Relationship between normalized, transformed %pool (proportion pool fis) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).



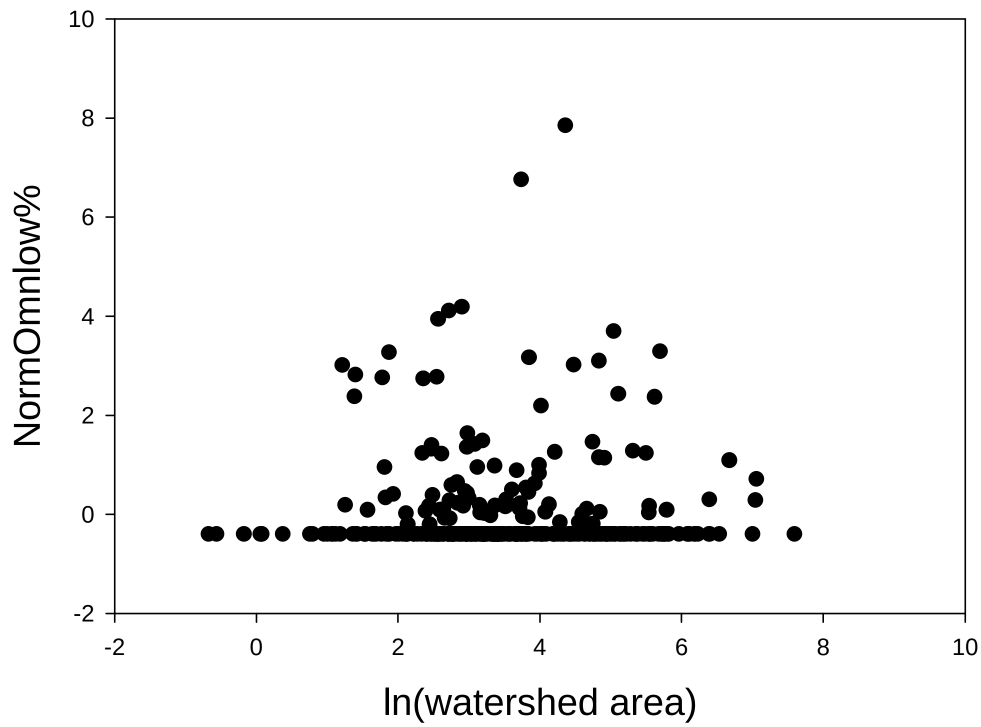


Figure 3.3.14. Relationship between normalized, transformed %Omnlow (proportion omnivorous fish) and %urban (top) and ln(watershed area) (bottom) for joint fish-macroinvertebrate sites. Points are coded by data source: ANS (closed squares), EPA, NAWQA and NJ FIBI (closed circles), PWD (gray circles) and TNC (open circles).

Table 3.3.7. Relationships between macroinvertebrate and fish principal components. Entries show r^2 and p-values for regressions between the fish PC (as dependent variable) and the four macroinvertebrate metrics (as independent variables). Pos and Neg indicate the sign of the slope of the relationship.

	Total model p	**Total model r2**	**MPca1**	**MPca2**	**MPca3**	**MPca4**
FPca1	<0.00016	0.067	<0.000079	<0.051	<0.065	ns
			Pos	Pos	Neg	
FPca2	<0.0000001	0.18	<0.0000001	<0.00015	<0.00029	ns
			Neg	Pos	Neg	
FPca3	<0.0000001	0.16	<0.0000001	ns	<0.00003	ns
			Neg		Neg	
FPca4	<0.29	0.015	ns	ns	ns	ns

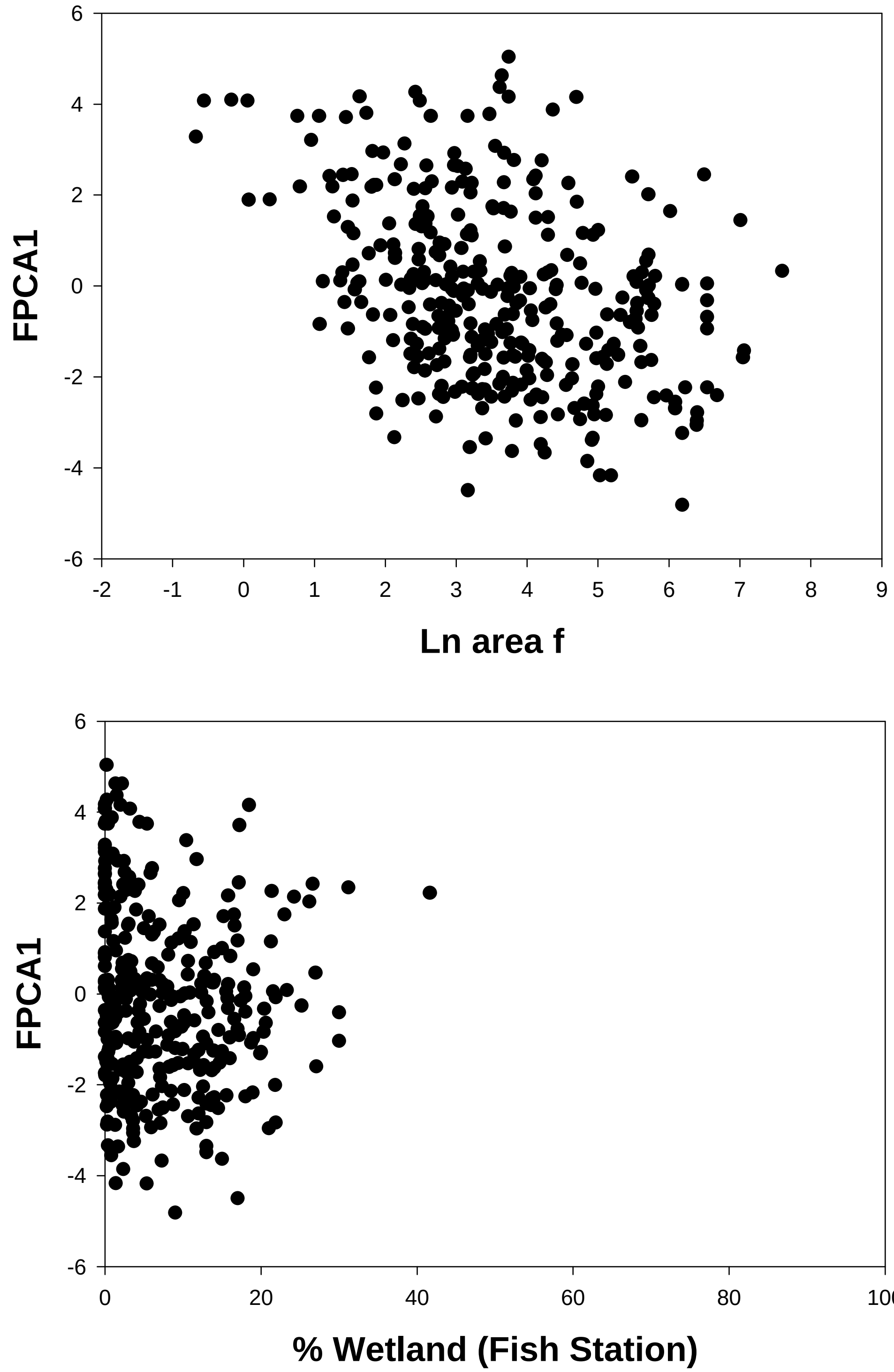


Figure 3.3.15. Relationship between Fpca1 (first fish principal component) and ln(watershed area) and square root of proportion wetland (bottom) for joint fish-macroinvertebrate sites.

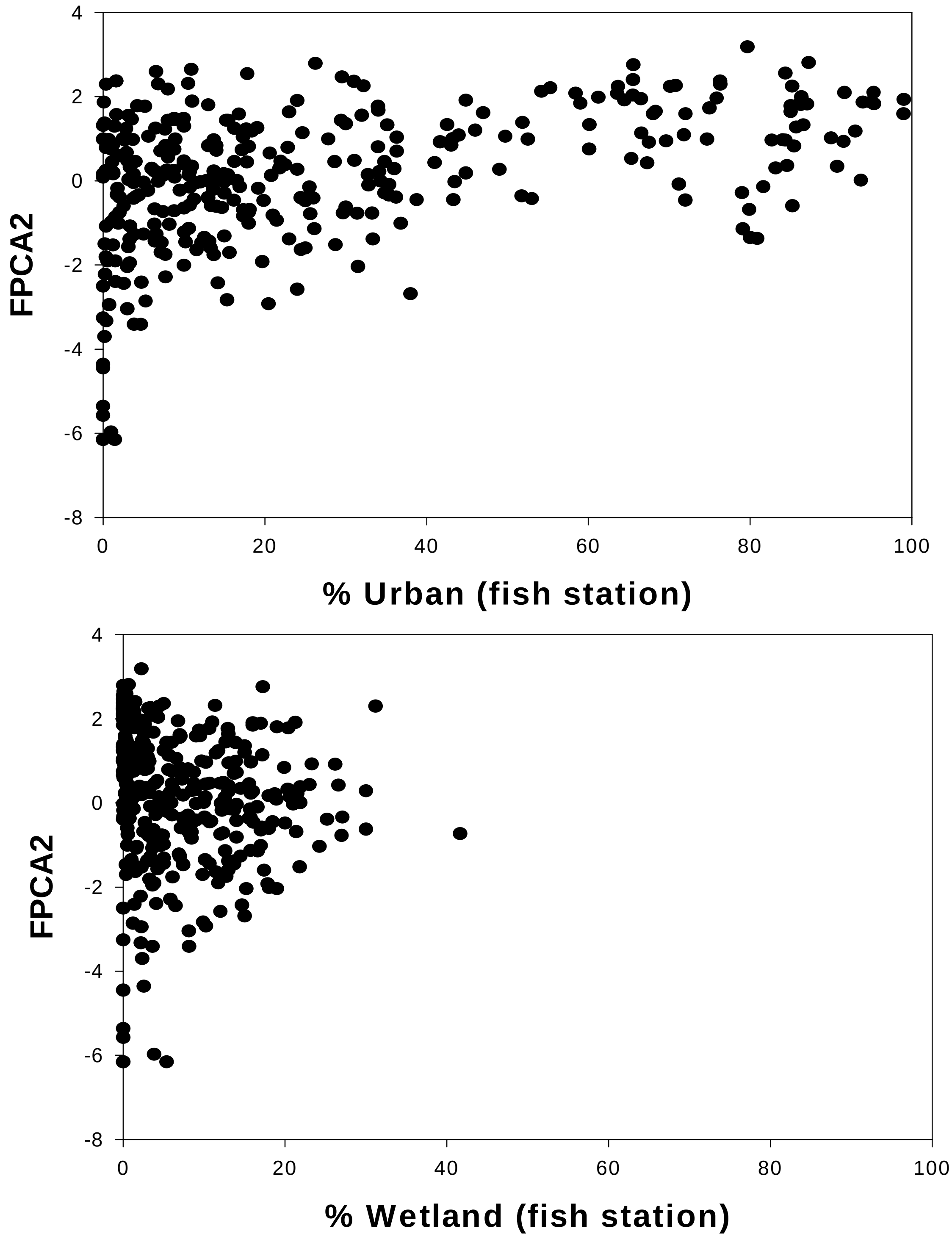


Figure 3.3.16. Relationship between Fpca2 (second fish principal component) and square root of proportion urban and square root of proportion wetland-open water (bottom) for joint fish-macroinvertebrate sites.

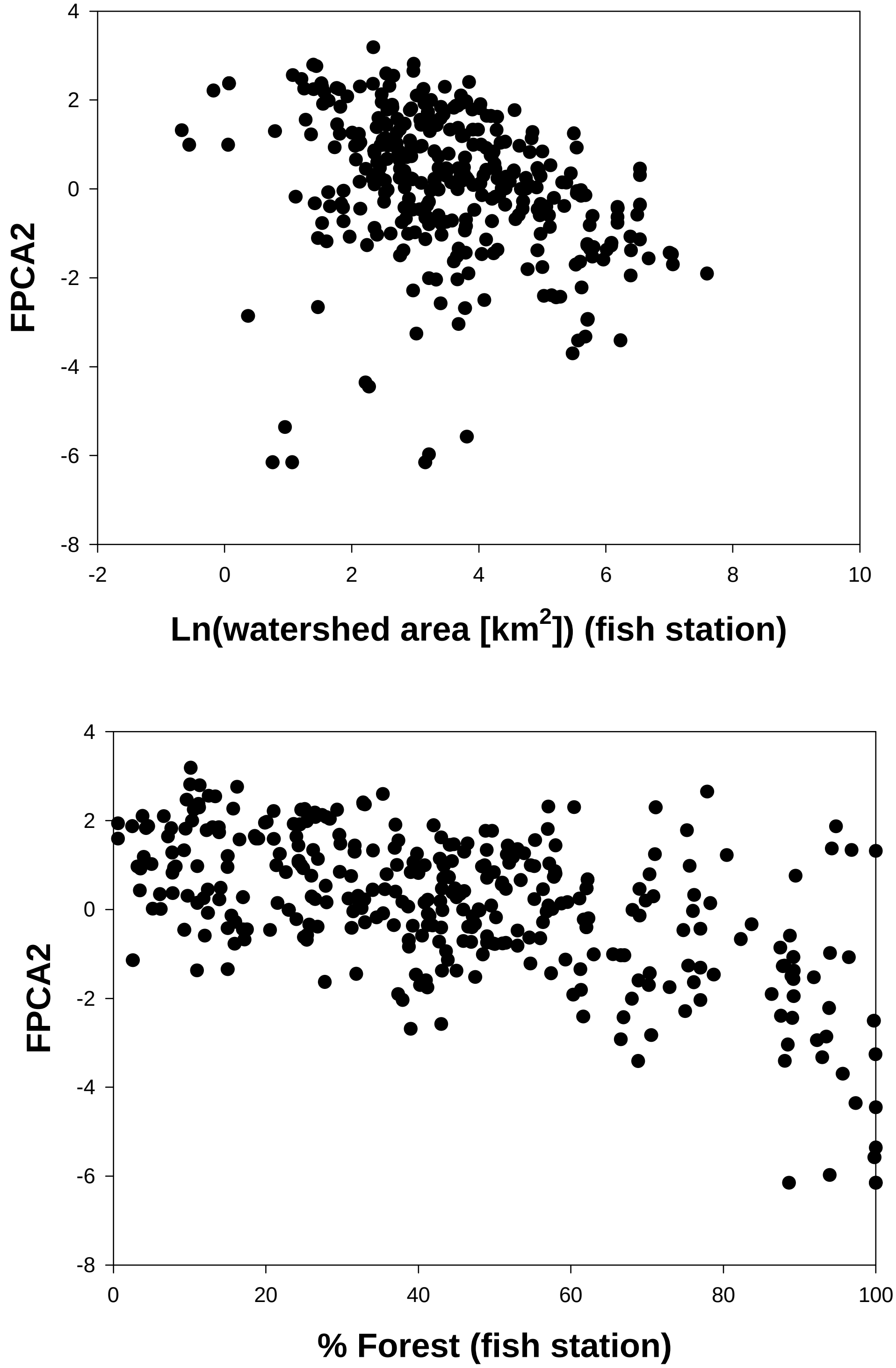


Figure 3.3.17. Relationship between Fpca2 (second fish principal component) and ln(watershed area) (top) and %forest (bottom) for joint fish-macroinvertebrate sites.

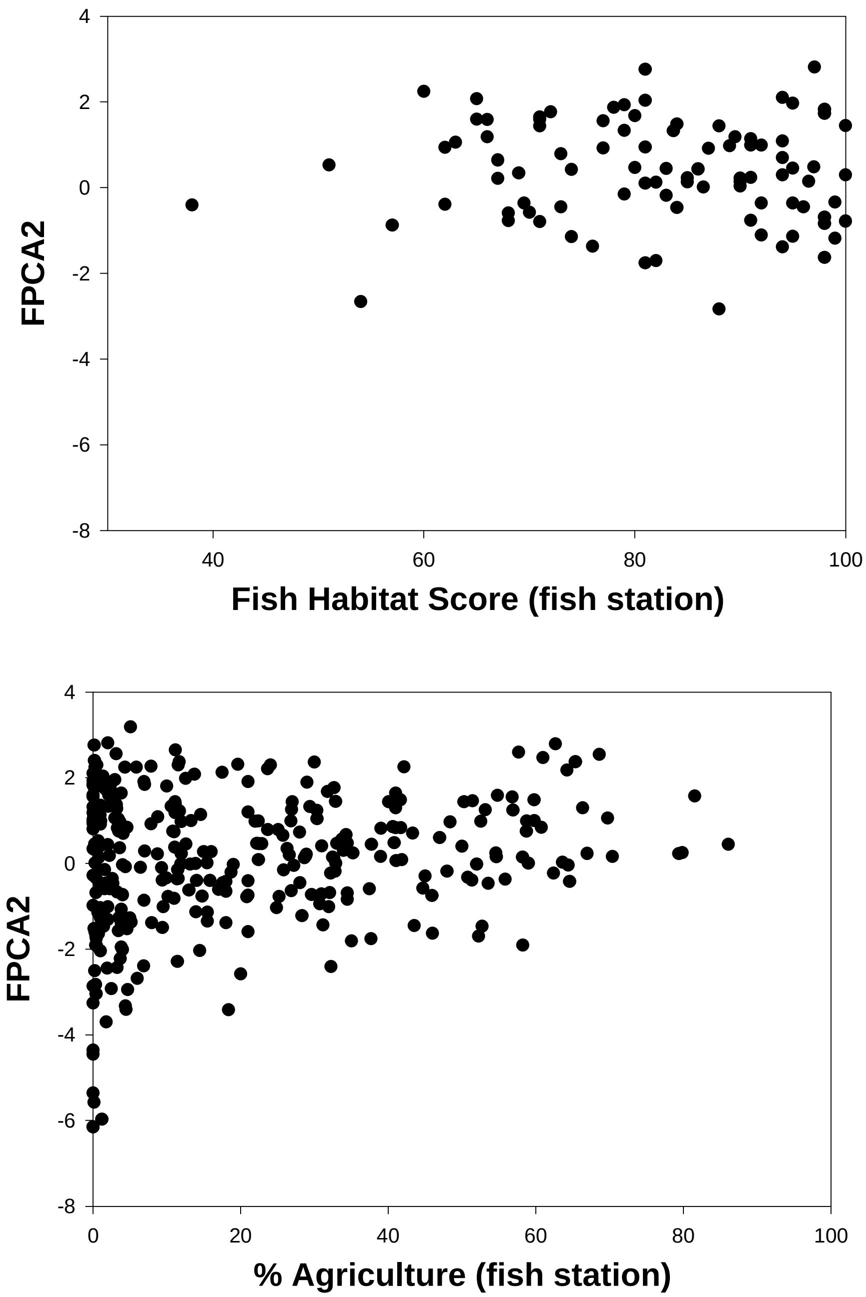


Figure 3.3.18. Relationship between Fpca2 (second fish principal component) and fish habitat score (top) and %agriculture (bottom) for joint fish-macroinvertebrate sites.

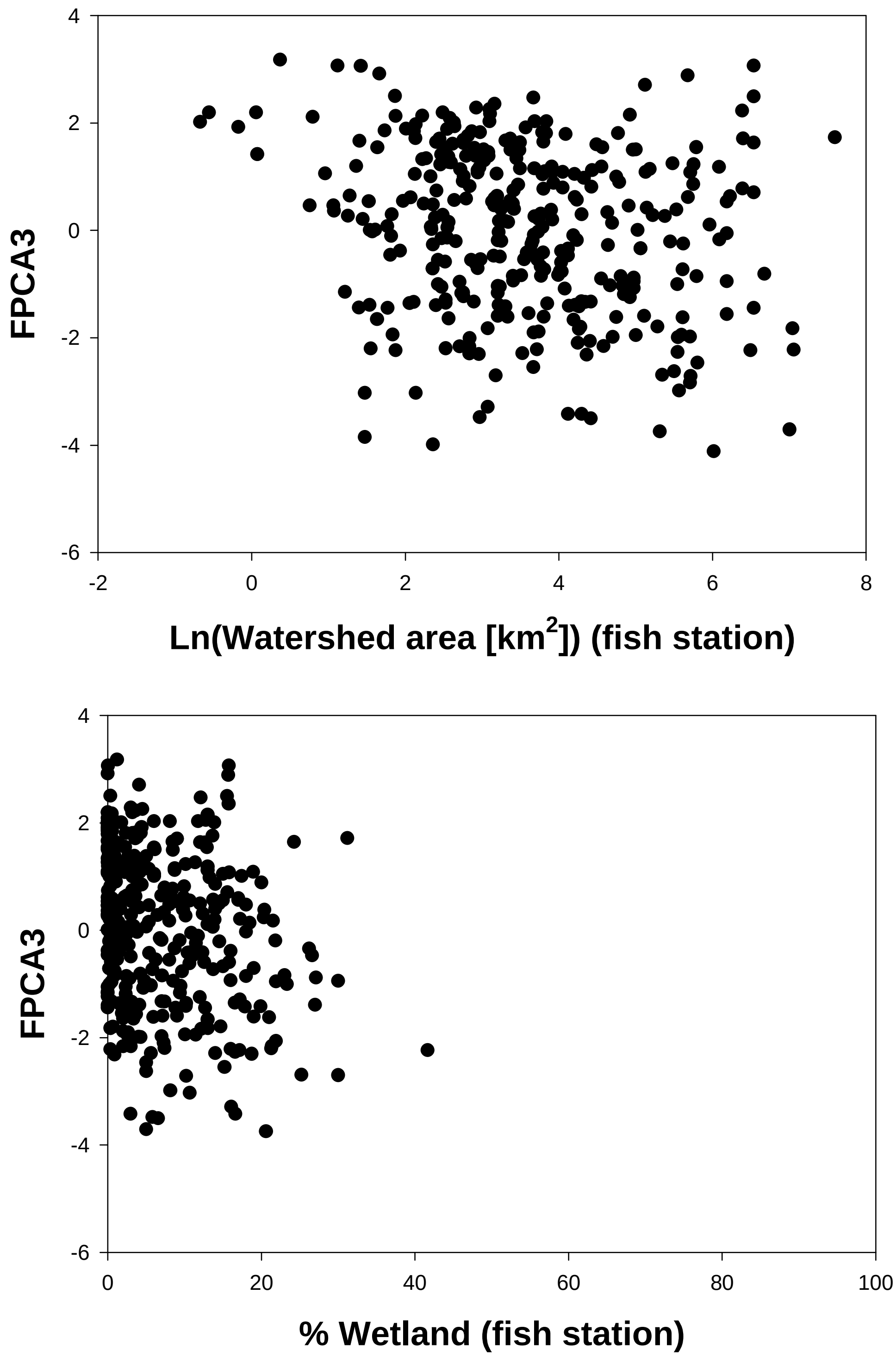


Figure 3.3.19. Relationship between Fpca3 (third fish principal component) and ln(watershed area) (top) and %urban (bottom) for joint fish-macroinvertebrate sites.

3.3.3 Fish-Macroinvertebrate Relationships

3.3.3.1 Relationship between NJ FIBI and AMNET Scores

A separate analysis of the NJ FIBI sites is directly relevant to use of the NJ FIBI data in watershed assessment. It also provides analysis of data from a subregion of the entire subregion collected by the same methods. Thus, the analysis reduces some of the variance resulting from spatial and methodological differences. The relationship between the fish metrics which form the NJ FIBI and the macroinvertebrate metrics which form the AMNET were analyzed for the NJ FIBI sites. AMNET sites were matched with 2000-2003 NJ FIBI sites. NJ FIBI sites were selected to be at AMNET sites. However, for some stations, a nearby site was chosen where the immediate AMNET site was not suitable for fish sampling (often lacking a pool-riffle-run habitat mix). Sufficient information was compiled for 66 sites with fish, macroinvertebrate, land use and associated habitat information. In addition to the overall FIBI score (available in the NJ database supplied by NJ DEP), individual FIBI metrics (excluding number of anomalies) were calculated. The ratio of observed to watershed-sized predicted species richness was calculated for the four fish richness metrics. The five AMNET metrics were calculated using the random subsetting procedure as discussed in the section on macroinvertebrate metrics. In analyses involving watershed characteristics (size and land use), the data for the NJ FIBI site were used.

The overall NJ FIBI and AMNET scores (Fig. 3.3.20) were significantly related, but with high variance (standard regression $p<0.0007$, adjusted r^2 of 0.15). The rank ordering was similarly correlated (Spearman r^2 of 0.17). One point, from the Elizabeth River, had a very low FIBI score, but without a correspondingly low AMNET score. Excluding this point increases r^2 slightly (Pearson to 0.19, Spearman to 0.18). The relationship is roughly triangular, i.e., sites with low AMNET scores tend to have low FIBI scores, as well, while sites with high AMNET scores had a range of FIBI scores, spanning virtually the entire range of FIBI scores.

For the NJ FIBI sites, correlations between individual macroinvertebrate and fish metrics were low to moderate (Table 3.3.8), except for the Number of intolerant fish species metric (Rintol) and EPT family richness (r^2 of 0.52). The macroinvertebrate metrics were generally highly intercorrelated, especially %EPT, EPT family richness and Average tolerance. Correlations among fish metrics were also relatively low, except for high correlations among three of the richness metrics (Rnspec, Rbi, and Rsalcent). Rbi was moderately correlated (r^2 of 0.47) with number of individuals in the sample.

Both the FIBI and AMNET scoring systems assume monotonic relationships between each metric and integrity, with higher values indicating higher quality for most metrics, and lower values indicating higher quality for Average tolerance, %Dominant family, %White sucker, and %Generalists. Thus, positive correlations would be expected between metrics scoring in the same direction, and negative for those scoring in the opposite direction. All macroinvertebrate metrics were correlated in the expected direction. Most of the among-fish correlations were also in the expected direction. However, several, especially those involving Rsalcent, were in the opposite direction (Table 3.3.8). For example, Rsalcent was negatively correlated (r^2 of -0.48) with %insectivorous cyprinids. Correlations among fish and macroinvertebrate metrics were also mainly in the expected direction, except for those involving Rsalcent, and the Rnspec-%EPT and the Rbi-%EPT correlations.

The macroinvertebrate metrics were generally weakly correlated with the FIBI score (Figs. 3.3.21-3.3.23). Regressions of these metrics on the FIBI score were weakly signifiant

($0.01<p<0.05$) for %Dominant family, %EPT and Average tolerance, with r^2 values less than 0.10. The regression was moderately significant ($p<0.003$) for Family richness (r^2 of 0.12), and highly significant for EPT family richness ($p<0.0003$, r^2 of 0.24). The total AMNET score (Fig. 3.3.24) was correlated with land use (%urban), when ln(watershed area) was included in the regression (although the area effect was not significant). Most of the macroinvertebrate metrics were correlated with land use (Figs. 3.3.25-3.3.27). EPT family richness was significantly ($p<0.000001$, $r^2=0.36$) correlated with %urban; family richness was significantly ($p<0.004$, r^2 of 0.11) correlated with % urban; and %EPT was significantly ($p<0.00003$, r^2 of 0.22) correlated with % forest. %Dominant family and Average tolerance were not correlated with %urban or % forest. None of the metrics was significantly related to ln(watershed area), even in models containing land use as well.

The IBI score was generally correlated with land use (Fig. 3.3.28). Very urban sites had low IBI scores, although the second-lowest score was seen at a relatively low urban site (a site on the Musconetcong River (FIBI061). There was high variability in scores among sites of intermediate and low urbanization, although the highest scores were seen for sites with less than about 25% urban land in the drainage. The IBI score is also correlated with watershed area (Fig. 3.3.28). Stepwise multiple linear regression (with ln(watershed area), %urban, %forest, %agriculture, habitat score and %pool as independent variables) showed highly significant relationships with %urban ($p<0.000001$) and ln(watershed area) ($p<0.002$), with model r^2 of 0.32. Similar stepwise regression models of the individual fish metrics generally showed significant relationships with %urban (Rnspec and Rbi) or %forest (Rintol). Several metrics (Rnspec, Rbi, and %Insectivorous cyprinids) were significantly correlated with ln(watershed area). Adjusted model r^2 for these metrics ranged from 0.24-0.36. Significant models were not seen for Rsalcent, %White sucker, and %Generalists.

The patterns of relationship between individual metrics and watershed size and land use suggest several possible reasons for the observed relationship between FIBI scores and AMNET scores (Fig. 3.3.20). The lack of correlation of %Dominant Family with other macroinvertebrate metrics or with land use suggests that it may not be a sensitive indicator of condition. Its inclusion in the AMNET score may introduce variability into the macroinvertebrate rating. Among fish metrics, the relationships of two of the watershed-area adjusted fish richness metrics (Rnspec and Rbi) indicates that the watershed-size adjustment does not completely remove the watershed-area dependence. Examination of scores for specific sites indicates that observed species richness values in small streams are usually within or above the highest quality group (i.e., scored as a 5). This probably reflects an increase in abundance of some species (many of which are insectivorous cyprinids and/or benthic invertivores) in small streams with urbanization (see section 3.4.1). Lower scores were found for some larger streams and rivers. As a result, small streams may be scored relatively highly on these metrics, while AMNET may indicate impairment. The relative abundance of some species of insectivorous cyprinids (e.g., the blacknose dace *Rhinichthys atratulus*) may decrease in larger streams even without impairment, due to decreased absolute abundance or increase in other groups such as centrarchids. Since there is no watershed area adjustment for %insectivorous cyprinids in the FIBI, larger streams may receive lower scores for this metric. The negative correlation of Rsalcent with other metrics reflects the increase in number of centrarchids in larger, more urban streams (since the most undeveloped streams are usually small, it is difficult to separate effects of stream size and urbanization for larger streams). This metric may compensate for lower values of other metrics in larger, developed streams. Examination of the relative abundance of species in samples indicates other possible effects on the FIBI. Presence of coldwater fish such as trout generally increases the FIBI (through metrics for number of intolerants and %salmonids/topcarnivores). Some trout (especially brown trout *Salmo trutta* and rainbow trout *Onchorhynchus mykiss*) may survive as adults in small areas with inputs of cool water, while macroinvertebrate assemblages may reflect the general conditions more closely. Some species which appear relatively intolerant of urbanization, such as margined madtom *Noturus insignis* and shield darter *Percina peltata*, were found in some sites with relatively low total IBI scores (32-36, rated as fair), suggesting possible underestimation of condition at some sites.

Some discrepancies between IBI metrics and watershed characteristics may be related to calibration of the IBI metrics. Calibrations were based on older samples, including NJFG data (J. Kurtenbach, pers. comm.). While these samples used a similar basic protocol to the FIBI protocol, the FIBI sampling may be more efficient at documenting small, nongame species. As a result, the calibration of metrics which are sensitive to abundance and presence of these species may be imprecise. The FIBI protocol also seeks out sites with relatively high habitat complexity. While this may reduce variability and make the index more sensitive to some stressors, it may also underestimate typical levels of impairment and weaken the relationship between metrics and land use.

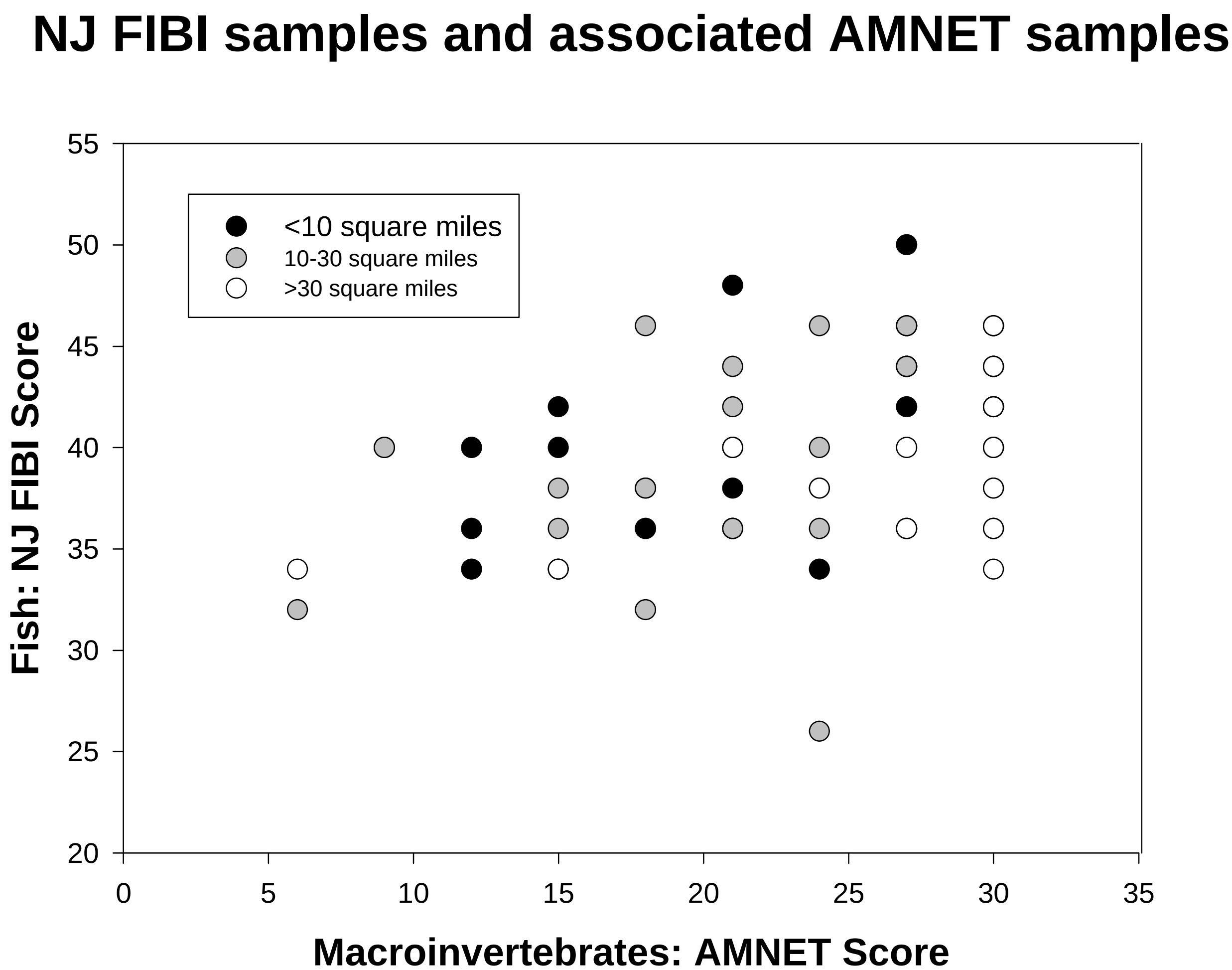


Figure 3.3.20. Relationship between AMNET and NJ FIBI score for NJ FIBI samples and associated AMNET sites.

Table 3.3.8. Pearson correlations among NJ FIBI fish metrics and AMNET macroinvertebrate metrics for 66 NJ FIBI sites. R metrics are the ratio of the observed species richness to the number predicted on the basis of watershed size. Correlations greater than 0.4 in absolute value are shown in bold. Correlations which are in the opposite direction of that assumed in the scoring of sites are underlined.

		DomFam	%EPT	AveTol	EPTfam	Famrich			
Macroinvertebrate metrics									
% Dominant family	DomFam								
% EPT	%EPT	-0.38							
Average tolerance	AveTol	**0.43**	-0.80						
EPT family richness	EPTfam	**-0.48**	**0.66**	**-0.66**					
Family richness	Famrich	**-0.58**	0.26	-0.36	0.70				
Fish metrics									
R Number of species	Rnspec	-0.04	-0.12	0.01	0.25	0.34			
R Number of benth. Invert. Spp.	Rbi	-0.12	-0.13	-0.05	0.27	0.35			
R Number of salcent species	Rsalcent	0.07	0.07	0.12	-0.21	-0.21			
R Number of intol species	Rintol	-0.39	0.25	-0.22	**0.52**	0.32			
%White sucker	%cacom	0.33	-0.27	0.16	-0.19	-0.29			
%Generalists	%gen	0.03	-0.19	0.22	-0.25	-0.05			
%Insectivorous cyprinids	%inscyp	-0.11	0.27	-0.27	0.30	0.10			
%Top Carnivores/salmonids	%topcarn	-0.12	0.26	-0.15	0.21	0.05			
Number of individuals	Nind	-0.04	0.01	-0.11	0.21	0.23			
		Rnspec	**Rbi**	**Rsalcent**	**Rintol**	**%cacom**	**%gen**	**%inscyp**	**%topcarn**
R Number of benth. Invert. Spp.	Rbi	**0.79**							
R Number of salcent species	Rsalcent	-0.19	**-0.68**						
R Number of intol species	Rintol	0.24	0.29	-0.23					
%White sucker	%cacom	0.10	0.13	-0.17	-0.10				
%Generalists	%gen	-0.04	0.03	-0.02	-0.29	-0.14			
%Insectivorous cyprinids	%inscyp	0.16	0.39	**-0.48**	0.38	-0.03	-0.01		
%Top Carnivores/salmonids	%topcarn	0.08	-0.15	0.30	0.35	-0.14	-0.09	-0.17	
Number of individuals	Nind	0.34	**0.47**	-0.31	0.03	0.21	-0.06	0.31	-0.14

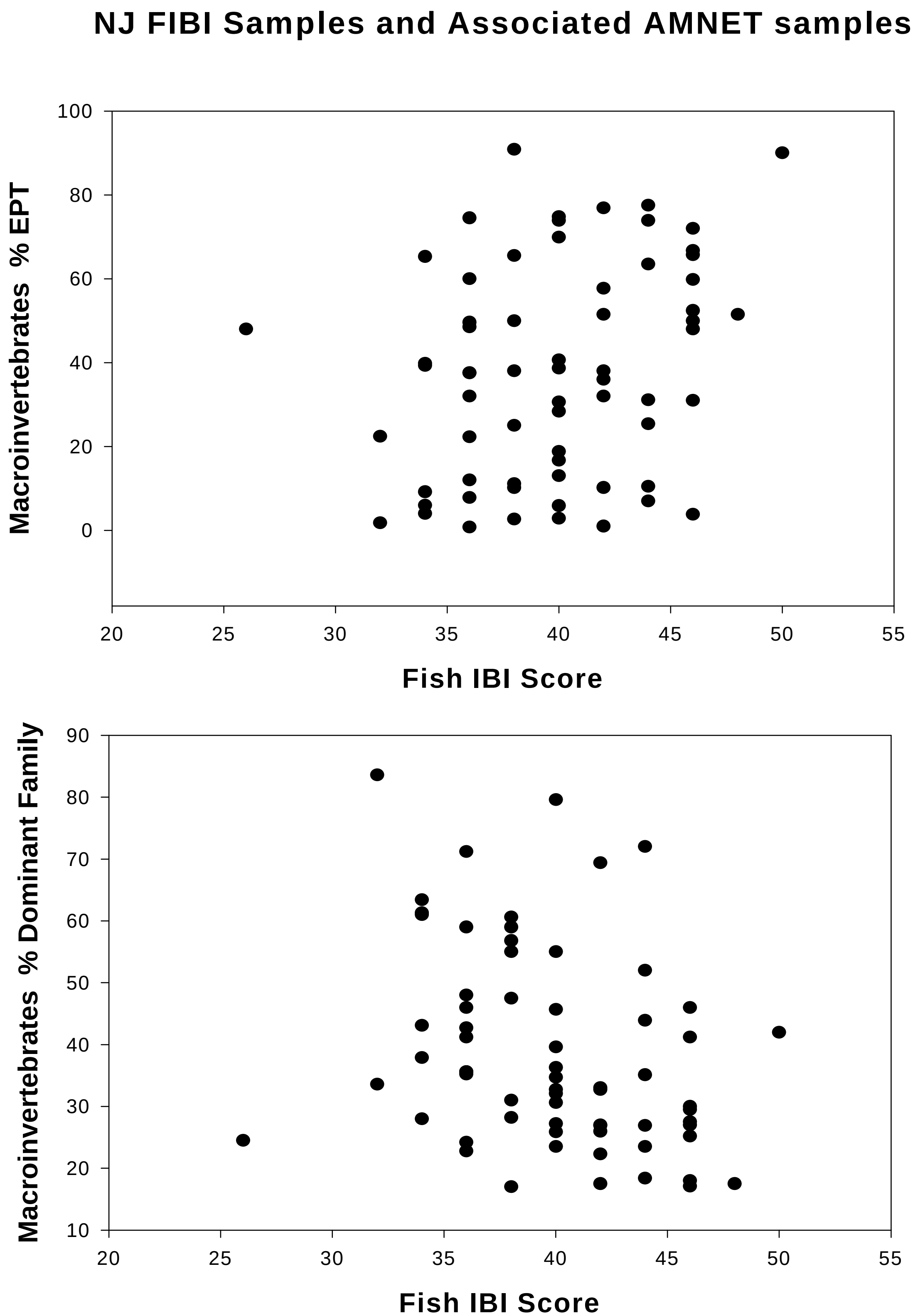


Figure 3.3.21. Relationship between NJ FIBI score and %EPT (top) and%DomFam (bottom) for NJ FIBI samples and associated AMNET sites.

NJ FIBI Samples and Associated AMNET Samples

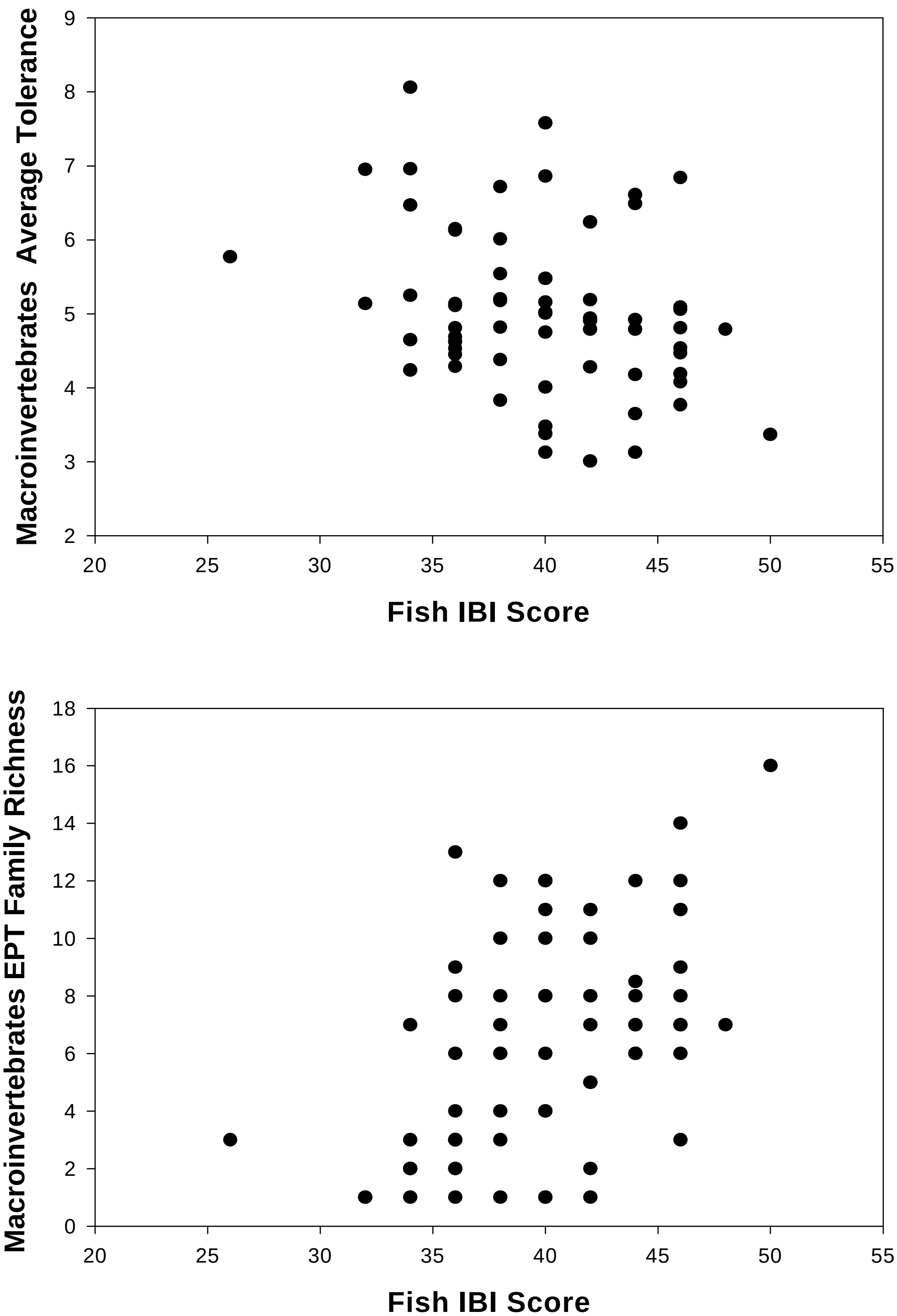
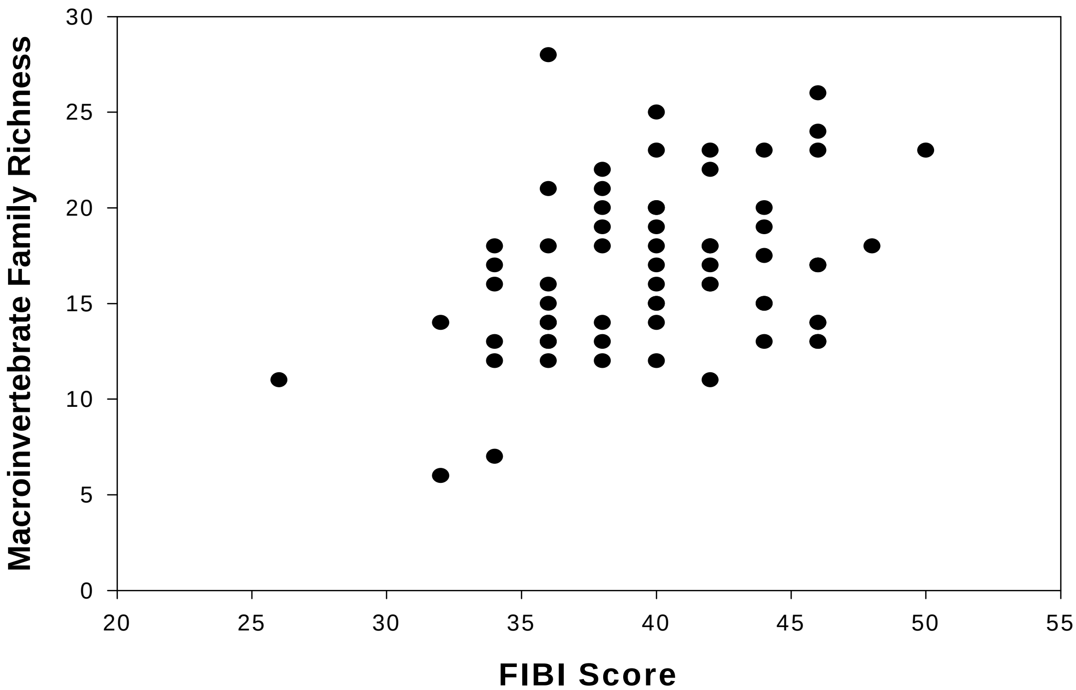


Figure 3.3.22. Relationship between NJ FIBI score and Average tolerance (top) and EPT family richness (bottom) for NJ FIBI samples and associated AMNET sites.

NJ FIBI Samples and Associated AMNET Samples



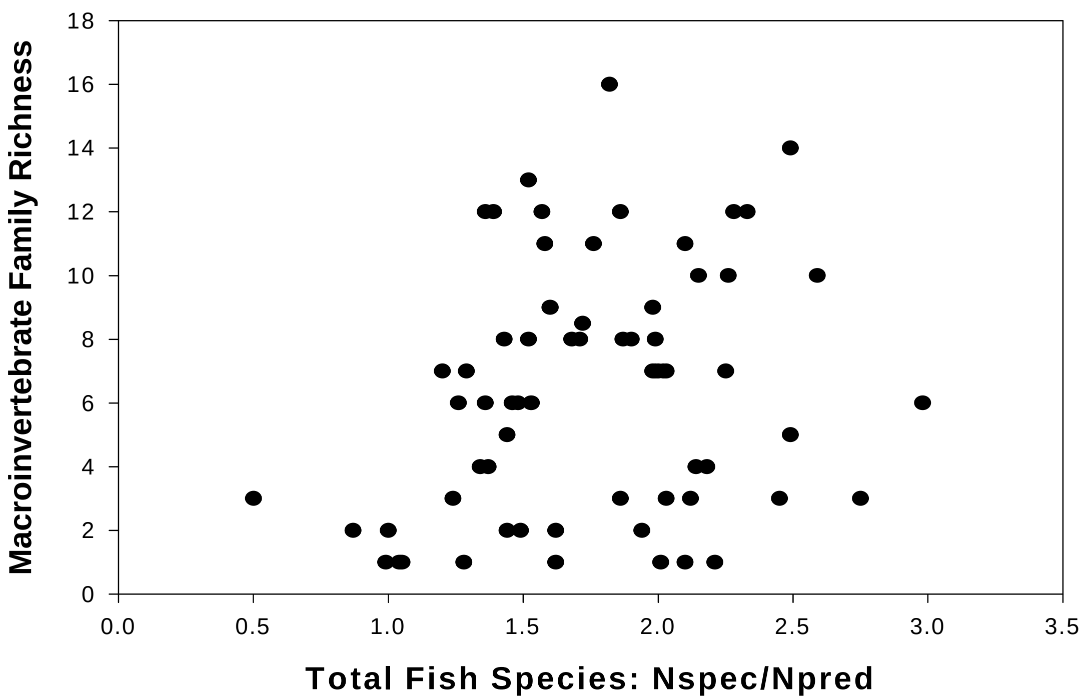


Figure 3.3.23. Relationship between macroinvertebrate family richness and IBI score (top) and Rnspec (watershed-area adjusted total number of fish species) (bottom) for NJ FIBI samples and associated AMNET sites.

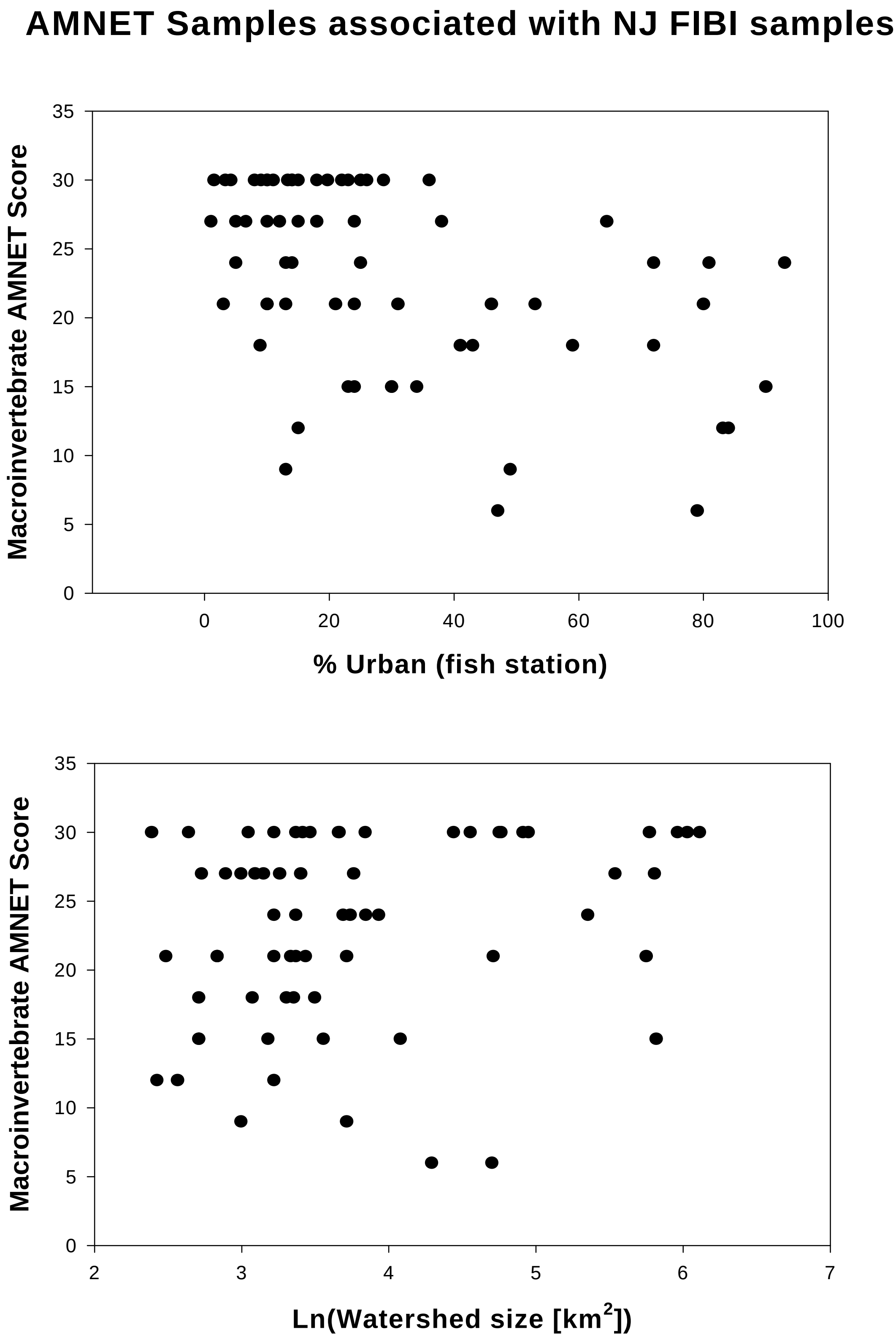


Figure 3.3.24. Relationship between AMNET score and %Urban (top) and watershed area (bottom).

AMNET samples associated with NJ FIBI samples

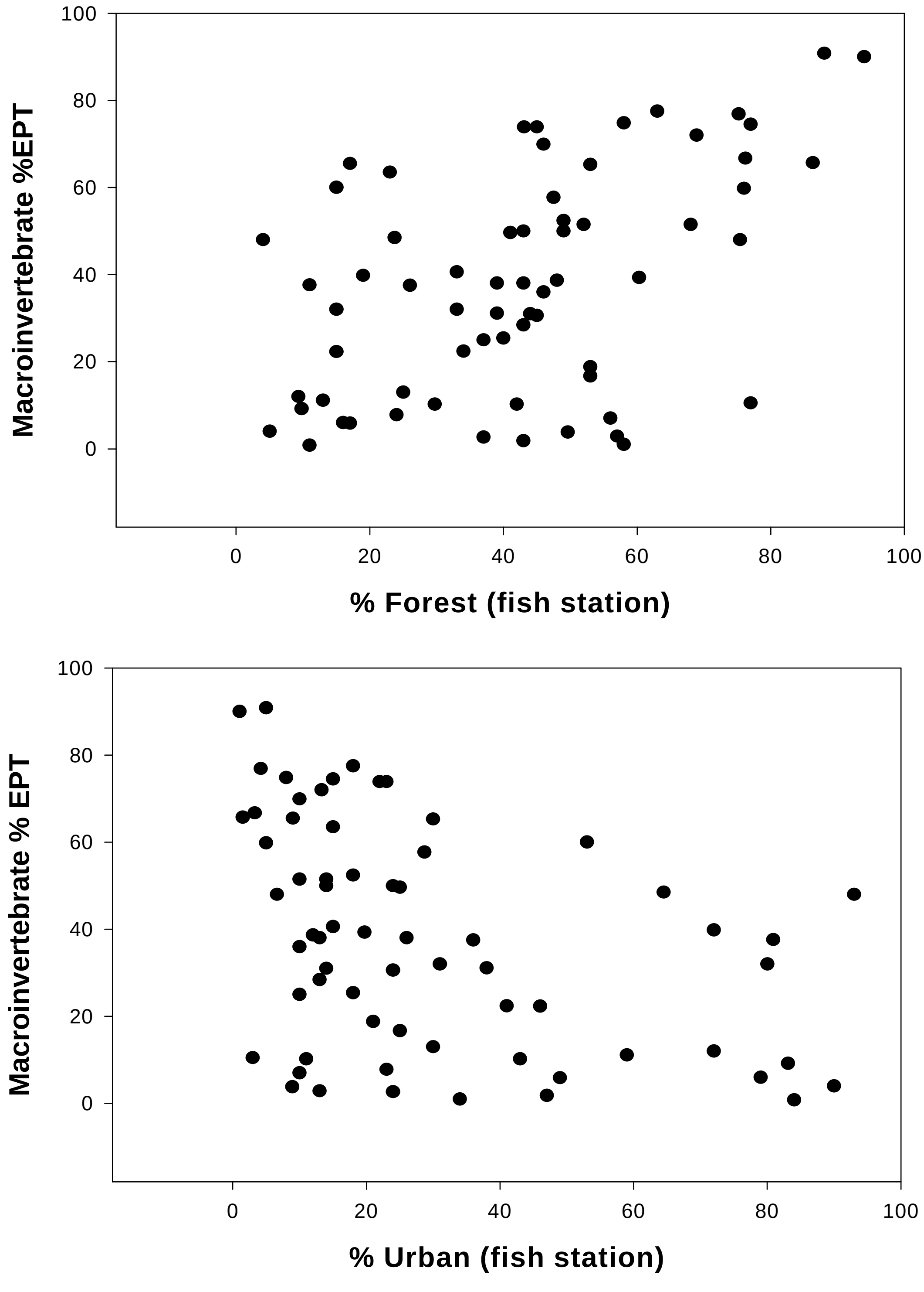
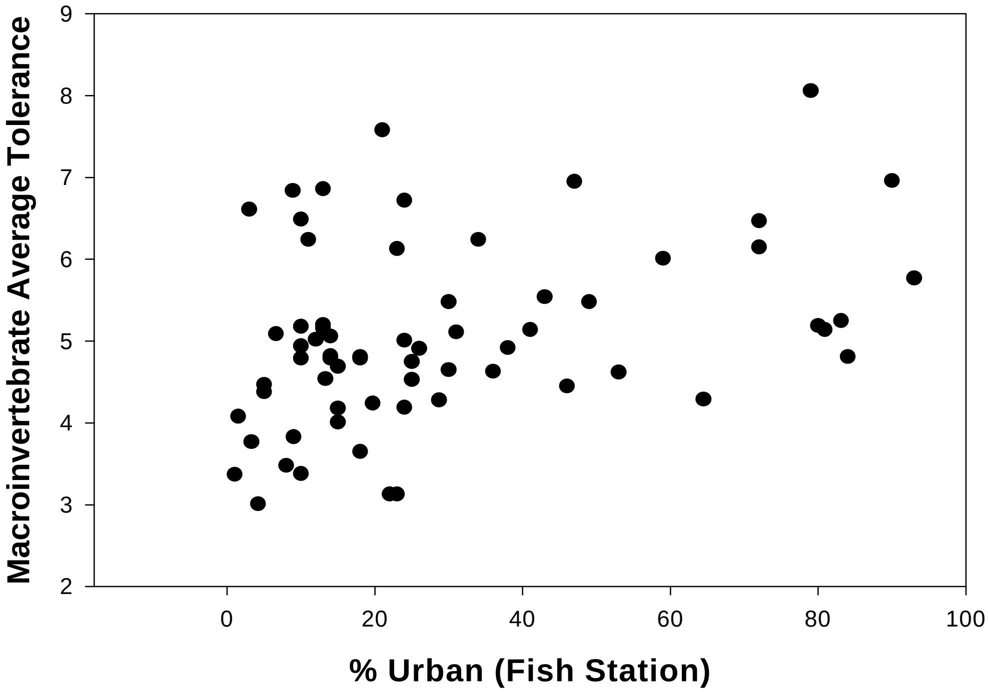


Figure 3.3.25. Relationship between %EPT and %urban (top) and %forest (bottom) for NJ FIBI samples and associated AMNET sites.

AMNET samples associated with NJ FIBI samples



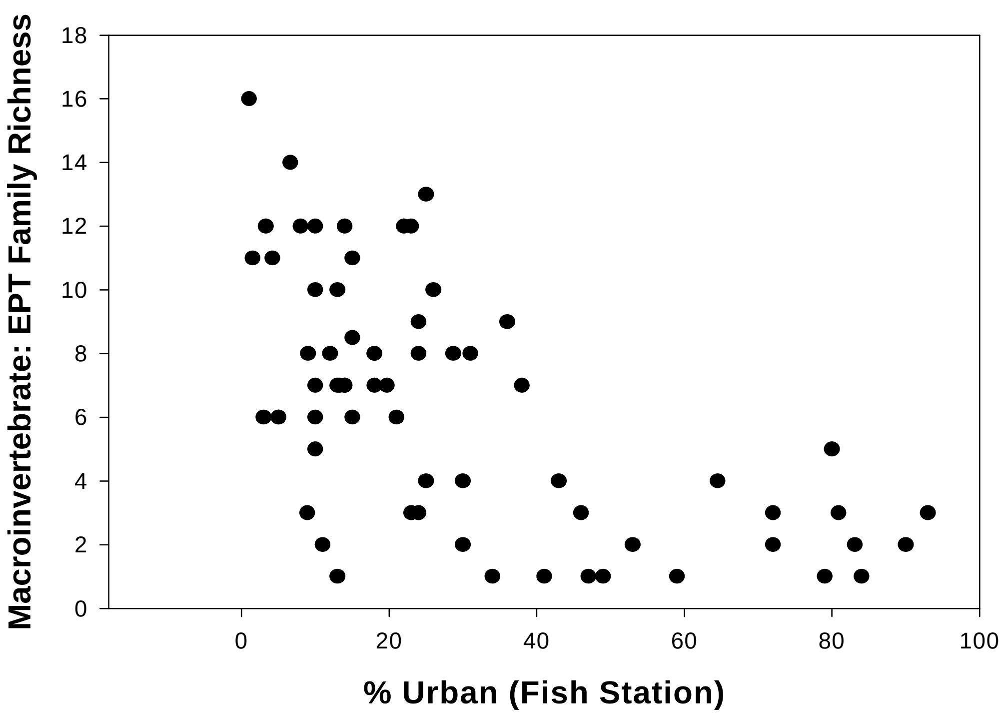
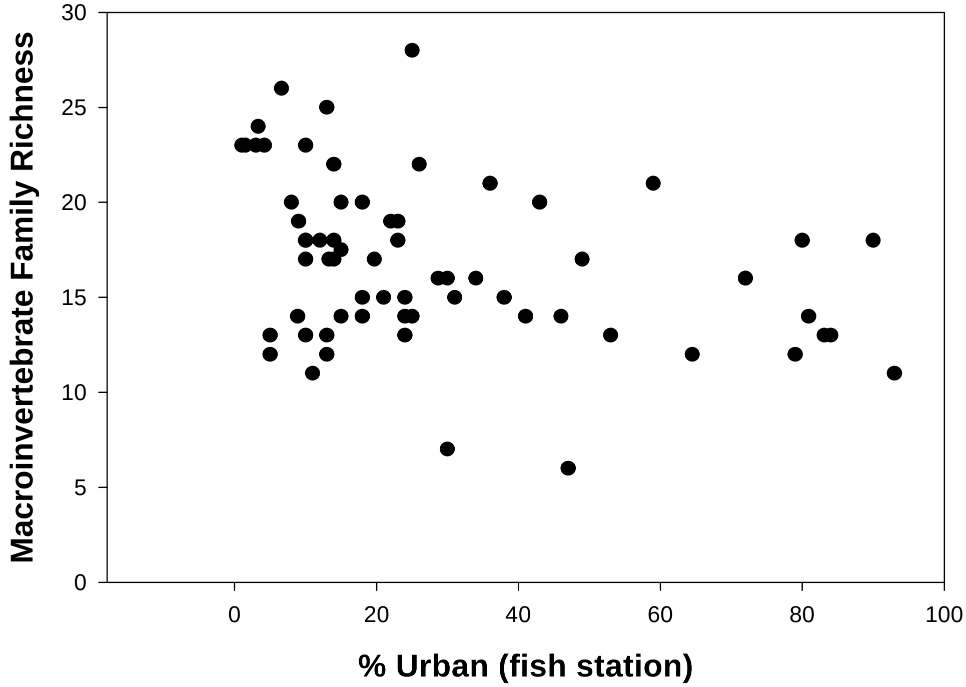


Figure 3.3.26. Relationship between %urban and Average tolerance (top) and EPT family richness (bottom) for NJ FIBI samples and associated AMNET sites.

NJ FIBI samples and associated AMNET samples



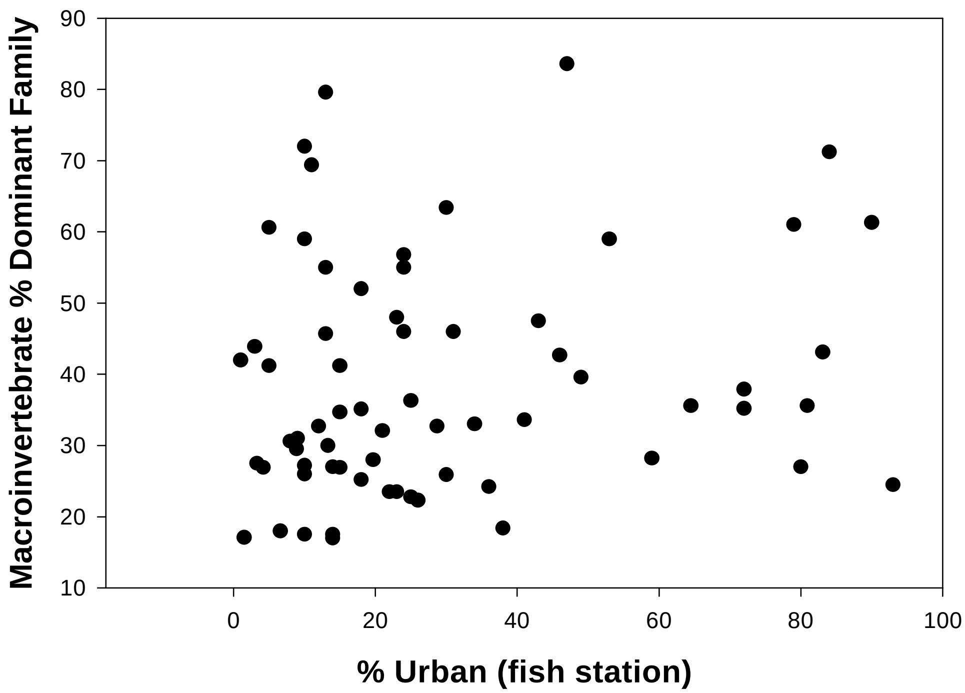
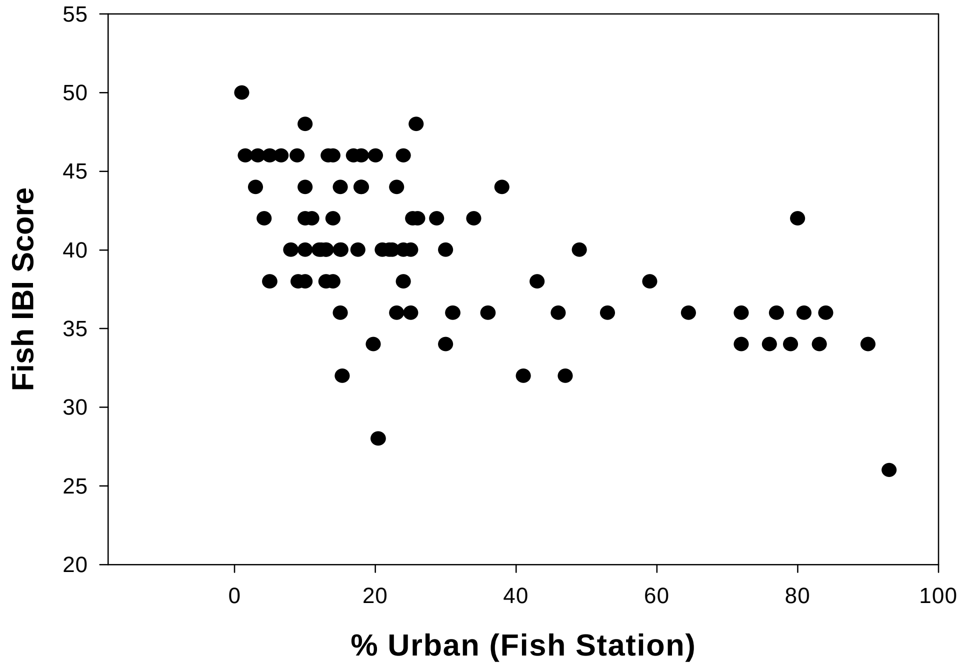


Figure 3.3.27. Relationship between %urban and macroinvertebrate family richness (top) and percentage dominant macroinvertebrate family (bottom) for NJ FIBI samples and associated AMNET sites.

NJ FIBI samples



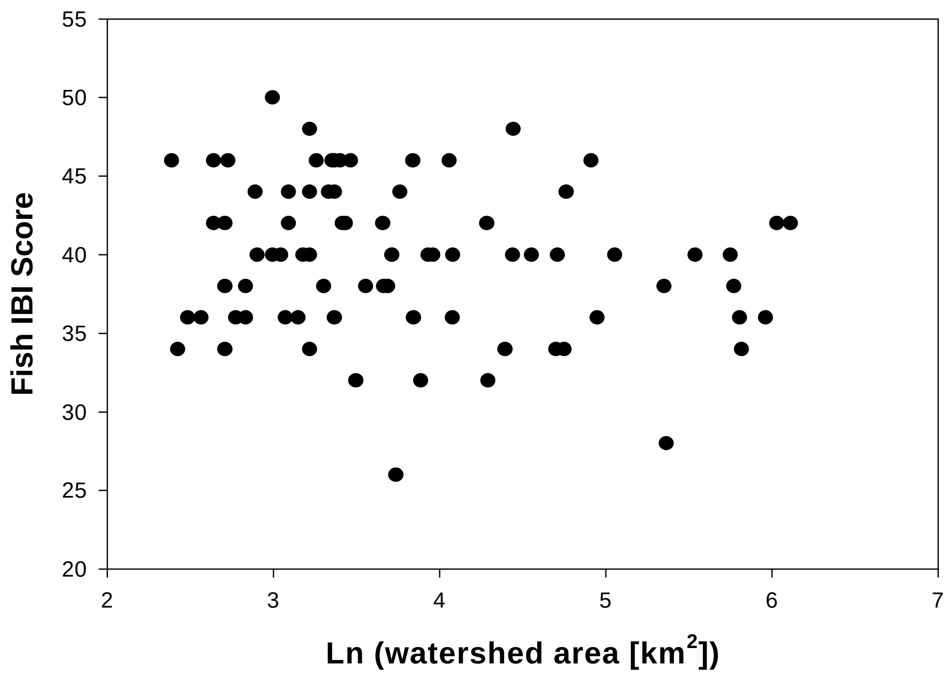


Figure 3.3.28. Relationship between Rbi (watershed-area adjusted number of benthic invertivorous fish species) and %urban (top) and ln(watershed area) (bottom) for NJ FIBI samples and associated AMNET sites.

3.3.3.2 Fish-Macroinvertebrate Principal Components Correlations

The analyses of macroinvertebrate and fish metrics found correlations among metrics and relationships between metrics and land use which indicate response to different aspects of watersheds. As a result, high correlations are only expected among fish and macroinvertebrate metrics responding to the same watershed conditions. Given the intercorrelations among macroinvertebrate metrics and among fish metrics, the inter-taxa correlations were investigated using the principal components described above.

Multiple regression was used to analyze these potential relationships among macroinvertebrate and fish PCs. Regressions of the fish principal components indicated significant relationships with macroinvertebrate principal components, watershed area, land use, and fish habitat scores. For some of these, the relationships with the macroinvertebrate principal components were better than those with land use, indicating that the Fpca-Mpca relationships were not just driven by joint relationships with land use.

Fpca1 was most strongly correlated (Table 3.3.9) with ln(watershed area). It was also correlated with Mpca1, but not with Mpca2-Mpca4. Model fit with Mpca1 was improved when the regression included only stations where the macroinvertebrate and fish samples were taken at the same site (i.e., less than 0.05 km apart). Fpca1 was correlated with fish habitat score, although it improved model fit only slightly over models containing ln(watershed area) and Mpca1. Fpca1 was weakly correlated with urban land (as square root of proportion urban land), but only when habitat score was not included in the model. The correlation with Mpca1 was driven by relationships at low levels of urbanization. Mpca1 was not significant in regressions done only on stations with urbanization > 50%. Ln(watershed area) was highly significant in subsets of data with high or low urbanization, but the slope of the regression was steeper (more negative) for low urban sites.

Fpca2 is significantly correlated with the first four macroinvertebrate principal components (Mpca1-Mpca4) and with ln(watershed area). Although significant, the watershed area relationship is not as strong as that of the Mpc's. Model fit is better for sites where the fish and macroinvertebrate samples were taken very close to each other (adjusted $r2= 0.38$). However, land use (specifically, urban and agricultural land) provides better model fit. A model with ln(watershed area), square root urban land and square root agricultural land has an r2 of 0.48 for stations where the taxonomic pairs were close to each other. Habitat score was not correlated with Fpca2.

Fpca3 is correlated with ln(watershed area) and with the first and third macroinvertebrate principal components. However, Fpca3 is better correlated with square root of forest and square root of wetland-open water. The slope of these two land use relationships are opposite, reflecting the gradient in Fpca3. High vales of Fpca3 were correlated with high proportions of insectivorous cyprinids and riffle species, while low values were correlated with abundance of centrarchids and pool species. The relationship with wetland-open water land use is consistent with lakes, impoundments and wetlands as sources for some fish species.

Fpca4 was primarily related to two urban sites with fish assemblages very different from those of all other samples. Mummichog was the most common species at these sites, with a few individuals of a few other species also present. This gradient is related to land use, but since it is controlled by a few points, statistical analyses of relationships with land use or macroinvertebrate principal components is not appropriate.

Table 3.3.9. Results of alternate linear regression models of fish principal components for joint fish-macroinvertebrate data set and macroinvertebrate principal components and watershed characteristics. Columns for independent variables show p-values and slope estimates. All models are significant at level p<0.00001. Fish habitat scores were not available for all sites, so models including fish habitat have fewer points; for comparison, some models are done over the same set of data.

Subset of data	r^2	adjusted r^2	n	ln(area)	Mpca1	Mpca2	Mpca3	Mpca4	square root urban	square root forest	square root ag	square root wetland	fhab
FPCA1													
all	0.078	0.075	338		0.000000								
					0.243165								
all	0.226	0.221	338	0.000000	0.000001								
				-0.548049	0.209190								
all	0.169	0.167	338	0.000000									
				-0.588733									
all	0.181	0.176	338	0.000000					0.031932				
				-0.574989					0.774079				
fhab>0	0.170	0.164	272	0.000000									0.000053
				-0.546893									-0.012857
fhab>0	0.160	1.5	272	0.000000					0.144285				0.003518
				-0.538735					0.648863				-0.010421
fhab>0	0.177	0.167	272	0.000000	0.004947								0.011492
				-0.429031	0.139018								-0.008405
fhab>0	0.173	0.167	272	0.000000	0.000029								
				-0.504013	0.196413								
urban<80	0.239	0.234	305	0.000001	0.000001								
				-0.526893	0.207607								
	0.178	0.175	305	0.000000									
urban<80				-0.562005									
urban<20	0.143	0.138	190	0.000000									
				-0.478259									
urban<20	0.173	0.164	184	0.000001	0.009526								
				-0.488781	0.173498								
	0.183	0.171	73	0.000162									
urban>50				-0.831839									
prox<.05	0.201	0.197	220	0.000000									
				-0.582866									
prox<.05	0.264	0.257	220	0.000000	0.000024								
				-0.534724	0.218021								

Table 3.3.9 (continued). Results of alternate linear regression models of fish principal components for joint fish-macroinvertebrate data set and macroinvertebrate principal components and watershed characteristics. Columns for independent variables show p-values and slope estimates. All models are significant at level p<0.00001. Fish habitat scores were not available for all sites, so models including fish habitat have fewer points; for comparison, some models are done over the same set of data.

Subset of data	r^2	adjusted r^2	n	ln(area)	Mpca1	Mpca2	Mpca3	Mpca4	square root urban	square root forest	square root ag	square root wetland	fhab
FPCA2													
	0.193	0.188	338	0.000000	0.000000								
				-0.314538	0.228079								
	0.093	0.091	338	0.000000									
				-0.358896									
	0.110	0.11	338		0.000000								
					0.239003								
prox<.05	0.134	0.133	220		0.000000								
					0.285311								
prox<.05	0.210	0.203	220	0.000011	0.000000								
				-0.321427	0.254501								
	0.278	0.267	338	0.000000	0.000000	0.000605	0.000509	0.001885					
				-0.309374	0.238860	-0.162403	-0.184941	0.214427					
prox<.05	0.397	0.383	220	0.000000	0.000000	0.000003	0.000001	0.008438					
				-0.343869	0.304326	-0.287284	-0.366548	0.228904					
	0.468	0.453	338	0.000000	0.004405	0.007863	0.007817	0.004576	0.000000	0.124538	0.000000	0.398981	
				-0.30360	0.10273	-0.12120	-0.13327	0.17855	3.69841	1.17750	2.93446	0.49846	
	0.463	0.452	338	0.000000	0.003725	0.010268	0.002277	0.015122	0.000000		0.000000		
				-0.27051	0.10415	-0.10564	-0.14300	0.14541	2.83880		2.46385		
	0.416	0.411	338	0.000000					0.000000		0.000000		
				-0.27875					3.55035		2.57939		
prox<.05	0.490	0.482	220	0.000007					0.000000		0.000000		
				-0.26504					3.82744		2.93371		

Table 3.3.9 (continued). Results of alternate linear regression models of fish principal components for joint fish-macroinvertebrate data set and macroinvertebrate principal components and watershed characteristics. Columns for independent variables show p-values and slope estimates. All models are significant at level p<0.00001. Fish habitat scores were not available for all sites, so models including fish habitat have fewer points; for comparison, some models are done over the same set of data.

Subset of data	r^2	adjusted r^2	n	ln(area)	Mpca1	Mpca2	Mpca3	Mpca4	square root urban	square root forest	square root ag	square root wetland	fhab
FPCA3													
	0.219	0.207	338	0.000000	0.000000	0.513850	0.013229	0.065030					
				-0.347569	-0.218933	-0.029027	0.124266	-0.119978					
	0.210	0.203	338	0.000000	0.000000		0.011840						
				-0.356386	-0.217096		0.126485						
	0.249	0.242	338	0.000000						0.000000		0.000034	
	0.277	0.268	338	0.000000	0.000001					0.000241		0.000403	
				-0.32037	1.90463					-1.88673		-0.12380	

3.4 Algae Metrics-Complete Dataset

3.4.1 Habitat and Land Use Relationships

Correlation coefficients among algae sample land use, drainage area, and habitat score are shown in Table 3.4.1A. Because several samples did not have habitat scores, an additional correlation analysis was conducted excluding habitat scores from the model (Table 3.4.1B). In both analyses %urban was negatively correlated with %forest and %agriculture. EPA habitat score was negatively correlated with %urban and positively correlated with %forest, although the relationship was not strong.

Table 3.4.1. Pearson correlation coefficients of algae environmental variables.

A. Including EPA Habitat Score (n=179)

	ln (area)	sqrt (prop Urban)	sqrt (prop Forest)	sqrt (prop Ag)	sqrt (prop Wet)	EPA Habitat
ln (area)	1					
sqrt (prop Urban)	0.082	1				
sqrt (prop Forest)	-0.129	-0.643	1			
sqrt (prop Ag)	-0.125	-0.55	-0.194	1		
sqrt (prop Wet)	0.417	0.114	-0.226	-0.324	1	
EPA Habitat	-0.011	-0.471	0.459	0.099	-0.078	1

B. Excluding EPA Habitat Score (n=223)

	ln (area)	sqrt (prop Urban)	sqrt (prop Forest)	sqrt (prop Ag)	sqrt (prop Wet)
ln (area)	1				
sqrt (prop Urban)	-0.10025	1			
sqrt (prop Forest)	0.08164	-0.62076	1		
sqrt (prop Ag)	-0.08406	-0.53564	-0.26141	1	
sqrt (prop Wet)	0.34417	0.0616	-0.13638	-0.2923	1

3.4.2 Algal Metric Correlations

Only seven algae metrics were able to be examined with four metrics examining diversity (Diatom Species Richness, % Dominant Taxon, % Dominants, and Shannon-Wiener Diversity Index), and the three remaining metrics examining autecological characteristics (Siltation Index, Percent of *Achnanthidium minutissimum*, and Centrales:Pennales ratio). Pearson correlation analyses (n=242) showed that the Shannon-Wiener Diversity index was highly positively correlated with Diatom Species Richness and highly negatively correlated with % Dominant Taxon and % Dominants metrics (Table 3.4.2). As such, Shannon-Wiener Diversity index was excluded from PCA analyses.

Correlation coefficients between algae metrics and PCA axes are shown in Table 3.4.3. The first principal component (APCA1) showed strong positive correlations with Siltation Index (PCA correlation: r=0.9223), and negative correlations with percent of *Achnanthidium minutissimum* (PCA correlation: r=-0.5175) and Dominant taxa (PCA correlation: r=-0.4258) metrics (Fig. 3.4.1). This component can be interpreted as a sedimentation or substrate index. Samples with high values for APCA1 contain high Siltation Index values indicating that they contain relatively high proportions of motile diatoms which are able to move through and are associated with fine sediments. Samples with high APCA1 scores tended to have a relatively low %*A. minutissimum*, a taxa that requires hard substrates and is dominant in headwater systems. A regression model of land use, watershed area, and EPA habitat scores (n=158) with APCA1 showed no relationship with environmental variables and explained only 9.0% of APCA1 variation, although the overall model was significant (p=0.024). However, when EPA habitat score was eliminated from the model and the sample size increased (n=223), APCA1 showed significant negative relationships with watershed area (p<0.001) and sqrt (proportion forest) (p=0.028), and significant positive relationships with sqrt (proportion Ag) (p=0.028) and sqrt (proportion Urban) (p=0.013) (Fig. 3.4.2). The overall regression model was significant (p<0.001), but still explained relatively little variation in APCA1 (r^2=0.144).

Samples were strongly positively correlated with %*A. minutissimum* and weakly positively correlated with Siltation Index along the second principal component (APCA2) (Table 3.4.3) (Fig. 3.4.3). *Achnanthidium minutissimum* is an early-successional species (Peterson and Stevenson 1992) and often is the first to colonize river beds after scouring during spates. This taxa requires hard substrate for colonization and is typically abundant in headwater streams. In NJ, *A. minutissimum* is often associated with low nutrient streams (M. Potapova, personal communication) and was often the dominant species in Montana streams receiving mining discharge and other chemicals (Barbour et al. 1999). Some samples showed high %AM and Siltation Index values, and APCA2 scores. For example, two samples from Miry Run, a tributary of the Assunpink River in the Delaware River basin, had normalized %AM values of 8.36 and 3.88, and APCA2 scores of 3.34 and 7.56, respectively. Miry Run is a low gradient stream with altered channel flow, low frequency of riffles, and poor bank stability. These conditions are well indicated by this metric and PC scores. Relatively low sediment deposition was reflected in low Sedimentation Index scores for this site. In contrast to APCA1, the regression model of environmental variables including EPA Habitat score (n=158) was significant (p=0.011) and accounted for 10.4% of sample variation despite no significance among individual environmental variables (Fig. 3.4.4). When EPA Habitat score was removed to increase the sample size (n=223), the overall model was not significant and explained only 3.2% of variation in APCA2.

The third component (APCA3) showed a strong positive correlation with the ratio of centric to pennate diatoms, and weaker positive and negative relationships with Diatom Species Richness and Percent Dominants, respectively (Table 3.4.3) (Fig. 3.4.5). APCA3 can be interpreted as a stream size index. The ratio of centric to pennate diatoms generally increased with increasing APCA3 scores, which approximately shows proportion of planktonic taxa in the diatom community. Sites with particularly high APCA3 scores and corresponding high C:P values were Doctors Creek at Allentown NJ (APCA3=5.816), Little Neshaminy Creek (APCA3=5.177), and Cooper River at Haddonfield.(APCA3=4.39, 4.52). Typically, C:P is higher in systems containing more planktonic taxa such as streams draining lakes or reservoirs, or in larger rivers with a developed phytoplankton community. Additionally, %Dominants tends to decrease and Diatom Species Richness increases with increasing stream size. Regression analysis of APCA3 with environmental variables (including EPA Habitat score) was significant ($p<0.001$) and accounted for the greatest variability among samples of all PCA axes. There was a significant positive relationship with APCA3 and ln (area) ($p<0.001$) with the model explaining 14.2% of sample variation, although there was no significant relationship with land use variables or EPA Habitat scores (Fig. 3.4.6). The regression model not including EPA Habitat score was significant ($p=0.034$), but explained less ($r^2=0.032$) than the model including Habitat scores.

Principal component 4 (APCA4) showed strong relationships with diversity metrics (Table 3.4.3). Diatom species richness decreased, and % Dominant Taxon and % Dominants metrics increased with increasing APCA4 scores (Fig. 3.4.7). Higher values of Diatom Species Richness and lower values of % Dominants, and % Dominant Taxon indicate higher diversity. Diatom species diversity is usually highest at the intermediate level of disturbance irrespective of the type of disturbance. For example, among sites ranging from forested, nutrient-limited sites to severely polluted sites, the expected trend would be the greatest diversity in moderately nutrient-enriched sites. APCA4 may be related to nutrient concentrations in streams. The regression model relating APCA4 to environmental variables accounted for virtually none of the variability among samples in the model including ($r^2=0.027$) and excluding EPA Habitat scores ($r^2=0.020$) with neither showing significant relationships between the variables examined and samples scores.

Table 3.4.2. Pearson correlation coefficients of normalized algae metrics (n=242). Shaded areas represent correlation coefficients $0.7 or #-0.7.

		PCA Axis		
	APCA1	**APCA2**	**APCA3**	**APCA4**
EIG	0.4769	0.3283	0.1412	0.0371
Metric				
Diatom Species Richness	0.0707	-0.1393	0.4299	-0.6897
% Dominant Taxon	-0.4258	-0.0806	-0.233	0.6092
Percent Dominants	0.2097	0.1676	-0.4529	0.7735
Centrales:Pennales	0.0969	-0.1301	0.9762	0.1436
% *Achnanthidium minutissimum*	-0.5175	0.8531	0.0664	-0.0043
Siltation Index	0.9223	0.386	0.0005	-0.0057

Table 3.4.3. Algae metric correlations with PCA axes for all algae samples (n=242).

	Diatom Richness	**Shannon-Wiener Index**	**% Dominants**	**% Dominant Taxon**	**Centralles: Pennales**	**% AM**	**Siltation Index**
Diatom Richness	1						
Shannon-Weiner Index	0.821	1					
% Dominants	-0.539	-0.736	1				
% Dominant Taxon	-0.614	-0.818	0.297	1			
Centralles:Pennales	0.345	0.351	-0.178	-0.332	1		
% AM	-0.123	-0.088	0.134	0.003	-0.097	1	
Siltation Index	0.014	0.06	-0.421	0.246	0.039	-0.148	1

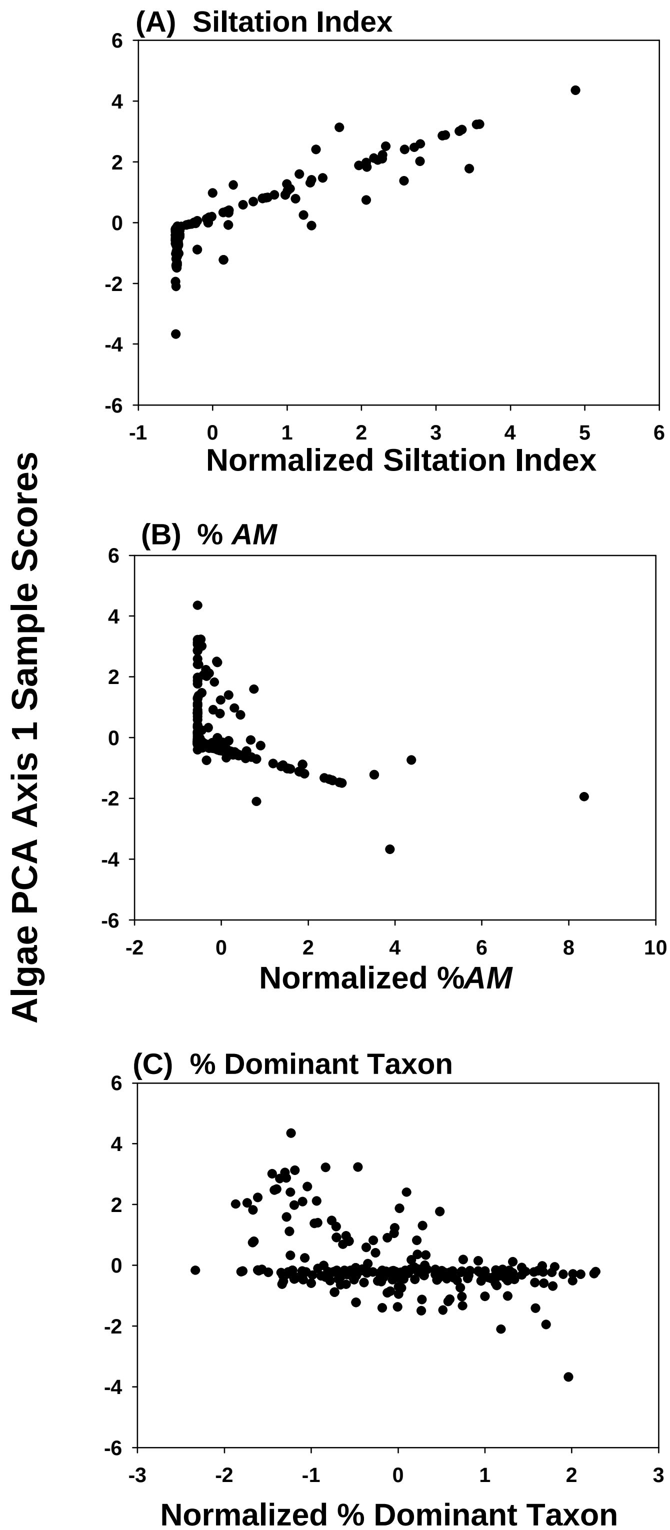


Figure 3.4.1. Relationship between APCA1 scores and normalized (A) Siltation Index, (B) %AM, and (C) %Dominant Taxon from algal samples in the Integrated Analyses dataset (n=242).

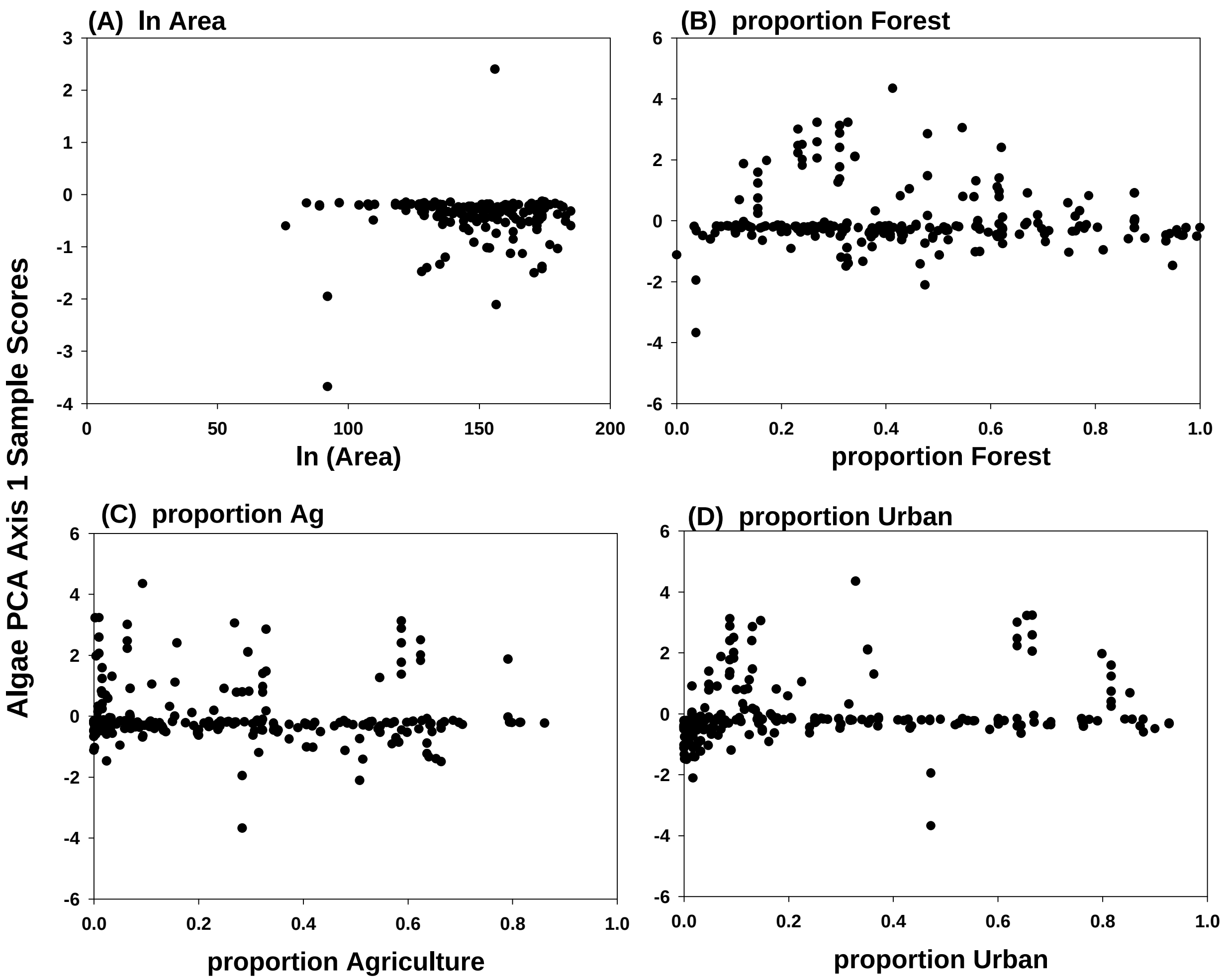


Figure 3.4.2. Relationship between APCA1 scores and (A) ln Area, (B) proportion Forest, (C) proportion Ag, and (D) proportion Urban from algal samples in the Integrated Analyses dataset (n=223).

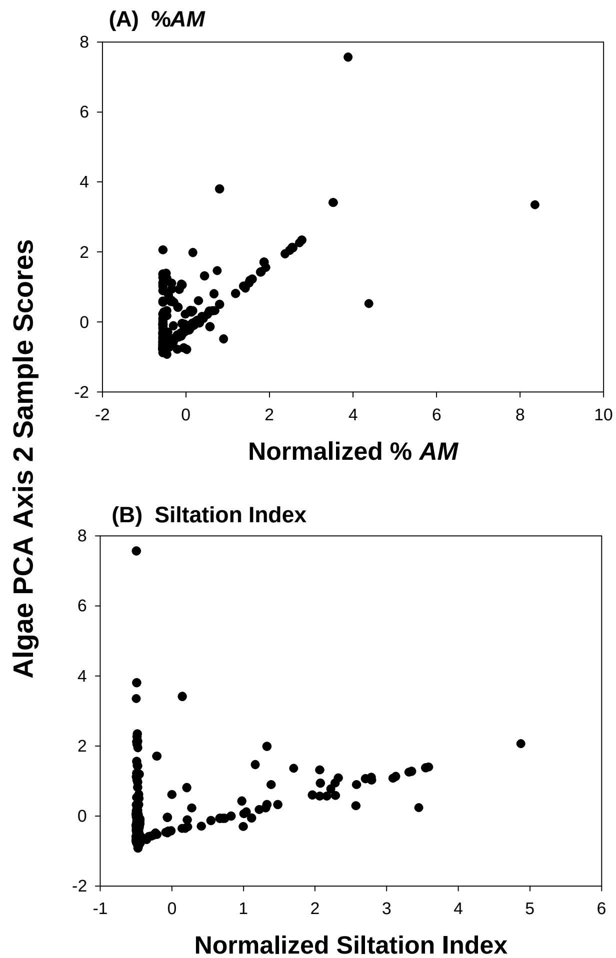


Figure 3.4.3. Relationship between APCA2 scores and normalized (A) %AM and (B) Siltation Index from algal samples in the Integrated Analyses dataset (n=242).

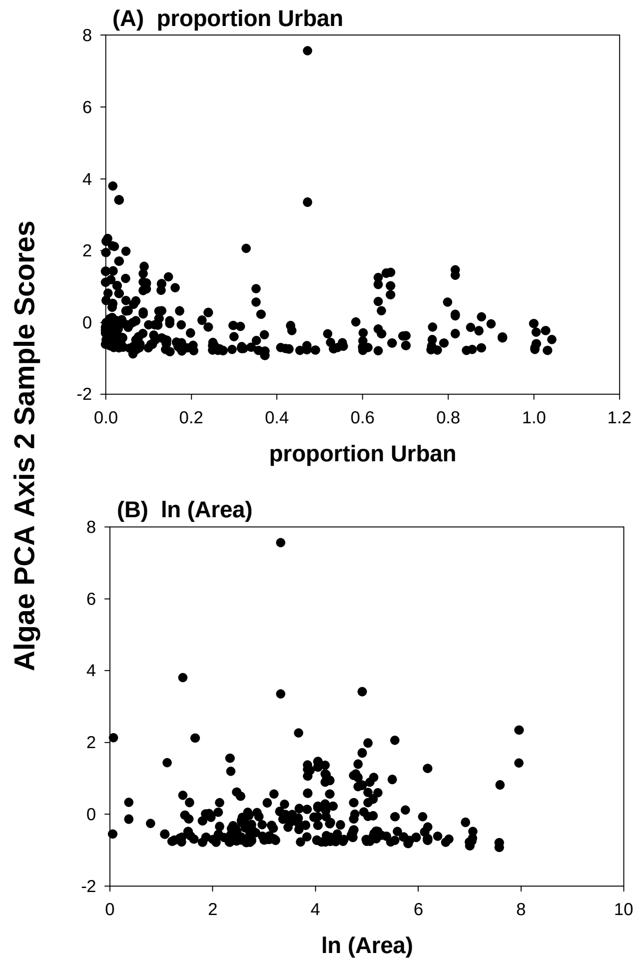


Figure 3.4.4. Relationship between APCA2 scores and (A) proportion Urban and (B) ln(Area) from algal samples in the Integrated Analyses dataset (n=223).

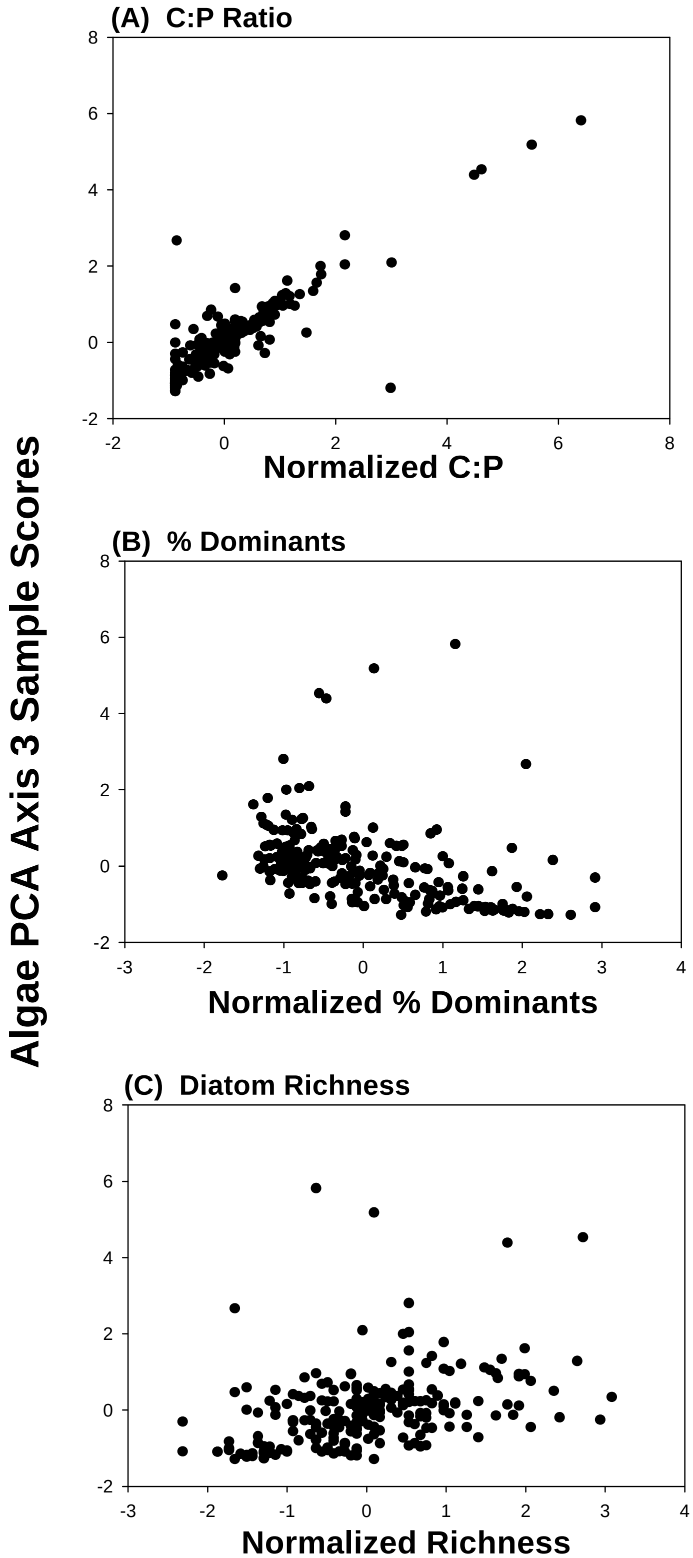


Figure 3.4.5. Relationship between APCA3 scores and normalized (A) C:P, (B) % Dominants, and (C) Diatom Richness from algal samples in the Integrated Analyses dataset (n=242).

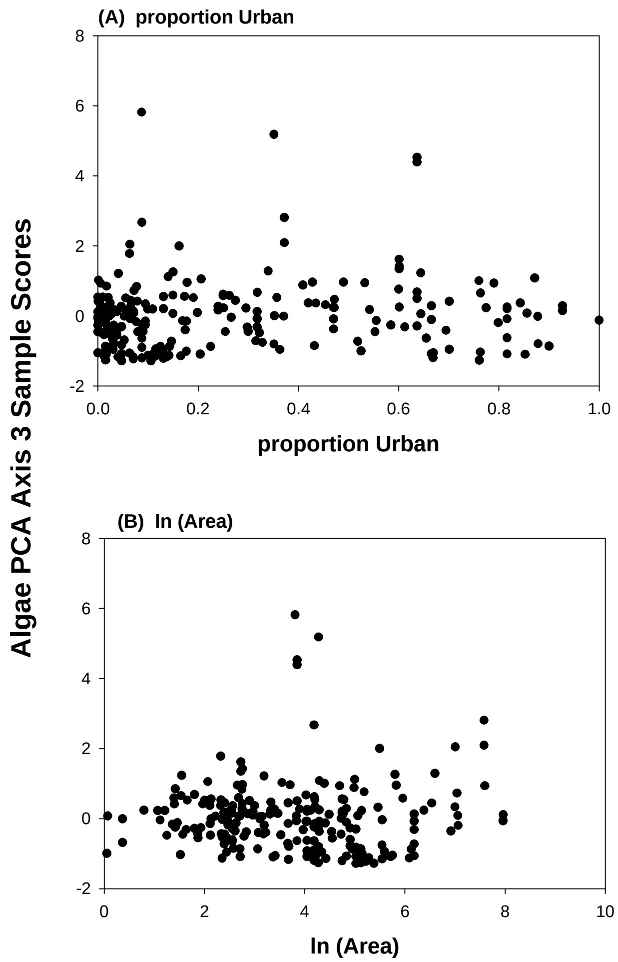


Figure 3.4.6. Relationship between APCA3 scores and (A) proportion Urban and (B) ln(Area) from algal samples in the Integrated Analyses dataset (n=223).

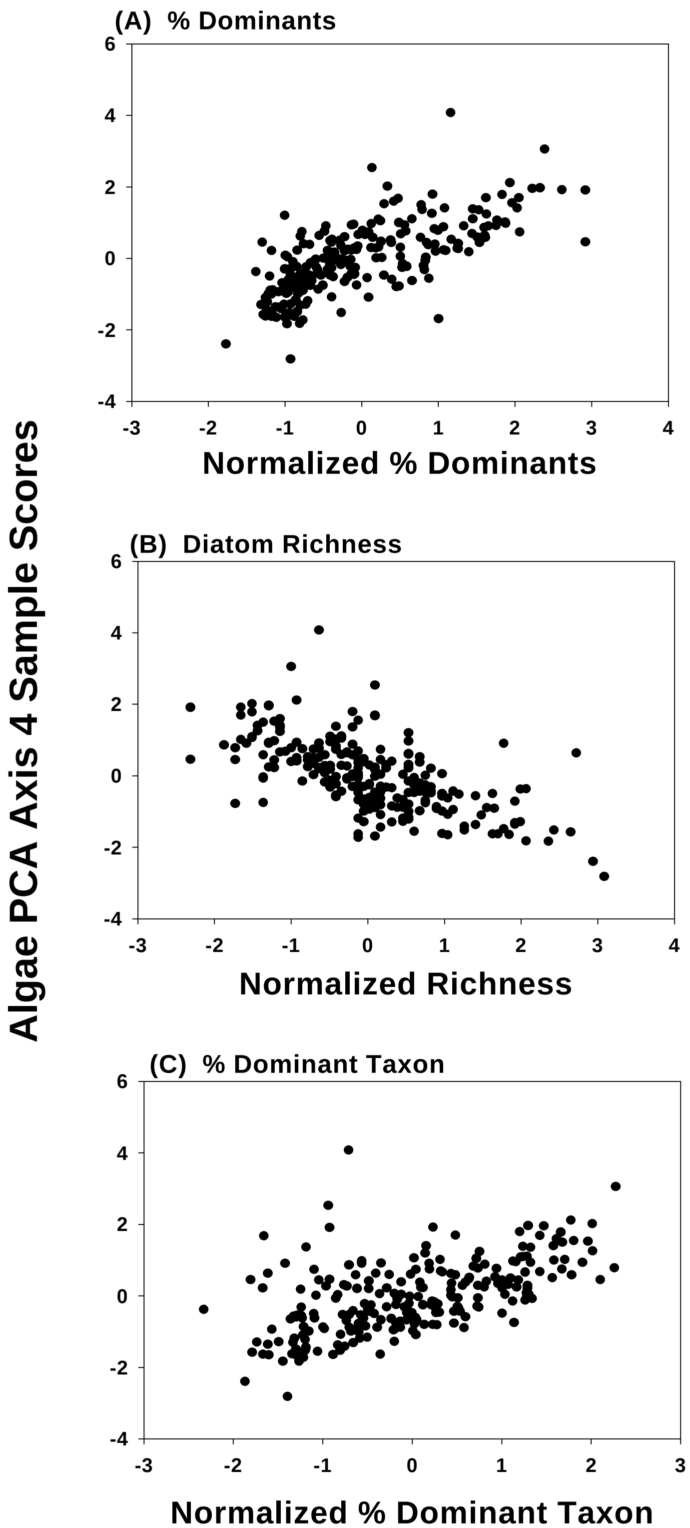


Figure 3.4.7. Relationship between APCA4 scores and normalized (A) % Dominants, (B) Diatom Richness, and (C) % Dominant Taxon from algal samples in the Integrated Analyses dataset (n=242).

3.4.3 Correlations among Algal and Macroinvertebrate Metrics

Algae and macroinvertebrate analyses showed relationships between metrics and land use indicating a response to different environmental variables. When algae and macroinvertebrate metrics respond to the same environmental condition, it is expected that these metrics be highly correlated. Multiple regression analyses of each algae PC and the first four macroinvertebrate PCs showed varying results. Regression analyses of macroinvertebrate PCs against APCA1 showed a significant negative relationship with MPCA4 ($p=0.021$) (Fig. 3.4.8). However, the overall regression model was not significant ($p=0.097$) and accounted for less than 4% of the variation in APCA1 scores ($=0.0342$). A similar relationship between APCA2 and macroinvertebrate metrics was seen with MPCA4 significantly negatively correlated with APCA2 ($p=0.016$) (Fig. 3.4.9) although the overall model was not significant ($p=0.12$) and explained very little ($r^2=0.0315$). APCA3 and APCA4 showed opposing relationships with MPCA2. APCA3 showed a significant negative ($p<0.001$) (Fig. 3.4.10) and APCA4 showed a significant positive ($p=0.014$) relationship with MPCA2 (Fig. 3.4.11). Neither model explained more than 7.6% of variation among algae sample scores, although the overall model was significant (APCA3: $p=0.001$) or almost so (APCA4: $p=0.061$).

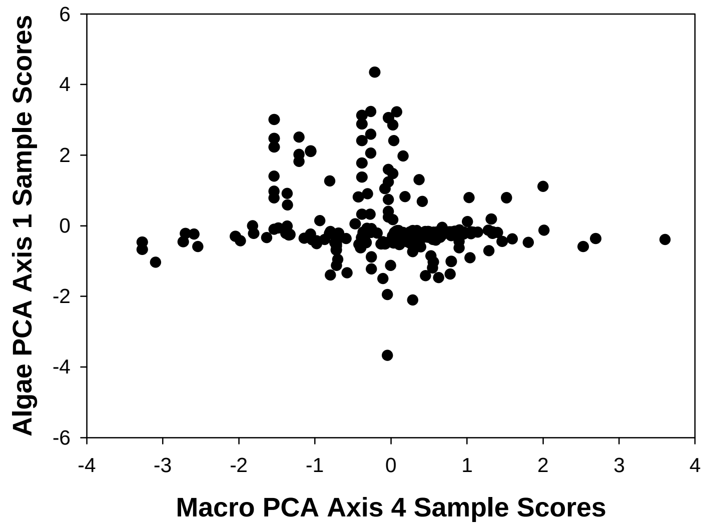


Figure 3.4.8. Relationship between APCA1 scores and MPCA4 scores (n=229).

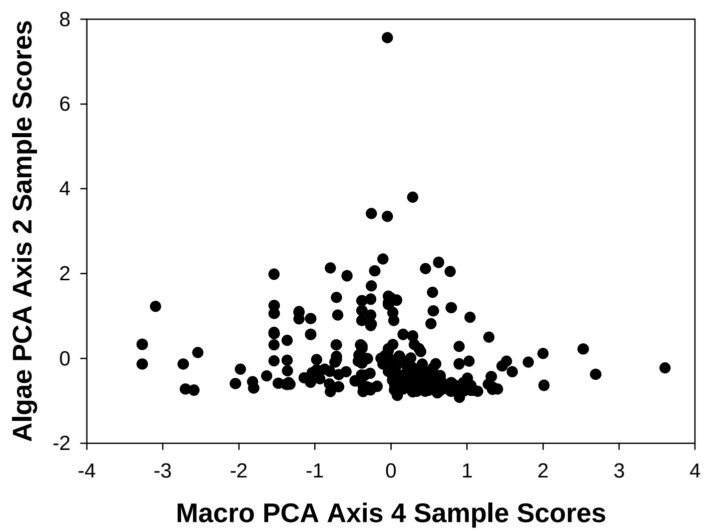


Figure 3.4.9. Relationship between APCA2 scores and MPCA4 scores (n=229).

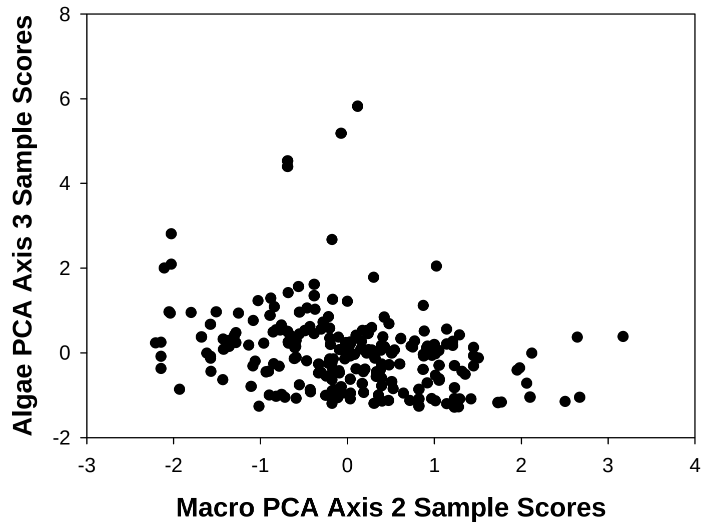


Figure 3.4.10. Relationship between APCA3 scores and MPCA2 scores (n=229).

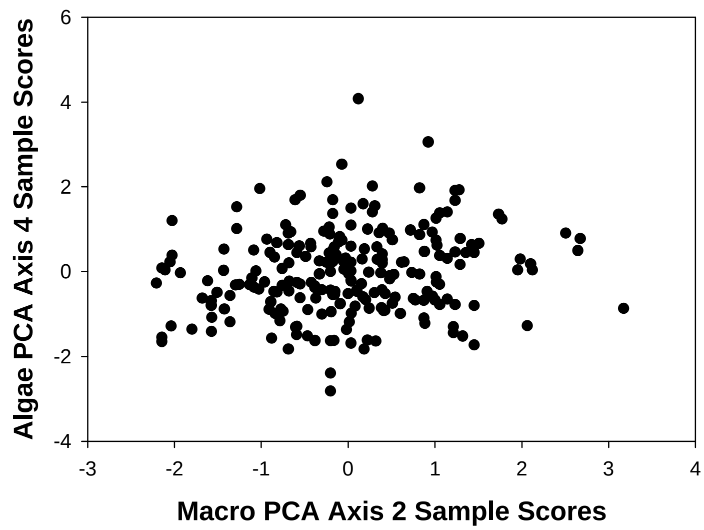


Figure 3.4.11. Relationship between APCA4 scores and MPCA2 scores (n=229).

3.5 Algae Sub-set Analyses

To determine if patterns differed for project-specific or regional samples, a subsample of the algae dataset was examined based on the project source (NJ Algal Indicators of Eutrophication) and sub-basin (Raritan). The following sections summarize habitat and land use relationships among algae sites within each subset, within-subset correlations among metrics, and correlations among algae and macroinvertebrate metric principal components within each subset.

3.5.1 NJ Algal Indicators of Eutrophication

Pearson correlation analyses (n=62) showed that the Shannon-Wiener Diversity index was highly positively correlated with Diatom Species Richness and highly negatively correlated with % Dominant Taxon and % Dominants metrics (Table 3.5.1). The % Dominants metric was positively correlated with % Dominant taxon. As a result, Shannon-Wiener Diversity Index and %Dominants were excluded from PCA analyses. Correlation analyses of land use, watershed are, and habitat scores (n=56) showed a strong negative relationship between sqrt (proportion forest) and sqrt (proportion urban) (Pearson correlation: r=-0.78), so sqrt (proportion forest) was excluded from further regression analyses. EPA Habitat scores were negatively correlated with sqrt (proportion urban) (Pearson correlation: r=-0.53) and positively correlated with sqrt (proportion forest) (Pearson correlation: r=0.62).

Correlation coefficients between NJ algae metrics and PCA axes are shown in Table 3.5.2. The first PC (NJ-APCA1) was highly positively correlated with % *Achnanthidium minutissimum* (Fig. 3.5.1). Samples with high NJ-APCA1 had a diatom community with a high percentage of *A. minutissimum*. This taxa requires hard substrates for colonization, and is typically abundant in headwater and low nutrient streams (M. Potapova, personal communication). NJ-APCA1 was also negatively associated with Siltation Index and Centrales:Pennales ratio, which typically increase with stream size and turbidity (Fig. 3.5.2). Because % *A. minutissimum* showed little relationship with stream size, NJ-APCA1 can likely be interpreted as substrate index. As with the overall dataset, Miry Run had high NJ-APCA1 scores and a high % of *A. minutissimum*. The second NJ PC was positively related to the ratio of Centrales:Pennales (Fig. 3.5.3). A high Centrales:Pennales ratio typically occurs in larger rivers or in streams receiving lake or pond inputs. In the NJ dataset, normalized C:P ratio values decreased with decreasing proportion Wet (Fig. 3.5.4), suggesting that this axis may be related to upstream inputs. NJ-APCA3 was strongly related to diversity metrics (Fig. 3.5.5). High NJ-APCA3 scores had low values of Diatom Species Richness and high values of % Dominant Taxon. There was a trend for increasing Diatom Richness and decreasing % Dominant Taxon with increasing stream size, suggesting NJ-APCA3 can be interpreted as a stream size gradient. The correlations of NJ-APCA4 are weak and contradictory with positive relationships with both Diatom Species Richness (PCA correlation: r=0.3748) and % Dominant Taxon (PCA correlation: r=0.4866) (Fig. 3.5.6).

Multiple regressions were performed to relate NJ Algae PCs to macroinvertebrate PCs. The regression models for the first two NJ algae principal components (NJ-APCA1 and NJ-APCA2) were not significant and accounted for very little variation among sample scores ($p > 0.05$, $r^2 < 0.088$). However, the regression of macroinvertebrate PCs on NJ-APCA3 was significant (p=0.018). The model explained 19.2% of the variation in NJ-APCA3 scores with MPCA4 being the only significant component (p=0.017) (Fig. 3.5.7). MCPA1 was a significant factor in the regression model against NJ-APCA4 (p=0.008) (Fig. 3.5.8) although the overall model was not significant with an r^2 value of 0.13.

Multiple regressions were also performed to determine the relationship between watershed factors (land use, area, EPA Habitat score) and NJ Algae PCs. Results from regression analysis of watershed factors NJ Algae PCs was similar for the first two PCs. The model for NJ-APCA1 was borderline significant (p=0.055) and accounted for 19.0% of the variation in NJ-APCA1 scores. Similar results were seen for NJ-APCA2 (p=0.060, r^2=0.186). For both PCs, ln (area) was the only significant model factor (NJ-APCA1: p=0.031, NJ-APCA2: p=0.003) (Fig. 3.5.9). The third and fourth NJ Algae PCs were negatively related to sqrt (prop Wet) (p=0.060) and positively related to sqrt (prop Wet) (p=0.0015), respectively (Fig. 3.5.10). Neither model was significant with the model accounting for 15.2% in NJ-APCA3 and 12.6% in NJ-APCA4.

Table 3.5.1. Pearson correlation coefficients of normalized algae metrics from samples collected for the NJ Algal Indicators of Eutrophication project (n=62). Shaded areas represent correlation coefficients $0.7 or #-0.7.

	Diatom Richness	Shannon-Wiener Index	% Dominants	% Dominant Taxon	Centralles: Pennales	% AM	Siltation Index
Diatom Richness	1						
Shannon-Wiener Index	0.887	1					
% Dominants	-0.776	-0.906	1				
% Dominant Taxon	-0.597	-0.85	0.716	1			
Centralles:Pennales	0.24	0.163	-0.047	-0.122	1		
% AM	-0.333	-0.457	0.328	0.484	-0.321	1	
Siltation Index	0.281	0.363	-0.25	-0.326	0.252	-0.488	1

Table 3.5.2. Algae metric correlations with PCA axes for samples collected for the NJ Algal Indicators of Eutrophication project (n=62).

	NJ-APCA1	NJ-APCA2	NJ-APCA3	NJ-APCA4
Diatom Species Richness	-0.3494	0.1489	-0.8457	0.3748
% Dominant Taxon	0.486	0.0575	0.7237	0.4866
Centrales:Pennales	-0.3592	0.9328	0.0232	-0.0158
% AM	0.999	0.0417	-0.0135	-0.0026
Siltation Index	-0.4939	0.0824	-0.116	-0.0051

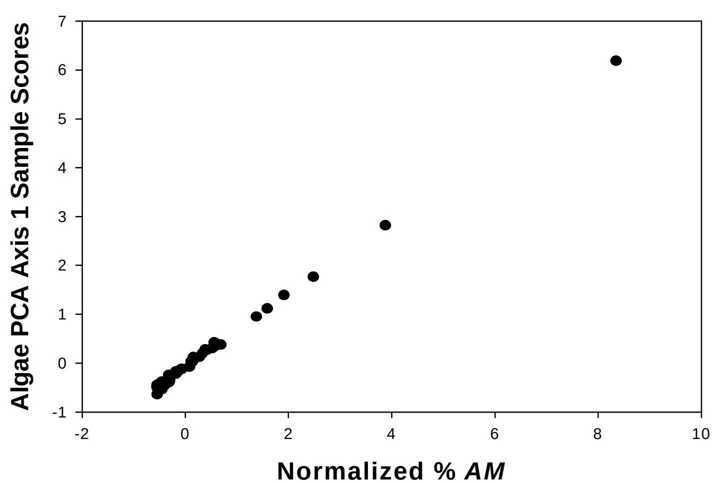


Figure 3.5.1. Relationship between NJ-APCA1 scores and normalized % AM from the NJ Algal Indicators of Eutrophication dataset (n=60).

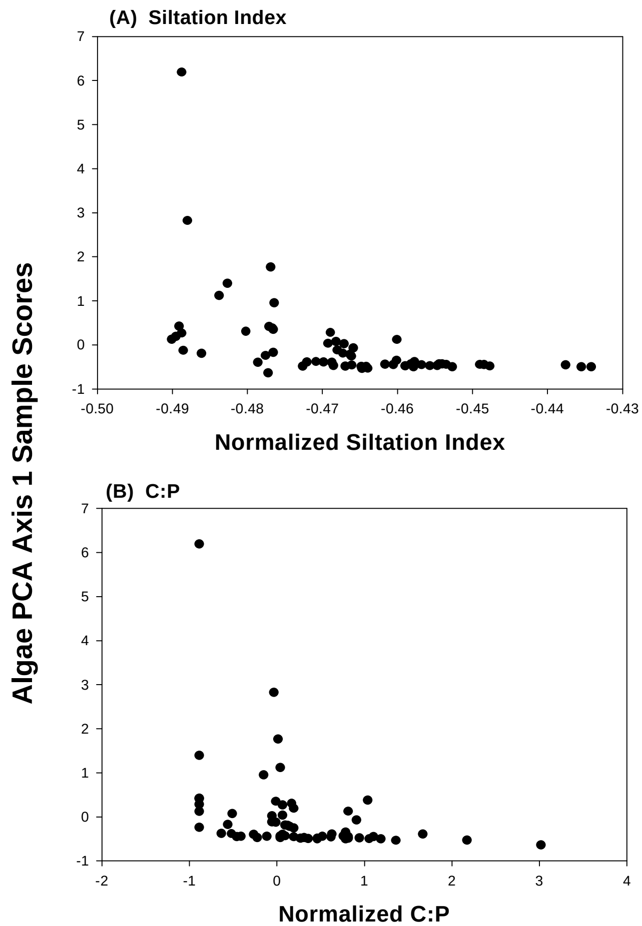


Figure 3.5.2. Relationship between NJ-APCA1 scores and normalized (A) Siltation Index and (B) C:P from the NJ Algal Indicators of Eutrophication dataset (n=60).

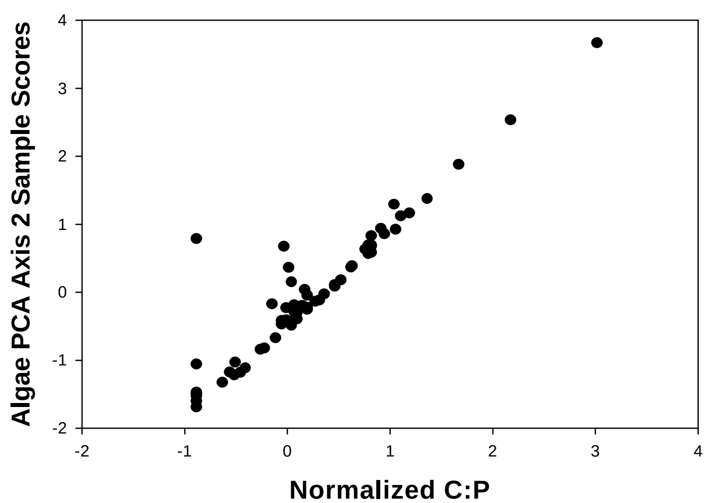


Figure 3.5.3. Relationship between NJ-APCA2 scores and normalized C:P from the NJ Algal Indicators of Eutrophication dataset (n=60).

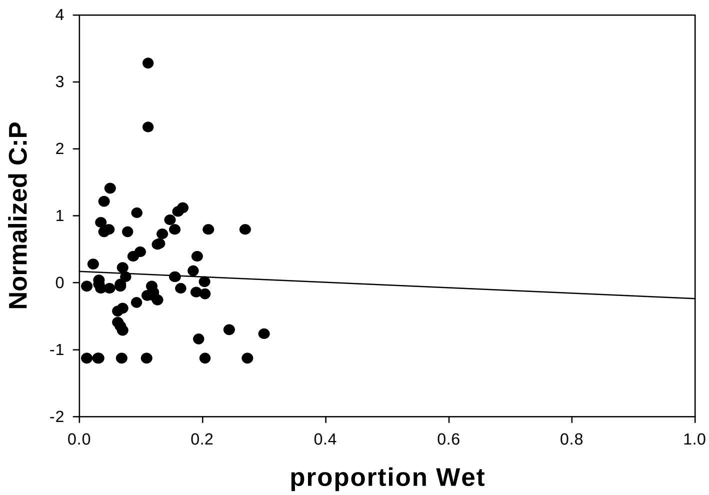


Figure 3.5.4. Relationship between normalized C:P metric and the proportion Wet from the NJ Algal Indicators of Eutrophication dataset (n=56).

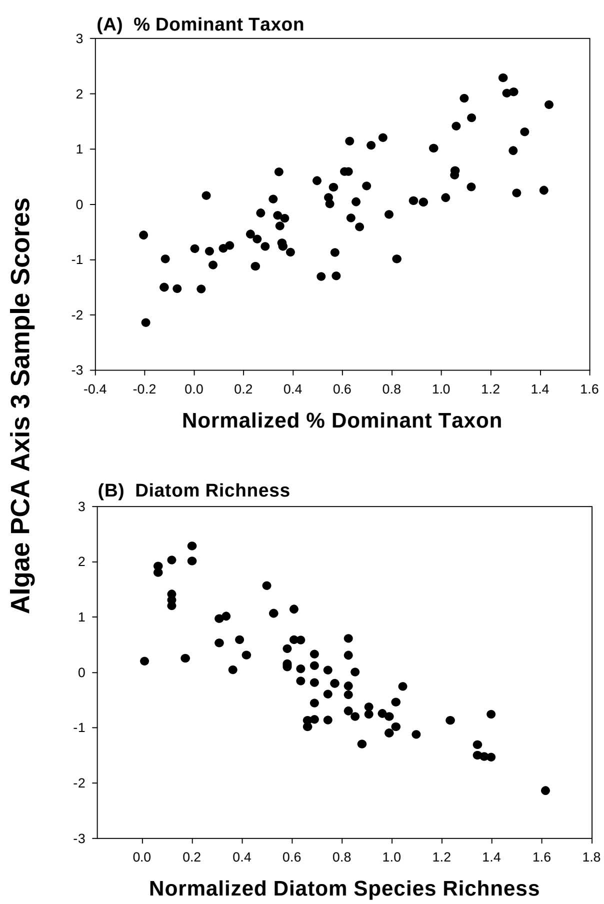


Figure 3.5.5. Relationship between NJ-APCA3 scores and normalized (A) Diatom Richness and (B) % Dominant Taxon from the NJ Algal Indicators of Eutrophication dataset (n=60).

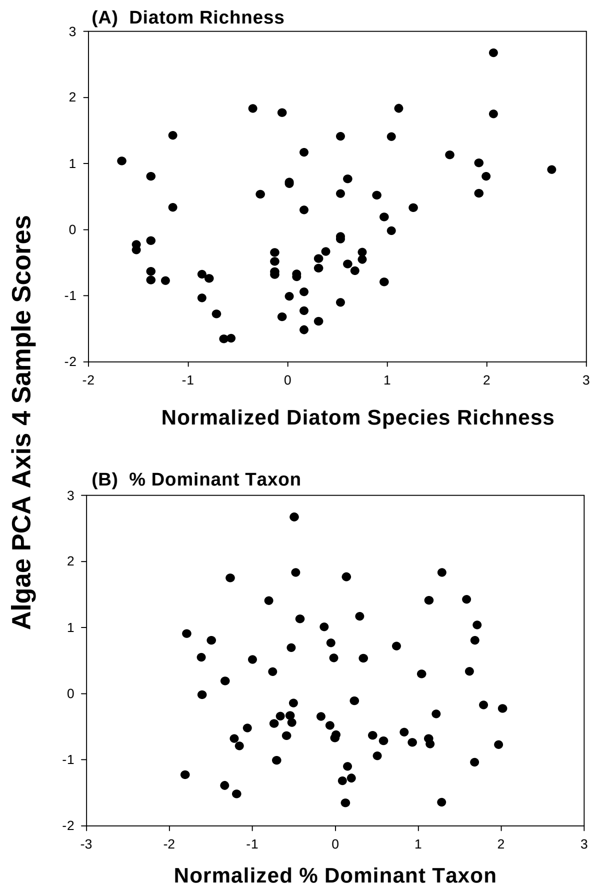


Figure 3.5.6. Relationship between NJ-APCA4 scores and normalized (A) Diatom Richness and (B) % Dominant Taxon from the NJ Algal Indicators of Eutrophication dataset (n=60).

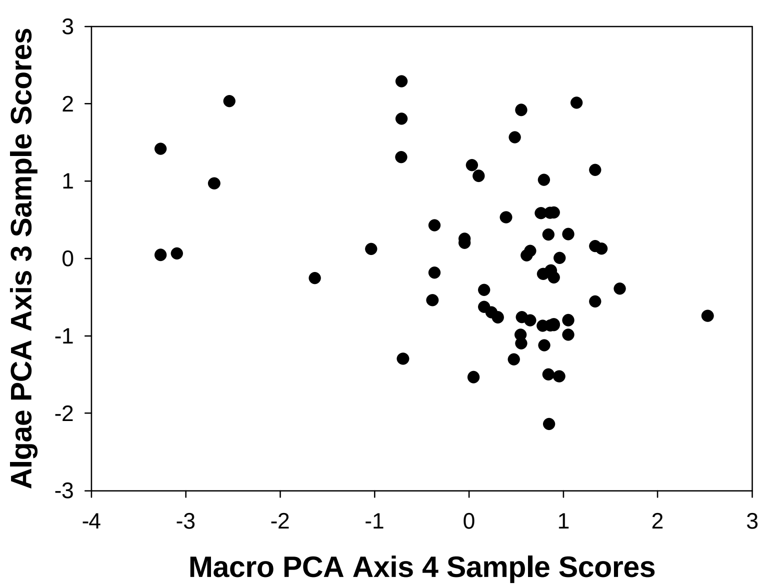


Figure 3.5.7. Relationship between NJ-APCA3 scores and MPCA4 scores (n=60).

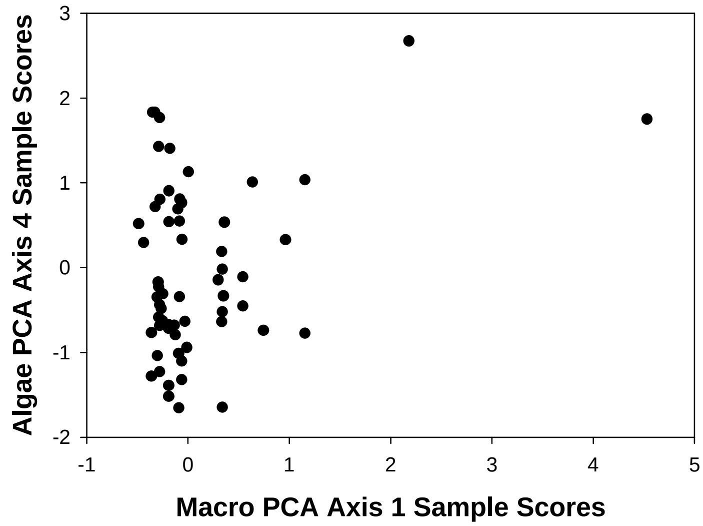


Figure 3.5.8. Relationship between NJ-APCA4 scores and MPCA1 scores (n=60).

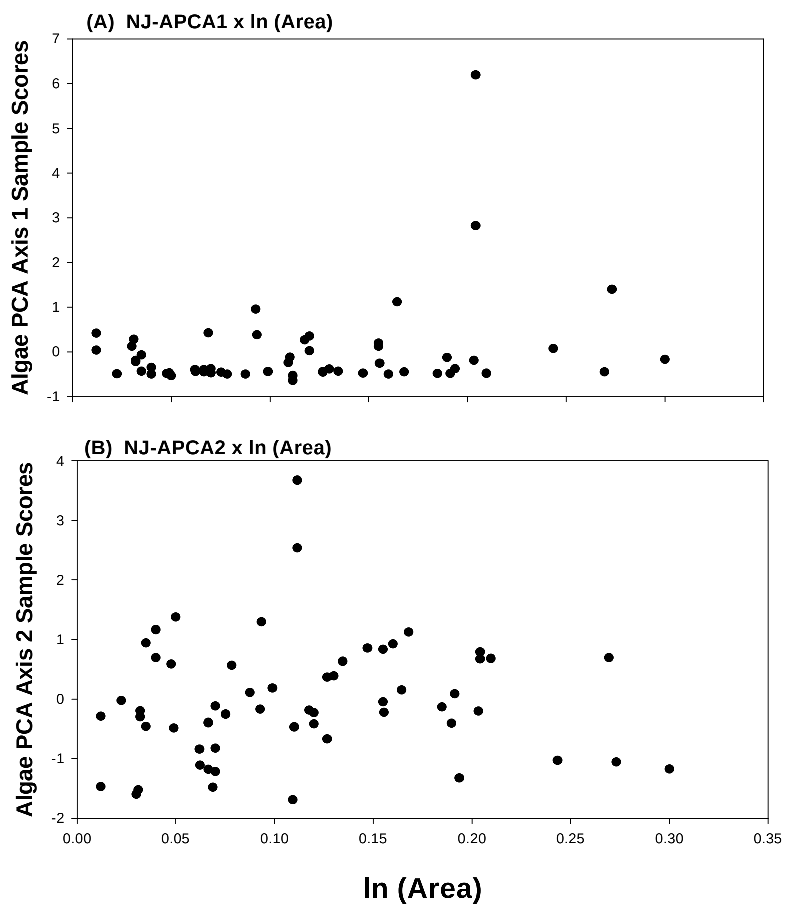


Figure 3.5.9. Relationship between ln (Area) and (A) NJ-APCA1 and (B) NJ-APCA2 scores (n=56).

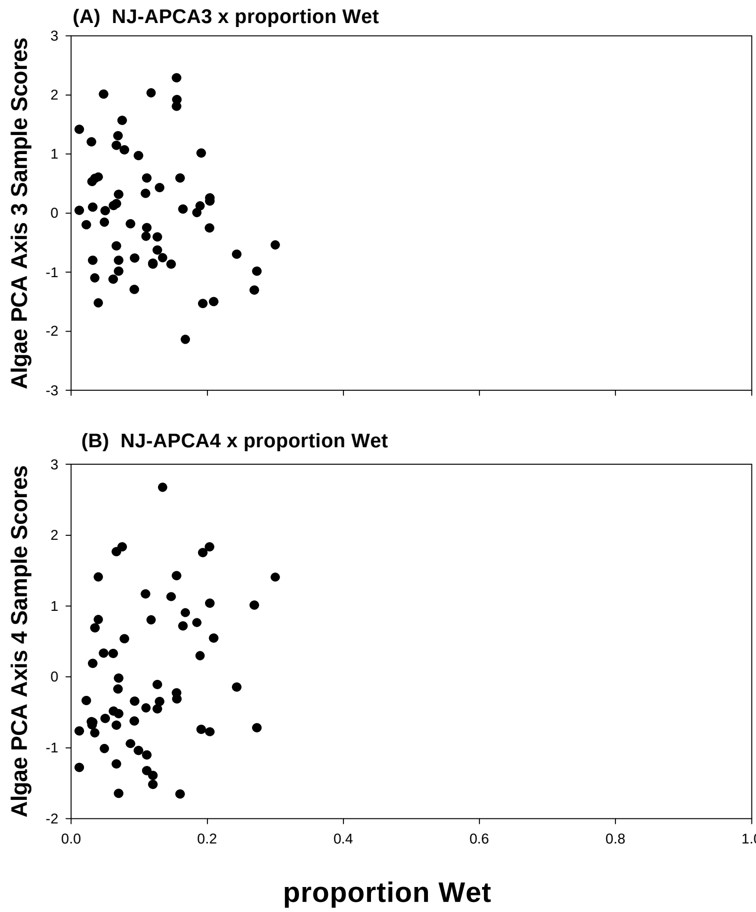


Figure 3.5.10. Relationship between proportion Wet and (A) NJ-APCA3 and (B) NJ-APCA4 scores (n=56).

3.5.2 Raritan River Sub-basin

Results of Pearson correlation analyses (n=43) for algae metrics in samples collected from the Raritan River sub-basin showed that the Shannon-Wiener Diversity index was highly positively correlated with Diatom Species Richness and highly negatively correlated with % Dominant Taxon and % Dominants metrics (Table 3.5.3). Therefore, the Shannon-Wiener Diversity Index was excluded from PCA analyses. Correlation analyses of land use, watershed area, and habitat scores (n=25) showed sqrt (proportion Urban) was negatively related to sqrt (proportion Forest) and sqrt (proportion Ag) (Table 3.5.4A) and so it was excluded from regression analyses using EPA Habitat score in the model. Inclusion of EPA Habitat scores in the correlation reduced the sample size by over 40% so a second correlation was conducted excluding this variable. In this correlation (n=43), sqrt (proportion Ag) was highly negatively correlated with sqrt (proportion Urban) (Table 3.5.4B). As a result, all regression analyses excluding EPA Habitat scores also excluded sqrt (proportion Ag).

Correlation coefficients for Raritan River sub-basin axes are shown in Table 3.5.5. There was a strong positive correlation between the first Raritan River sub-basin PC (RAR-APCA1) and Siltation Index (Fig. 3.5.11). There is a positive relationship between watershed area and Siltation Index, although Siltation Index decreases with increasing urbanization in the watershed (Fig. 3.5.12). This suggests that this axis is primarily a stream size gradient. The second PC(RAR-APCA2) is strongly positively correlated with the ratio of Centrales:Pennales, and less so with Diatom Richness (Fig. 3.5.13). This ratio is typically higher in large rivers with well developed plankton communities, in streams that drain lakes, ponds, or reservoirs, or in small rivers with increased nutrient loading. Because this metric is only weakly positively correlated with drainage area (Fig. 3.5.14), this axis is likely related to stream inputs or nutrient concentrations. Samples from the Neshaic River showed unusual and opposing patterns for C:P and RAR-APCA2. One sample had high RAR-PCA2 scores (3.6113) but a mow normalized C:P value, whereas another sample showed the opposite pattern with a C:P value of 2.99 and RAR-APCA2 score of -0.3774. The third Raritan River sub-basin axis (RAR-APCA3) related to diversity. Samples with high RAR-APCA3 had low Diatom Richness and high % Dominant Taxon and %Dominants values (Fig. 3.5.15). The last axis (RAR-APCA4) was positively correlated with % *Achnanthidium minutissimum* (Fig. 3.5.16). This axis may be related to stream size and water quality as % *A. minutissimum* scores tended to decrease with increasing watershed area (Fig. 3.5.17).

Multiple regressions were used to examine the relationship between Raritan River basin PCs and macroinvertebrate PCs. The model for RAR-APCA1 was significant p=0.039) and accounted for 22.7% of the variation among RAR-APCA1 scores. There was a significant trend for increasing MPCA3 scores with increasing RAR-APCA1 scores (p=0.006) (Fig. 3.5.18). The regression model for RAR-APCA2 was not significant (p=0.47) and accounted for the lowest percentage of variation in Raritan River basin PCs (r2=0.086). Although the pattern was not significant, RAR-APCA2 tended to decrease with increasing MPCA2 scores (p=0.072) (Fig. 3.5.19). There was a significant decrease in RAR-APCA3 score with increasing MPCA1 scores (p=0.047) although the over all model was not significant and explained only 15.9% of RAR-APCA3 variation (Fig. 3.5.20). However, the outcome of this regression model was dominated by high MPCA1 scores (4.5312) from a site on Millstone River (AMNET AN0382). This site is dominated by Chironomids (47.5%) and non-insects (50%). When this site was removed from the regression model, there was no relationship between RAR-PCA3 and any macroivertebrate PCs ($p=0.41$, $r^2=0.999$). Macroinvertebrate PCs did not account for a significant amount of variation in RAR-APCA4 ($p=0.27$, $r^2=0.159$), although RAR-APCA4 tended to decrease with increasing MPCA4 ($p=0.086$) (Fig. 3.5.21).

Because sample size was affected by the presence of EPA Habitat score in Raritan River basin PCs (n=25 vs. n=43), two regression analyses were conducted for each PC. One model (all variables, AV) included all land use variables, EPA Habitat score, and drainage area. The other model (variable subset, VS) included drainage area, sqrt (prop Urban), sqrt (prop Forest), and sqrt (prop Wet), but excluded Habitat score and sqrt (prop Ag) (due to covariance). Both models relating RAR-APCA1 to environmental variables were significant (VS: $p=0.004$, AV: $p=0.038$) with RAR-APCA1 tending to increase with increasing area (Fig. 3.5.22). However, the AV model accounted for a greater amount of variation ($r^2=0.490$) in RAR-APCA1 than the VS model ($r^2=0.324$). RAR-APCA2 environmental VS and AV models were not significant and explained 14.6% and 23.1% of variation among samples, respectively. In the VS model RAR-APCA2 decreased significantly with decreasing sqrt (prop Forest) ($p=0.027$) (Fig. 3.5.23). Despite differences in sample size, variation in RAR-APCA3 explained by the two models did not differ substantially (VS: $r^2=0.179$, AV: $r^2=0.174$). In the analyses of RAR-APCA4 relationships with environmental variables, both the VS ($p=0.005$) and AV ($p=0.009$) models were significant. However, in addition to the significant relationship of RAR-APCA4 with watershed area in both models ($p<0.05$), there was a strong negative relationship with sqrt (prop Ag) in the AV model ($p=0.079$) (Fig. 3.5.24) which resulted in a higher r^2 value (0.576) than in the VS model ($r^2=0.320$).

Table 3.5.3. Pearson correlation coefficients of normalized algae metrics for samples collected from the Raritan River sub-basin (n=43). Shaded areas represent correlation coefficients $0.7 or #-0.7.

	Diatom Richness	Shannon-Wiener Index	% Dominants	% Dominant Taxon	Centralles: Pennales	% AM	Siltation Index
Diatom Richness	1						
Shannon-Wiener Index	0.87179	1					
% Dominants	-0.54481	-0.74938	1				
% Dominant Taxon	-0.66634	-0.78171	0.26138	1			
Centralles:Pennales	0.20565	0.19102	0.1909	-0.42089	1		
% AM	0.0034	0.0264	0.16085	-0.33442	0.01702	1	
Siltation Index	-0.21648	-0.06399	-0.40165	0.48463	-0.18592	-0.35524	1

Table 3.5.4. Pearson correlation coefficients of environmental variables from samples collected from the Raritan River sub-basin. Shaded areas represent correlation coefficients $0.7 or #-0.7.

A. Including EPA Habitat Score (n=25)

	EPA Habitat	sqrt (prop Urban)	sqrt (prop Forest)	sqrt (prop Ag)	sqrt (prop Wet)	ln (area)
EPA Habitat	1					
sqrt (prop Urban)	-0.38876	1				
sqrt (prop Forest)	0.49893	-0.69587	1			
sqrt (prop Ag)	-0.07183	-0.68751	0.08708	1		
sqrt (prop Wet)	0.21205	-0.00095	-0.23362	-0.26286	1	
ln (area)	0.02295	-0.06756	-0.00211	0.18945	-0.11863	1

B. Excluding EPA Habitat Score (n=43)

	sqrt (prop Urban)	sqrt (prop Forest)	sqrt (prop Ag)	sqrt (prop Wet)	ln (area)
sqrt (prop Urban)	1				
sqrt (prop Forest)	-0.67054	1			
sqrt (prop Ag)	-0.73561	0.06512	1		
sqrt (prop Wet)	0.27693	-0.34689	-0.39937	1	
ln (area)	-0.0582	0.11167	0.05925	-0.16067	1

Table 3.5.5. Algae metric correlations with PCA axes for samples collected from the Raritan River sub-basin (n=43).

	Rar-APCA1	Rar-APCA2	Rar-APCA3	Rar-APCA4
Diatom Species Richness	-0.1934	0.4166	-0.6551	-0.3039
% Dominant Taxon	-0.3639	-0.2166	0.7164	0.2992
Percent Dominants	0.3796	-0.5211	0.6931	-0.0166
Centrales:Pennales	-0.1484	0.9549	0.2524	0.0264
% AM	-0.3456	0.0142	-0.276	0.887
Siltation Index	0.9996	0.0219	-0.006	0.0136

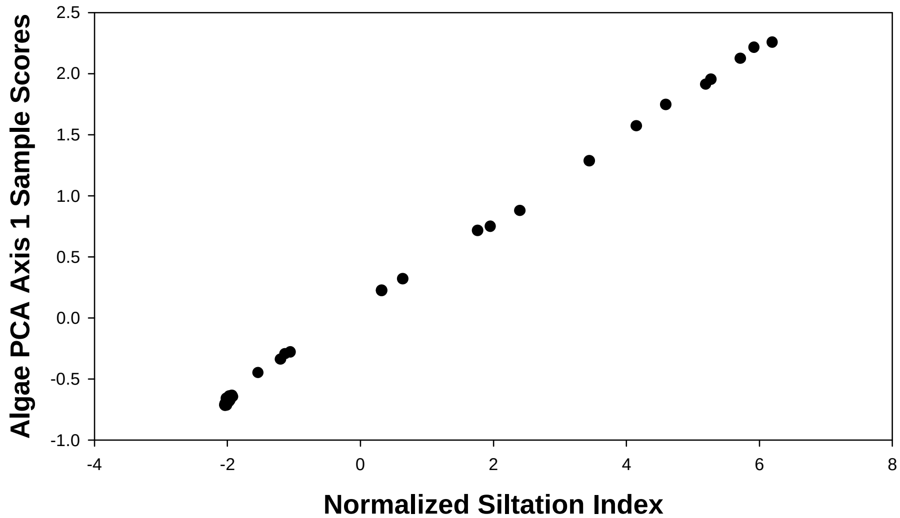


Figure 3.5.11. Relationship between RAR-APCA1 scores and normalized Siltation Index values from the Raritan River sub-basin dataset (n=43).

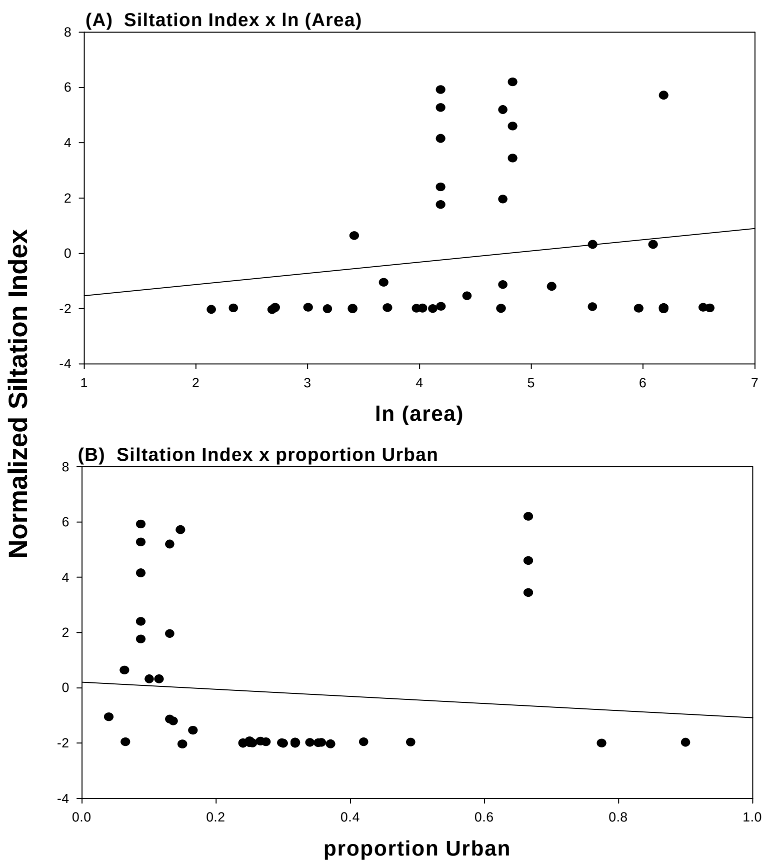


Figure 3.5.12. Relationship between normalized Siltation Index values and (A) ln (area) and proportion Urban values from the Raritan River sub-basin dataset (n=43).

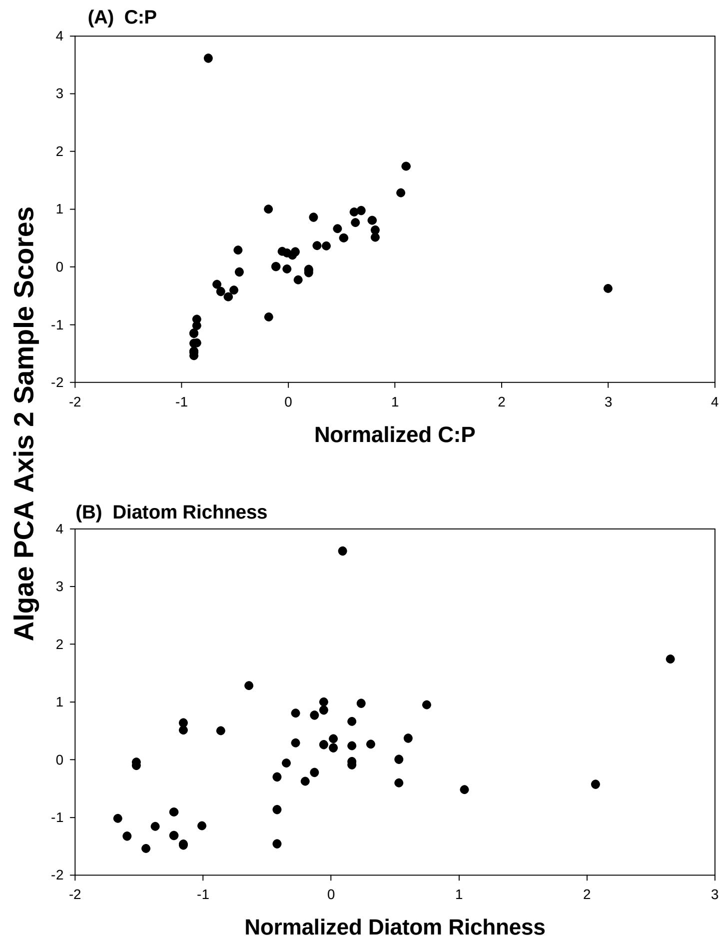


Figure 3.5.13. Relationship between RAR-APCA3 scores and normalized (A) C:P and (B) Diatom Richness values from the Raritan River sub-basin dataset (n=43).

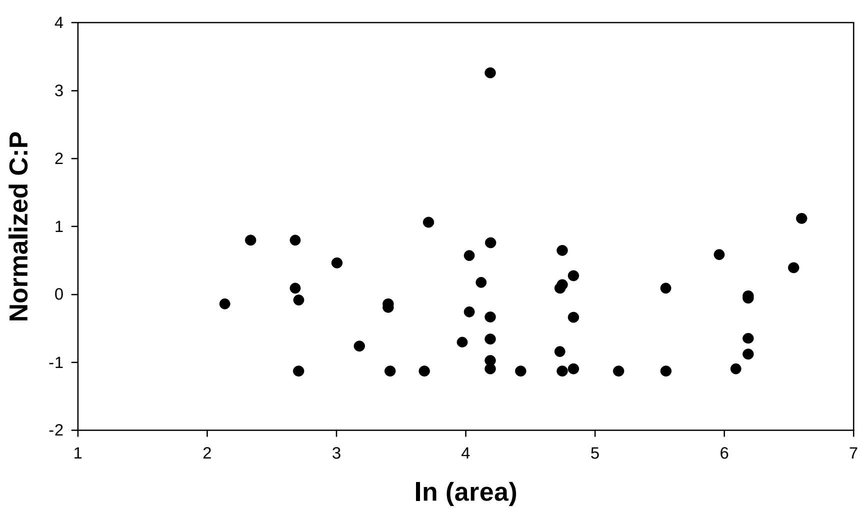


Figure 3.5.14. Relationship between normalized C:P values and ln (area) from the Raritan River sub-basin dataset.(n=43).

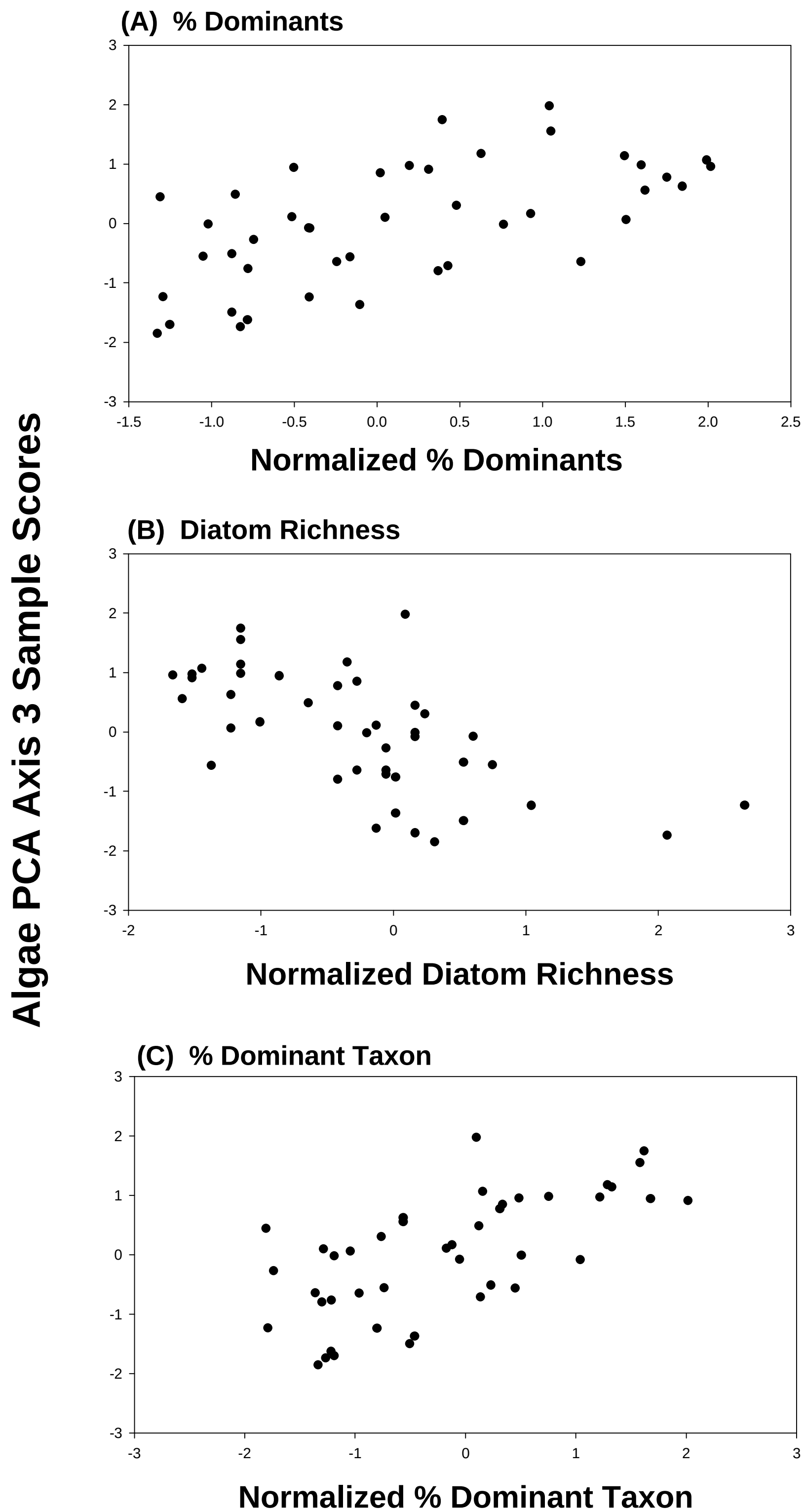


Figure 3.5.15. Relationship between RAR-APCA3 scores and normalized (A) % Dominants, (B) Diatom Richness, and (C) % Dominant Taxon values from the Raritan River sub-basin dataset (n=43).

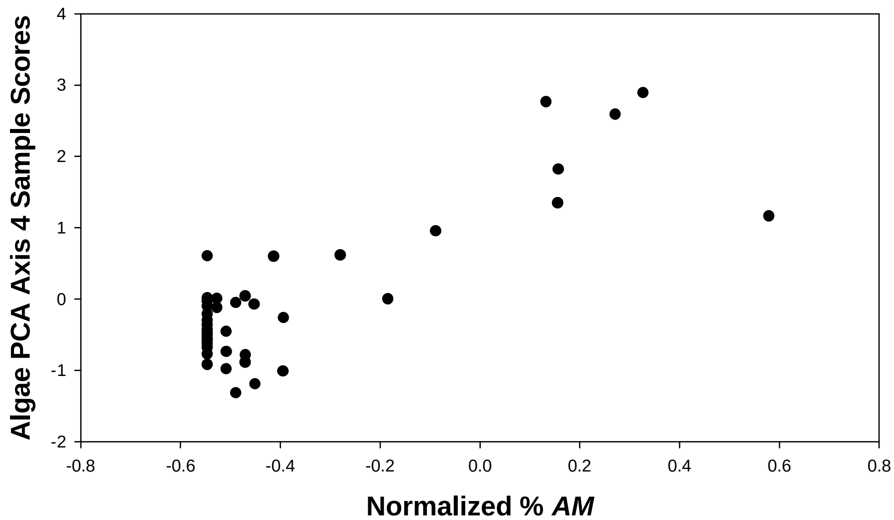


Figure 3.5.16. Relationship between RAR-APCA4 scores and normalized %AM values from the Raritan River sub-basin dataset (n=43).

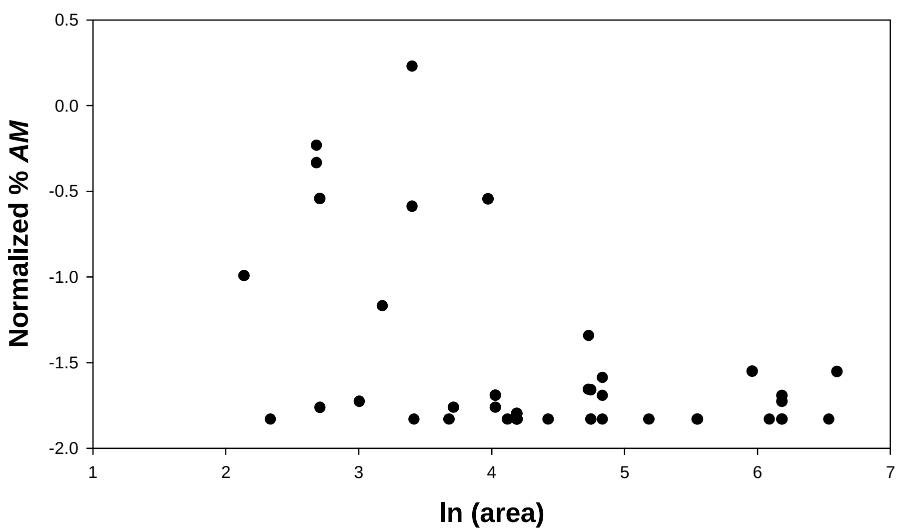


Figure 3.5.17. Relationship between normalized %AM values and ln (area) from the Raritan River sub-basin dataset.(n=43).

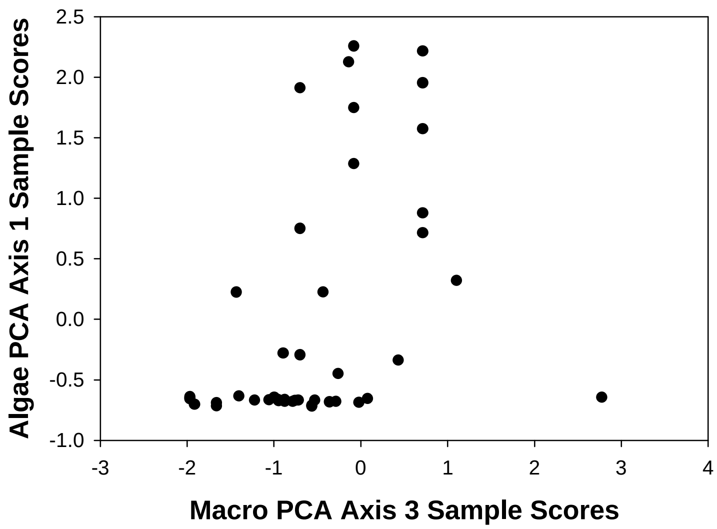


Figure 3.5.18. Relationship between RAR-APCA1 scores and MPCA3 scores (n=43).

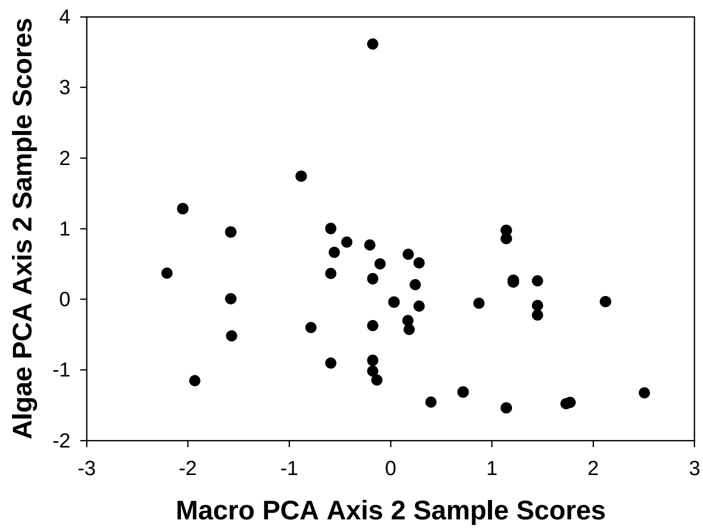


Figure 3.5.19. Relationship between RAR-APCA2 scores and MPCA2 scores (n=43).

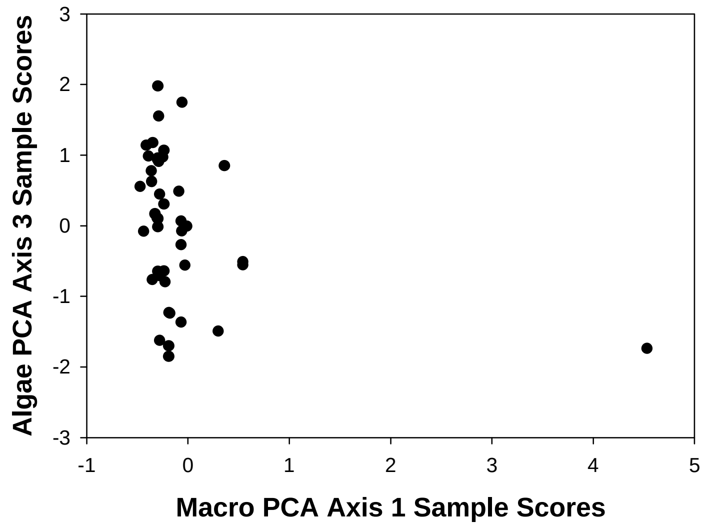


Figure 3.5.20. Relationship between RAR-APCA3 scores and MPCA1 scores (n=43).

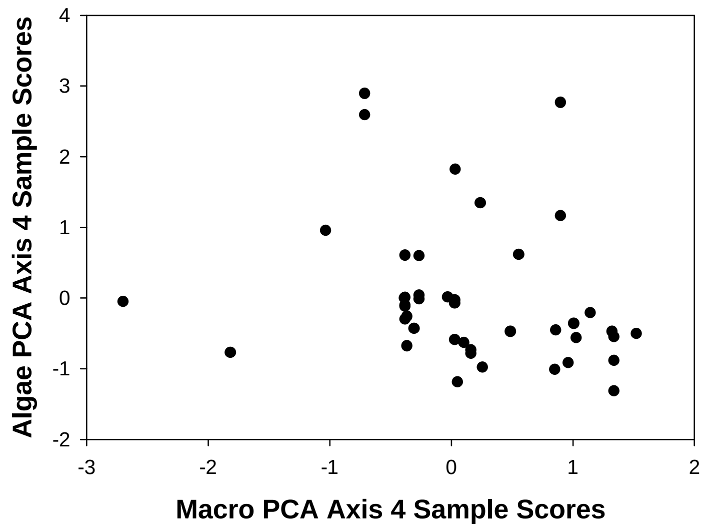


Figure 3.5.21. Relationship between RAR-APCA4 scores and MPCA4 scores (n=43).

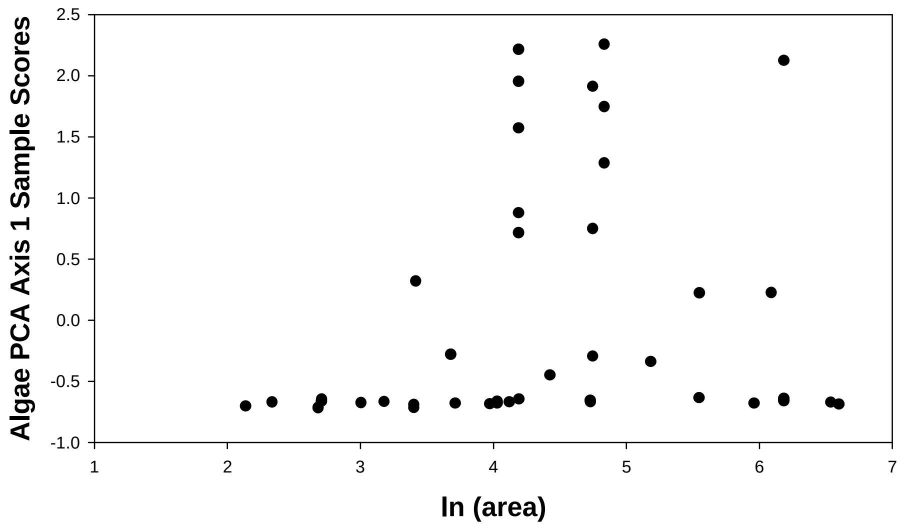


Figure 3.5.22. Relationship between RAR-APCA1 scores and ln (Area) values (n=43).

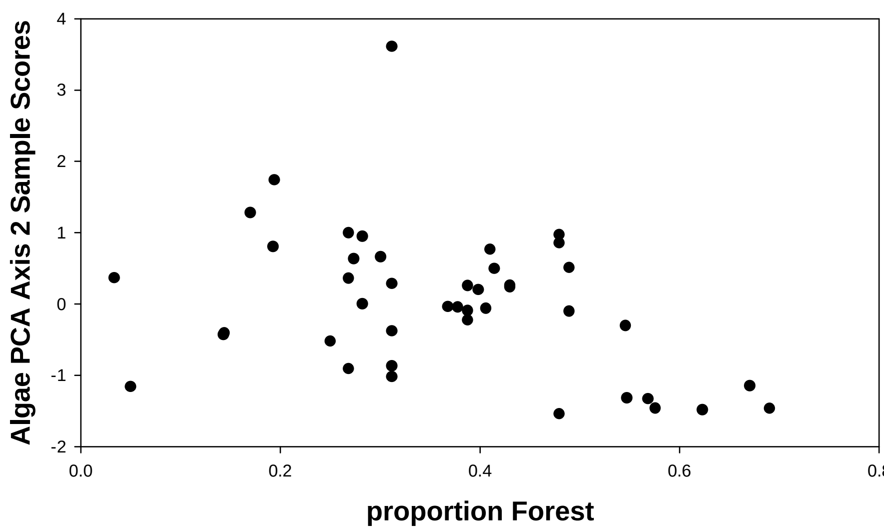


Figure 3.5.23. Relationship between RAR-APCA2 scores and proportion Forest values (n=43).

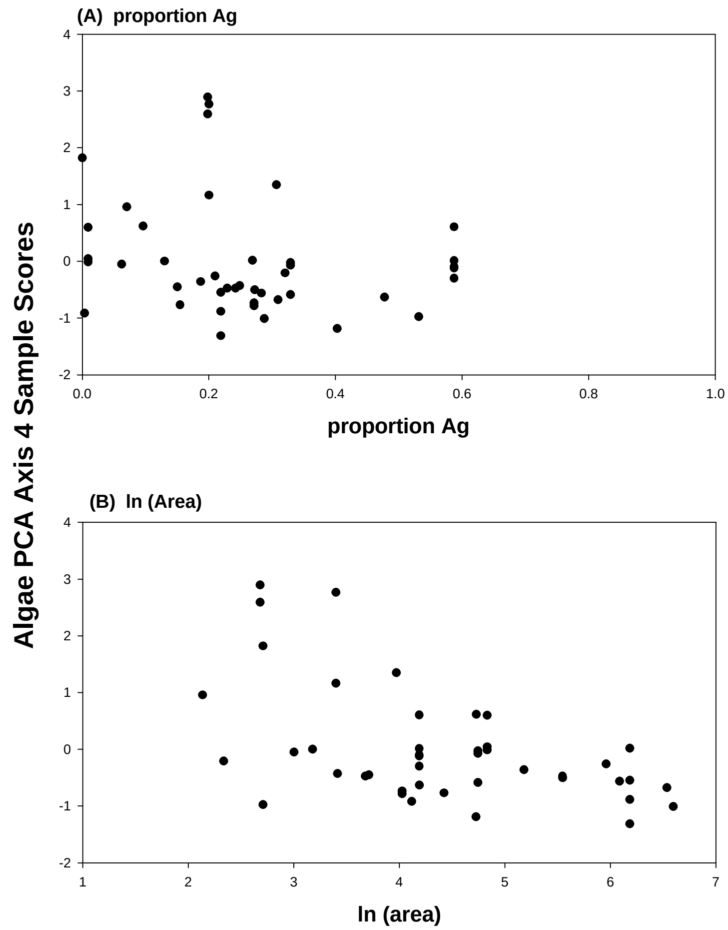


Figure 3.5.24. Relationship between RAR-APCA4 scores and (A) ln (area) and (B) proportion Ag values (n=43).

3.6 Mussels and Relationships Between Mussel Assemblages and Macroinvertebrate Metrics

Live individuals of eight taxa (Tables 3.6.1 and 3.6.2) of mussels were found, including tentative records of the paper pondshell (*Anodonta imbecillis*). Shells of two other species (the Eastern pondshell *Lampsilis radiata* and the alewife floater *Anodonta implicata*) were found. At a few stations, the triangle floater (*Alasmidonta undulata*) was represented only by shells. Including these shells as representing existing or recent occurrence, mussel distribution could be divided into several groups:

1) Specimens of *Alasmidonta*, alewife floater, and/or creeper (*Strophitis undulatus*) recorded. These are species which are often considered sensitive to impairment and which have decreased in many systems. The dwarf wedge mussel, alewife floater and Eastern pondshell were each found at only one station, often with other species (usually Eastern elliptio and/or Eastern floater *Pyganodon cataracta*). The triangle floater was found at 10 stations, and the creeper was found at 3 stations (with the triangle floater at 2 of these).

2) Specimens of Eastern lampmussel was found at one station, with the Eastern elliptio.

3) The Eastern pondmussel (*Ligumia nasuta*) was found at three stations; the Eastern elliptio and/or Eastern floater were found at all three of these. The Eastern pondmussel has also decreased in some parts of its range (e.g., Nedeau 2005).

4) Specimens of the Eastern floater were found at four stations, with only the Eastern elliptio (three stations) or with no other species (one station). The Eastern floater was also found at four stations with the Eastern pondmussel, triangle floater and/or creeper (these stations are treated in group 1).

5) Only Eastern elliptio (*Elliptio complanata*) found. This is a widespread species tolerant of a variety of stream conditions. It was the only species found at nine stations.

6) No mussels were caught at 12 stations.

The distribution of mussels is correlated with river size, land use and physiographic region (Table 3.6.3). Group 1 stations were mainly in forested watersheds with low-moderate amounts of urban and agricultural land. Almost all were in relatively large rivers, although creeper and triangle floater were found in some small streams. Almost all are in northern New Jersey. Several (e.g., Pequest River, Lamington River, Musconetcong River and Paulins Kill) have significant areas of carbonate geology in their watersheds (e.g., Cambrian and Ordivician dolomites and limestones, Drake et al. 1996). Land use data were not available for the single station at which the Eastern pondshell was caught (Walkill River). Groups 3, 4 and 5 (with Eastern pondmussel, Eastern floater, and/or Eastern elliptio) included smaller streams, streams with greater amounts of agriculture or urbanization, and several streams in the Inner Coastal Plain. Streams with no mussels tended to be very small (e.g., tributary Primrose Brook, Squankum Brook, Lopatcong Creek), or relatively high proportions of agricultural or urban land in the watershed.

Mussel sites were linked with macroinvertebrate data from AMNET sites (Table 3.6.4). Assemblages were mapped onto the first three principal components (Fig 3.6.1). These do not demonstrate unambiguous relationships between the mussel assemblages and the macroinvertebrate characteristics as indicated by the principal components. However, sites with no mussels, *Ligumia nasuta.* or only *Pyganodon cataracta* occur in sites with higher Mpca1 values, i.e., in sites with more tolerant macroinvertebrates. *Alasmodonta undulata* occurs over a range of macroinvertebrate assemblages. Comparisons of assemblages with selected metrics indicates that the assemblages containing *Alasmodonta* species and *Strophitus undulatus* tended to have values of several metrics indicating less tolerant macroinvertebrate communities (Figs. 3.6.2-3.6.4). In contrast, other assemblages occurred in sites with a much greater range of conditions.

Table 3.6.1. Scientific, common and code names for mussels (mollusca, unionidae) collected in the NJGFW mussel survey.

Code	Scientific	Common name
Alhet	*Alasmidonta heterodon*	Dwarf wedge mussel
Alund	*Alasmidonta undulata*	Triangle floater
Alvar	*Alasmidonta varicosa*	Brook floater
Animb	*Anodonta imbecillis*	Paper pondshell
Elcom	*Elliptio complanata*	Eastern elliptio
Larad	*Lampsilis radiata*	Eastern lampmussel
Linas	*Ligumia nasuta*	Eastern pondmussel
Pycat	*Pyganodon cataracta*	Eastern floater
Pyspe	*Pyganodon* species	Floater species
Stund	*Strophitus undulatus*	Creeper
Animp	*Anodonta implicata*	Alewife floater

Table 3.6.2. Numbers of live mussels collected in NJGFW mussel survey. Unidentified mussels are not included. Totunc is the total number of all species except *Elliptio complanata* and *Pyganodon cataracta*. Addl shells are species found as shells, but not found live at the site. See Table 3.6.1 for keys to species names.

Stream	ANSID	AMNET Site	Group	Live unc	Addl shells	TOTUnc	Alhet	Alund	Other	Elcom	Pycat Pyg
Pequest River	ANS0490	AN0043A	Alhet	Alhet	alund	2	2	0		50	
Lamington River	ANS0890	AN0370	Anvar, Alund	Anvar, Alund		2		1	1 Alvar	400	
Paulins Kill	ANS0128	AN0032A	Alund, Stund	Alund, Stund		11		4	7 Stund	354	0
Paulins Kill	ANS0473	AN0025A	Alund, Stund	Alund, Stund	alimp	33		9	24 Stund	202	2
Lubber's Run	ANS0123	AN0065	Alund	Alund		26		26		570	7
Paulins Kill	ANS0128	AN0032AA	Alund	Alund		2		2		95	0
Musconetcong River	ANS0077	AN0072	Alund	Alund		1		1		76	
Miry Run	ANS0577	AN0115A	Alund	Alund		1		1		1579	
Raritan River - SB	ANS0836	AN0322	Alund	Alund	Pycat	1		1		5	0
Musconetcong River	ANS0535	AN0073A	Alund		alund	0		0		83	
South Branch Raritan River	ANS0091	AN0326	Alund		alund	0		0		29	
Musconetcong River	ANS1347	AN0064	Stund						1 Stund	127	
Pompeston Creek	ANS0667	AN0177A	Linas	Linas		8			8 Linas	90	25
Raccoon Creek - RM	ANS1250	AN0685A	Linas	Linas		1			1 Linas	188	3
Dukes Brook	ANS0896	AN0375	Linas	None	Linas	0				1	0
Wallkill River	ANS0813	AN0302	Larad		Larad	0				85	
Haynes Creek	ANS0650	AN0168	Pycat	Pycat		0				87	1
Raccoon Creek	ANS1244	AN0679	Pycat	Pycat		0				457	29
Scotland Run	ANS1296	AN0725	Pycat	Pycat		3			2 cf Animb	300	78
Millstone River	ANS0903	AN0382A	Pycat	None	Pycat	0					0
		Elliptio only									
Paulins Kill Tributary	ANS0460	AN0014	Elo	Elo		0				10	
Beaver Brook	ANS0120	AN0047	Elo	Elo		0				9	
Musconetcong River	ANS0515	AN0063	Elo	Elo		0				86	
Lamington River	ANS0883	AN0363	Elo	Elo		0				18	
Camp Harmony Branch	ANS0917	AN0390	Elo	Elo						2	
Pequest River	ANS0161	AN0041	Elo	None	Elcom	0				0	
Six Mile Run	ANS0254	AN0409	Elo	None	Elcom	0				0	
Stony Brook	ANS0264	AN0391	Elo	None	Elcom					0	
Big Timber Creek	ANS1220	AN0658	Elo	None	Elcom					0	
		No Catch									
Dry Brook (Jumping Bk?)	ANS0582	AN0019	None	None							
Lopatcong Creek	ANS0499	AN0052	None	None							
Assiscunk Creek	ANS0130	AN0141	None	None							
Primrose Brook (t Primrose?)	ANS0702	AN0215	None	None							
Whippany River	ANS0729	AN0233	None	None							
Trout Brook	ANS0879	AN0359A	None	None							
South Br Rockaway Creek	ANS0887	AN0367A	None	None							
Millstone River	ANS0906	AN0382D	None	None							
Yellow Brook	ANS1006	AN0472	None	None							
Squankum Brook	ANS1041	AN0497A	None	None							
Four Mile Creek	ANS1179	AN0622	None	None							
Manumuskin River	ANS1335	AN0762	None	None							

Table 3.6.3. Mussel assemblages at sample stations of the by NJDFW mussel survey, with land use and first four macroinvertebrate principal components.

Stream	ANSID	Group	Area (km^2)	% Forest	% Ag	% Wet	% Urban	mpca1	mpca2	No Catch
Pequest River	ANS0490	Alhet	273.6	47.3	24.2	13.9	12.6	-0.76	1.05	
Lamington River	ANS0890	Anvar, Alund	256.6	40.6	24.2	7.5	26.6			
Paulins Kill	ANS0128	Alund, Stund	415.0	52.0	17.0	17.0	14.0			
Paulins Kill	ANS0473	Alund, Stund	327.6	49.1	20.1	12.1	14.7	-2.00	2.17	
Lubber's Run	ANS0123	Alund	18.0	63.0	2.0	17.0	18.0	-1.10	-1.97	
Paulins Kill	ANS0128	Alund	415.0	52.0	17.0	17.0	14.0			
Musconetcong River	ANS0077	Alund	314.0	53.0	11.0	14.0	21.0	1.75	1.69	
Miry Run	ANS0577	Alund								
Raritan River - SB	ANS0836	Alund	292.4	42.7	16.2	10.8	27.3	-1.32	0.02	
Musconetcong River	ANS0535	Alund								
South Branch Raritan River	ANS0091	Alund	388.0	41.0	21.0	13.0	25.0	-2.37	1.25	
Musconetcong River	ANS1347	Stund	38.8	47.5	0.2	21.8	28.7			0
Pompeston Creek	ANS0667	Linas								
Raccoon Creek - RM	ANS1250	Linas	92.1	17.2	53.4	9.9	18.8			
Dukes Brook	ANS0896	Linas	11.0	14.9	49.5	13.4	21.2			
Wallkill River	ANS0813	Larad								
Scotland Run	ANS1296	Pycat	70.9	38.0	16.5	17.2	27.1			
Haynes Creek	ANS0650	Pycat	70.2	37.7	1.1	17.3	40.4			
Raccoon Creek	ANS1244	Pycat	10.1	22.0	51.0	12.7	12.8			
Millstone River	ANS0903	Pycat								
Paulins Kill Tributary	ANS0460	Elo	31.0	44.7	21.0	13.9	17.5			
Pequest River	ANS0161	Elo	232.7	45.5	26.3	14.5	11.0	-0.57	0.27	
Beaver Brook	ANS0120	Elo	95.0	46.0	31.0	13.0	10.0	-2.61	1.07	
Musconetcong River	ANS0515	Elo								
Lamington River	ANS0883	Elo	137.8	41.4	22.2	10.2	25.1	-2.78	0.53	
Camp Harmony Branch	ANS0917	Elo	6.5	46.9	4.0	41.6	7.4			0
Six Mile Run	ANS0254	Elo	43.2	14.1	40.8	12.4	31.1	1.43	0.49	
Stony Brook	ANS0264	Elo	41.6	37.9	28.8	18.0	14.9	-0.26	1.56	0
Big Timber Creek	ANS1220	Elo	19.5	25.6	12.6	6.2	54.1			0
Dry Brook (Jumping Bk?)	ANS0582	None	52.9	33.2	6.6	44.0	14.7			0
Lopatcong Creek	ANS0499	None	18.6	45.9	35.4	2.6	16.0	-0.26	1.25	0
Assiscunk Creek	ANS0130	None	81.9	6.2	59.0	21.9	12.8	-0.03	1.31	0
Primrose Brook (t Primrose?)	ANS0702	None	1.4	93.5	0.0	1.2	5.3	-4.19	1.87	0
Whippany River	ANS0729	None	21.3	54.7	1.9	6.2	35.9			0
Trout Brook	ANS0879	None	2.8	28.3	27.6	17.2	26.8			0
South Br Rockaway Creek	ANS0887	None	16.3	32.5	29.6	3.6	31.4	-2.40	1.39	0
Millstone River	ANS0906	None								0
Yellow Brook	ANS1006	None	25.1	9.8	26.0	15.7	47.8			0
Squankum Brook	ANS1041	None	6.8	11.9	18.0	50.9	18.7			0
Four Mile Creek	ANS1179	None	20.6	20.0	12.0	20.5	46.8			0
Manumuskin River	ANS1335	None	23.8	59.1	15.4	18.4	7.0			0

Table 3.6.4. Relationship between mussel assemblages and macroinvertebrate metrics.

				Selected AMNET Metrics				Additional Macroinvertebrate Metrics						
Waterbody	**ANS ID**	**AMNET**	**Mussel assemblage**	**%EPT**	**Ave Tol**	**EPT Fams**	**Fam Rich**	**%Scrap**	**%C-G**	**%Preds**	**%Baetid**	**%Chiro**	**%Hydrops**	**%nonIns**
Paulins Kill	ANS0473	AN0025	Alund Alimp Stund	50.0	4.82	12	22	16.0	20.0	3.0	0.0	2.0	8.0	38.0
Paulins Kill	ANS0481	AN0032	Alund Stund	51.9	3.78	8	15	2.8	3.7	13.0	0.0	0.9	26.9	46.3
Musconetcong River	ANS1347	AN0064	Stund	57.7	4.28	8	16	7.7	11.5	9.6	7.7	26.0	32.7	11.5
Lamington River	ANS0890	AN0370	Anvar Alund	46.4	3.79	12	25	13.4	27.7	7.1	10.7	5.4	8.9	17.0
Musconetcong River	ANS0077	AN0072	Alund	18.8	7.58	6	15	0.0	0.0	1.8	2.7	1.8	10.7	45.5
Raritan R S Br	ANS0091	AN0326	Alund	49.6	4.53	13	28	12.6	7.9	18.1	2.4	3.9	22.0	23.6
Lubbers Run	ANS0123	AN0065	Alund	77.5	3.65	7	15	2.0	4.9	26.5	2.0	9.8	52.0	0.0
Paulins Kill	ANS0128	AN0032	Alund	51.9	3.78	8	15	2.8	3.7	13.0	0.0	0.9	26.9	46.3
Musconetcong River	ANS0535	AN0073A	Alund	43.8	5.12	4	15	5.0	8.3	29.8	3.3	40.5	37.2	7.4
Miry Run	ANS0577	AN0115A	Alund	25.5	4.85	3	11	1.8	0.0	5.5	0.0	39.1	12.7	7.3
S Br Raritan River	ANS0836	AN0322	Alund	55.4	4.09	6	17	4.0	12.9	19.8	2.0	22.8	26.7	9.9
Pequest River	ANS0490	AN0043	Alhet	38.7	5.02	8	18	2.7	15.3	5.3	4.7	32.7	24.0	16.0
Wallkill River	ANS0410	AN0302	Larad	23.0	4.73	7	20	1.0	5.0	7.0	2.0	7.0	9.0	29.0
Pompeston Ck	ANS0666	AN0177	Linas	2.0	7.62	2	18	2.0	40.2	2.9	1.0	16.7	1.0	64.7
Dukes Brook	ANS0896	AN0375	Linas	15.1	6.25	1	16	2.8	9.4	8.5	0.0	12.3	15.1	67.9
Raccoon Ck	ANS1250	AN0685	Linas	0.0	8.61	0	10	2.9	44.7	3.9	0.0	10.7	0.0	66.0
Haynes Creek	ANS0650	AN0168	Pycat	25.5	5.99	4	24	5.9	9.8	25.5	0.0	8.8	22.5	55.9
Millstone River	ANS0903	AN0382A	Pycat	0.0	6.81	0	3	0.0	25.9	0.0	0.0	55.6	0.0	43.0
Raccoon Ck	ANS1244	AN0679	Pycat	4.7	7.29	2	14	2.8	0.9	9.4	0.0	0.9	0.9	87.7
Scotland Run	ANS1296	AN0725	Pycat	51.0	5.40	6	12	4.0	11.0	18.0	0.0	13.0	40.0	34.0
Beaver Brook	ANS0120	AN0047	Elo	69.9	3.38	12	23	10.7	15.5	3.9	1.0	2.9	17.5	13.6
Pequest R	ANS0161	AN0041	Elo	36.6	4.57	7	17	4.0	4.0	19.8	1.0	15.8	25.7	19.8
Sixmile Run	ANS0254	AN0409	Elo	22.3	4.45	3	14	0.0	4.9	11.7	0.0	2.9	12.6	50.5
Stony Bk	ANS0264	AN0391	Elo	28.3	5.16	10	25	3.9	11.0	8.7	2.4	45.7	6.3	7.9
Wallkill River	ANS0410	AN0302	Elo	23.0	4.73	7	20	1.0	5.0	7.0	2.0	7.0	9.0	29.0
UNT to Paulins Kill	ANS0460	AN0014	Elo	25.7	5.79	5	17	7.9	4.0	5.9	0.0	9.9	19.8	56.4
Musconetcong River	ANS0515	AN0063	Elo	39.3	4.24	7	17	2.8	5.6	0.9	13.1	15.9	14.0	37.4
Lamington River	ANS0883	AN0363	Elo	73.9	3.13	12	19	5.9	13.4	9.2	1.7	17.6	21.8	0.8
Camp Harmony Branch	ANS0917	AN0390	Elo	21.2	5.79	9	23	1.0	9.6	1.9	1.9	4.8	0.0	27.9
S Br Big Timber Ck	ANS1220	AN0658	Elo	15.5	6.37	4	19	14.6	34.0	18.4	5.8	14.6	1.0	45.6
Assicunk Creek	ANS0130	AN0141	None	22.0	6.51	3	19	3.0	29.0	30.0	0.0	15.0	19.0	44.0
Dry Brook	ANS0465	AN0019	None	50.0	5.66	9	27	14.7	18.6	17.6	2.9	15.7	21.6	30.4
Lopatcong Ck	ANS0499	AN0052	None	25.4	4.79	8	20	6.1	12.3	6.1	2.6	35.1	7.0	6.1
trib Primrose Brook	ANS0702	AN0215	None	75.9	3.14	15	25	18.5	20.4	0.9	0.9	14.8	4.6	6.5
Whippany River	ANS0729	AN0233	None	30.4	4.98	8	17	0.0	9.8	3.9	2.9	27.5	7.8	28.4
Trout Bk	ANS0879	AN0359	None	40.6	4.43	11	22	5.9	5.9	5.0	8.9	9.9	5.0	41.6
S Br Rockaway Ck	ANS0887	AN0367	None	50.5	3.66	8	20	25.7	41.6	7.9	5.0	18.8	5.9	16.8
Millstone River	ANS0906	AN0382D	None	5.0	6.30	2	14	10.9	22.8	5.0	0.0	55.4	0.0	19.8
Yellow Bk	ANS1006	AN0472	None	19.0	6.08	2	15	3.0	19.0	13.0	0.0	28.0	18.0	43.0
Squankum Bk	ANS1041	AN0497	None	29.4	5.06	7	27	7.8	14.7	18.6	0.0	14.7	3.9	23.5
Four Mile Branch	ANS1179	AN0622	None	33.6	5.80	11	21	21.5	26.2	10.3	1.9	19.6	0.9	41.1
Manumuskin River	ANS1335	AN0762	None	38.6	4.86	9	21	0.0	25.7	5.9	0.0	24.8	5.0	19.8

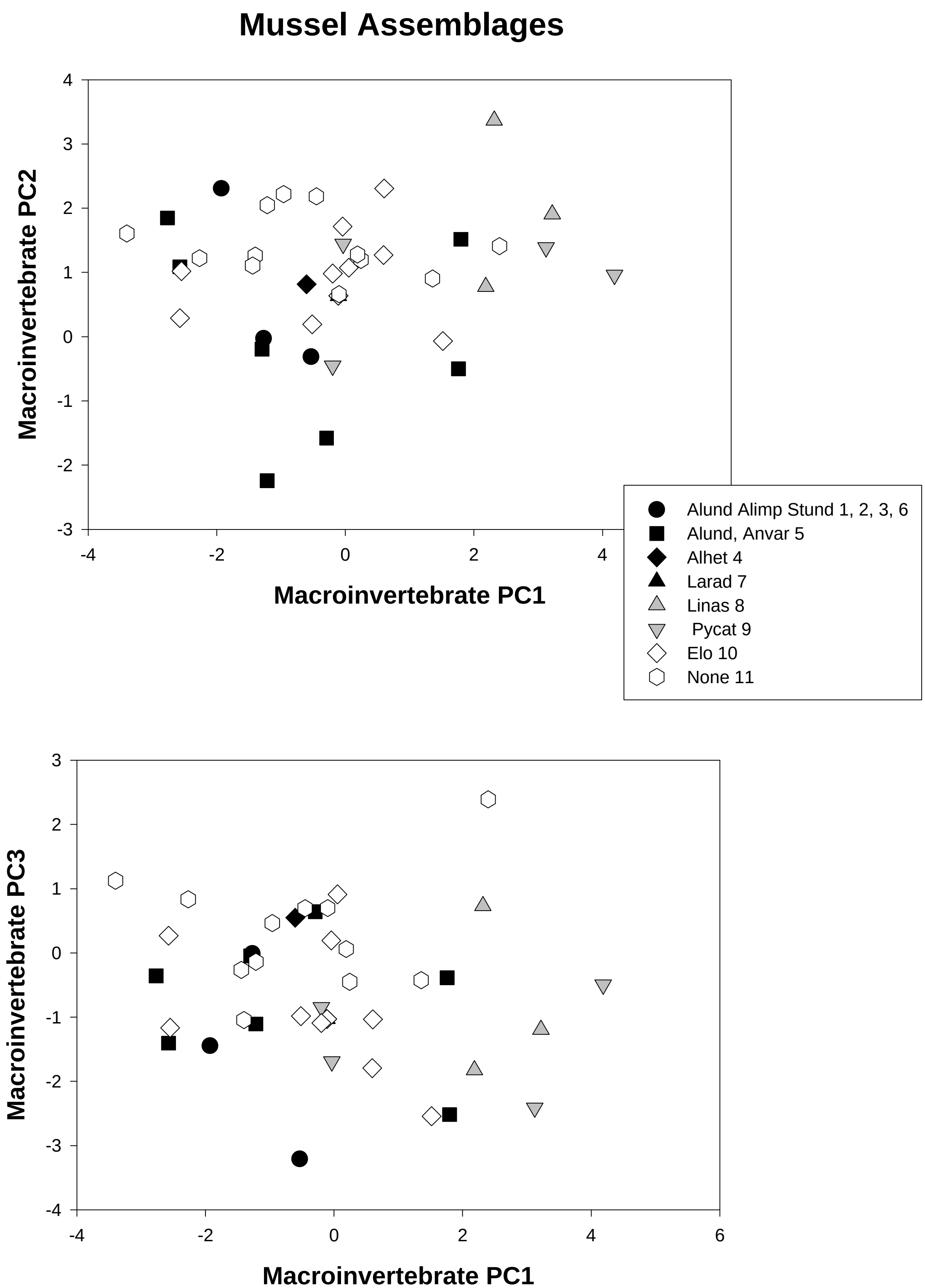
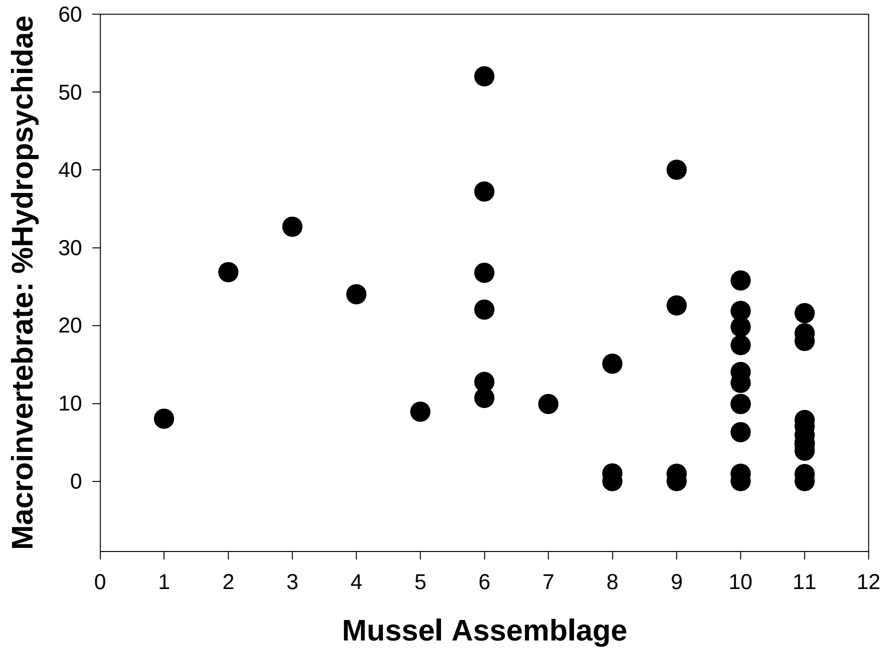


Figure 3.6.1. Position of mussel sites on macroinvertebrate principal component coordinates: mpca1 and mpca2 (top) and mpca1 and mpca3 (bottom). Points are labeled by observed mussel assemblage at the site. See text for explanation of assemblages and Table 3.6.1 for mussel species names.

Mussel Assemblage and associated AMNET samples



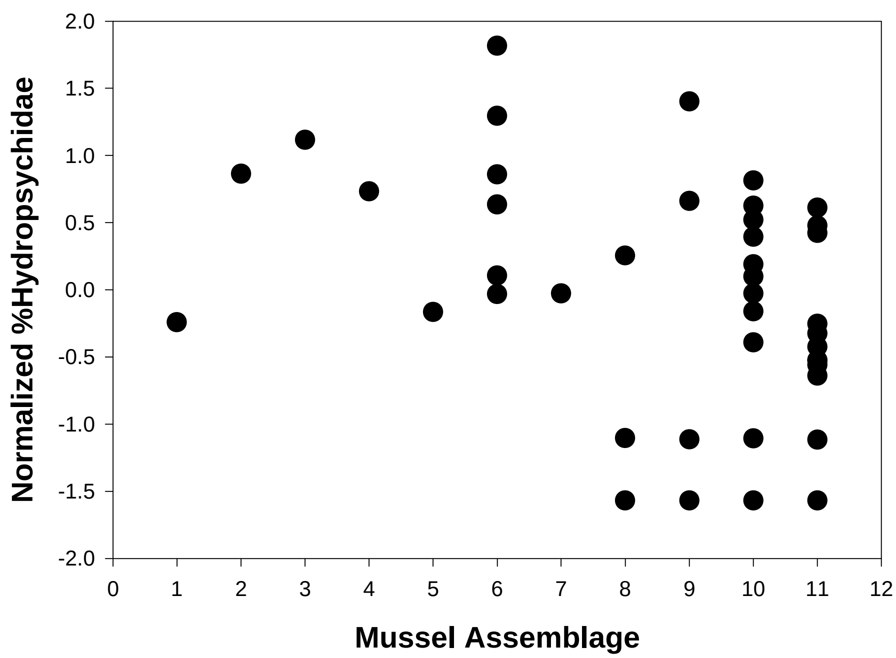
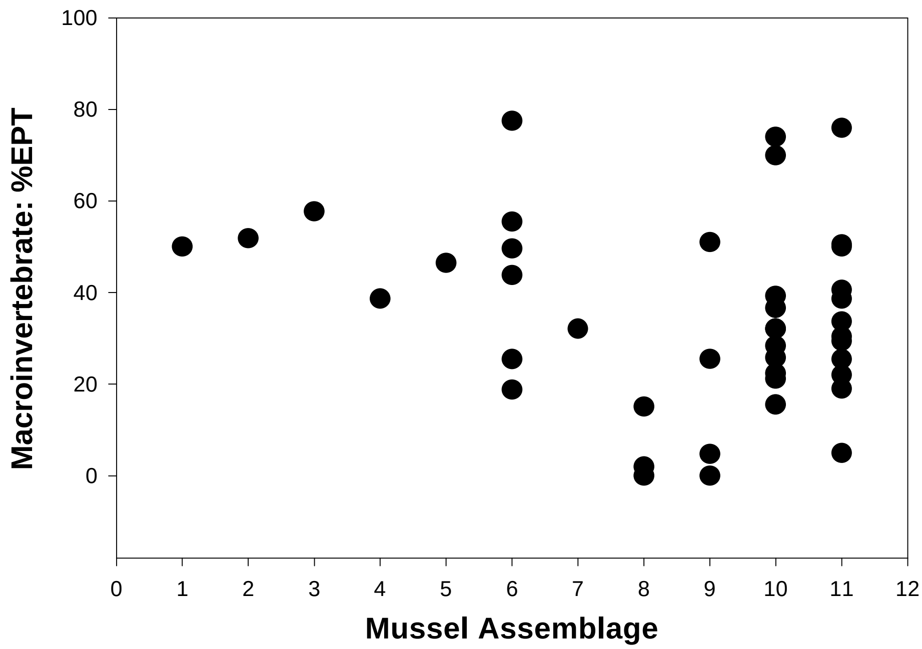


Figure 3.6.2. Relationship between %Hydrop (percentage hydropsychid caddisflies) for sites with different observed mussel assemblages: Alun Alvar Stun (1), Alund Stund (2), Stund (3), Alhet (4), Anvar Alund (5), Alund (6), Larad (7), Linas (8), Pycat (9), Elo (10), None (11). See text for explanation of mussel assemblages and Table 3.6.1 for mussel species names.

Mussel Assemblages and associated AMNET samples



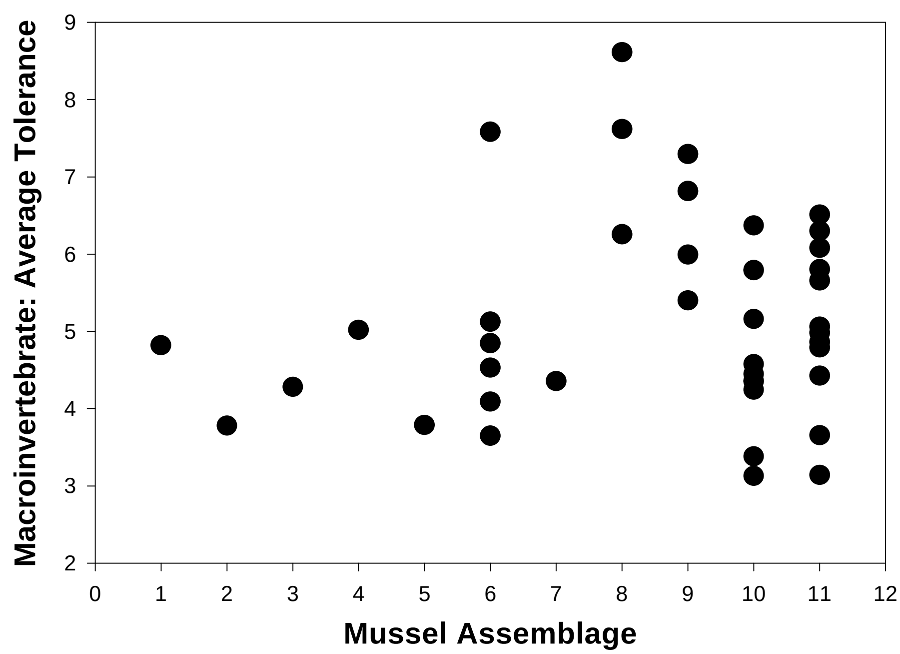
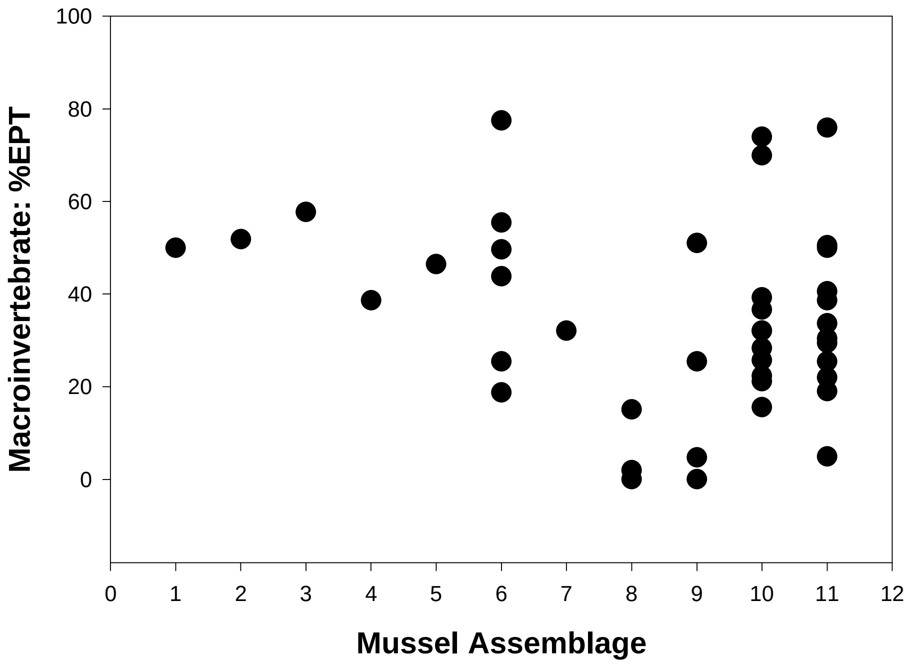


Figure 3.6.3. Relationship between %EPT (percentage EPTs) (top) and Average Tolerance (bottom) for sites with different observed mussel assemblages: Alun Alvar Stun (1), Alund Stund (2), Stund (3), Alhet (4), Anvar Alund (5), Alund (6), Larad (7), Linas (8), Pycat (9), Elo (10), None (11). See text for explanation of mussel assemblages and Table 3.6.1 for mussel species names.

Mussel Assemblages and associated AMNET samples



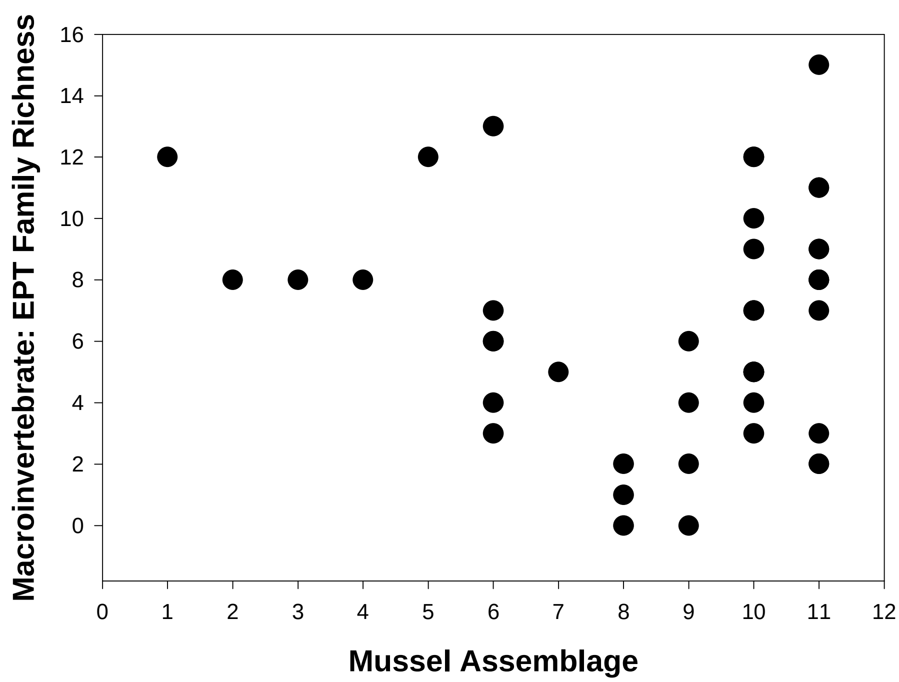


Figure 3.6.4. Relationship between %EPT (percentage EPTs) (top) and EPT family richeness (bottom) for sites with different observed mussel assemblages: Alun Alvar Stun (1), Alund Stund (2), Stund (3), Alhet (4), Anvar Alund (5), Alund (6), Larad (7), Linas (8), Pycat (9), Elo (10), None (11). See text for explanation of mussel assemblages and Table 3.6.1 for mussel species names.

3.7 Odonate Communities and Macroinvertebrate Relationships

Evidence (adult, larvae, exuvia, ovipositioning, or mating) of 105 odonate species were seen at sampling stations (Table 3.7.1). *Calopteryx maculata* was the most common species present in 67 of the 68 samples. *Boyeria vinosa, Argia moesta, Ischnura verticalis,* and *Argia fumipennis* were present in more than 30 samples. All 61 sites had adult odonates present, although only 33 and 38 sites, respectively, had mating and ovipositoning species. Larvae were identified at 17 sites and exuvia at 25 sites.

Although no official listing process has been completed, preliminary status rankings have identified some of the taxa as Species of Special Concern or Threatened Species (subsequent discussions will use these unofficial rankings). Species of Special Concern applies to species that warrant special attention because of some evidence of decline, inherent vulnerability to environmental deterioration, or habitat modification that would result in their becoming a Threatened species. This category would also be applied to species that meet the foregoing criteria and for which there is little understanding of their current population status in the state. The classification “Threatened” applies to species that may become Endangered (prospects for survival within the state are in immediate danger) if conditions surrounding it begin to or continue to deteriorate. Thus, a Threatened species is one that is already vulnerable as a result of, for example, small population size, restricted range, narrow habitat affinities, significant population decline, etc.

Seven species (*Cordulegaster obliqua, Ophiogomphus mainensis, Enallagma recurvatum, Stylurus scudderi, Libellula auripennis, Macromia alleghaniensis, Enallagma pictum*) observed were classified as Special Concern, while three species (*Gomphus apomyius, Ophiogomphus aspersus, Epitheca spinosa*) were classified as threatened. Most species with conservation concerns were collected from fewer than five sites, with only *Macromia alleghaniensis* and *Enallagma pictum* (special concern) collected from eight sites each.

Although the models were not significant ($p>0.23$), macroinvertebrate metrics and land use variables accounted for 39.3% and 47.6% of the variation in Odonate adult and larvae richness, respectively. Odonate Adult Richness was positively associated with EPA Habitat Scores (Fig. 3.7.1) ($p=0.087$) and Odonate Larvae Richness was positively associated with the abundance of collector-gatherers ($p=0.098$) and scrapers ($p=0.073$) (Fig. 3.7.2). Both richness measures showed positive relationships with normalized macroinvertebrate family richness and Simpson’s diversity metrics ($p=0.040$-0.061) (Figs. 3.7.3 and 3.7.4). Regression models predicting species richness of exuvia, oviposition, and mating accounted for 23.9%, 22.3%, and 19.1%. Like larvae richness, exuvia richness was positively associated with scraper abundance while both oviposition and mating richness metrics were positively related to Chiro:EPT metrics.

With the exception of odonate adult richness, regression of macroinvertebrate PCs against odonate richness metrics were not significant and did not account for much variation in odonate richness. There was a significant positive relationship between odonate adult richness and MPCA3 ($p=0.003$) and MPCA4 ($p=0.049$) (Fig. 3.7.5) with the model accounting for 18.6% of variation among samples ($p=0.012$). Remaining odonate metric-MPCA regressions were not significant and accounted for less than 8% of variation.

Table 3.71. Common and scientific names, number of samples and status of odonates collected in the NJDFGW survey.

Common Name	Scientific Name	# Samples	Status
Arrowhead Spiketail	*Cordulegaster obliqua*	1	Special Concern
Ashy clubtail	*Gomphus lividus*	1	
Attenuated Bluet	*Enallagma daeckii*	1	
Banner clubtail	*Gomphus apomyius*	1	Threatened
Brook snaketail	*Ophiogomphus aspersus*	1	Threatened
Carolina Saddlebags	*Tramea carolina*	1	
Chalk-fronted corporal	*Ladona julia*	1	
Common baskettail	*Epitheca cynosura*	1	
Common bluet		1	
Dot-tailed whiteface	*Leucorrhinia intacta*	1	
Double-ringed Pennant	*Celithemis verna*	1	
Great blue skimmer	*Libellula vibrans*	1	
Green-striped darner	*Aeshna verticalis*	1	
Maine Snaketail	*Ophiogomphus mainensis*	1	Special Concern
Mottle darner	*Aeshna clepsydra*	1	
Pine Barrens bluet	*Enallagma recurvatum*	1	Special Concern
Russet-tipped Clubtail	*Stylurus plagiatus*	1	
Rusty snaketail	*Ophiogomphus rupinsulensis*	1	
Seaside dragonlet	*Erythrodiplax berenice*	1	
Sphagnum sprite	*Nehalennia gracilis*	1	
Stripe-winged baskettail	*Epitheca costalis*	1	
Stygian shadowdragon	*Neurocordulia yamaskanensis*	1	
Taper-tailed darner	*Gomphaeschna antilope*	1	
Turquois bluet	*Enallagma divagans*	1	
Atlantic bluet	*Enallagma doubledayi*	2	
Aurora damsel	*Chromagrion conditum*	2	
Blue dragonlet	*Erythrodiplax minuscula*	2	
Blue-ringed Dancer	*Argia sedula*	2	
Calico Pennant	*Celithemis elisa*	2	
Cherry-faced Meadowhawk	*Sympetrum internum*	2	
Comet darner	*Anax longipes*	2	
Dusky Dancer	*Argia translata*	2	
Eastern Red damsel	*Amphiagrion saucium*	2	
Fine-lined Emerald	*Somatochlora filosa*	2	
Seepage dancer	*Argia bipunctulata*	2	

Table 3.7.1 (continued). Common and scientific names, number of samples and status of odonates collected in the NJDFGW survey.

Common Name	Scientific Name	# Samples	Status
Southern Pygmy clubtail	*Lanthus vernalis*	2	
Stream cruiser	*Didymops transversa*	2	
Swift river cruiser	*Macromia illinoiensis*	2	
Treetop emerald	*Somatochlora provocans*	2	
Twin-spotted Spiketail	*Cordulegaster maculata*	2	
Umber shadowdragon	*Neurocordulia obsoleta*	2	
Yellow-sided skimmer	*Libellula flavida*	2	
Zebra clubtail	*Stylurus scudderi*	2	Special Concern
American rubyspot	*Hetaerina americana*	3	
Big Bluet	*Enallagma durum*	3	
Common spreadwing	*Lestes disjunctus*	3	
Cyrano darner	*Nasiaeschna pentacantha*	3	
Familiar forktail		3	
Least clubtail	*Stylogomphus albistylus*	3	
Lilypad Forktail	*Ischnura kellicotti*	3	
Mantled Baskettail	*Epitheca semiaquea*	3	
Shadow darner	*Aeshna umbrosa*	3	
Spot-winged Glider	*Pantala hymenaea*	3	
White Corporal	*Ladona exusta*	3	
Blackwater Bluet	*Enallagma weewa*	4	
Citrine Forktail	*Ischnura hastata*	4	
Elfin Skimmer	*Nannothemis bella*	4	
Martha's pennant	*Celithemis martha*	4	
Petite Emerald	*Dorocordulia lepida*	4	
River jewelwing	*Calopteryx aequabilis*	4	
Robust baskettail	*Epitheca spinosa*	4	Threatened
Spine-crowned clubtail	*Gomphus abbreviatus*	4	
Arrow clubtail	*Stylurus spiniceps*	5	
Banded pennant	*Celithemis fasciata*	5	
Golden-winged Skimmer	*Libellula auripennis*	5	Special Concern
Spangled Skimmer	*Libellula cyanea*	5	
Black Saddlebags	*Tramea lacerata*	6	
Blue-tipped dancer	*Argia tibialis*	6	
Swamp Spreadwing	*Lestes vigilax*	6	
Bar-winged Skimmer	*Libellula axilena*	7	
Dragonhunter	*Hagenius brevistylus*	7	

Table 3.7.1 (continued). Common and scientific names, number of samples and status of odonates collected in the NJDFGW survey.

Common Name	Scientific Name	# Samples	Status
Rambur's forktail	*Ischnura ramburii*	7	
Allegheny River Cruiser	*Macromia alleghaniensis*	8	Special Concern
Eastern Amberwing	*Perithemis tenera*	8	
Prince Baskettail	*Epitheca princeps*	8	
Scarlet Bluet	*Enallagma pictum*	8	Special Concern
Springtime darner	*Basiaeschna janata*	8	
Wandering glider	*Pantala flavescens*	8	
Familiar bluet	*Enallagma civile*	9	
Harlequin Darner	*Gomphaeschna furcilata*	9	
Painted Skimmer	*Libellula semifasciata*	9	
Black-shouldered spinyleg	*Dromogomphus spinosus*	10	
Blue corporal	*Ladona deplanata*	10	
Twelve-spotted skimmer	*Libellula pulchella*	10	
Common sanddragon	*Progomphus obscurus*	11	
Sparkling Jewelwing	*Calopteryx dimidiata*	11	
Widow Skimmer	*Libellula luctuosa*	11	
Yellow-legged meadowhawk	*Sympetrum vicinum*	11	
Lancet clubtail	*Gomphus exilis*	12	
Common whitetail	*Plathemis lydia*	13	
Illinois River cruiser	*Macromia illinoiensis*	14	
Blue-fronted dancer	*Argia apicalis*	15	
Swamp Darner	*Epiaeschna heros*	15	
Skimming Bluet	*Enallagma geminatum*	16	
Stream bluet	*Enallagma exsulans*	16	
Eastern Pondhawk	*Erythemis simplicicollis*	20	
Fragile forktail	*Ischnura posita*	25	
Slatey Skimmer	*Libellula incesta*	25	
Common green darner	*Anax junius*	26	
Blue dasher	*Pachydiplax longipennis*	27	
Fawn darner	*Boyeria vinosa*	32	
Powdered dancer	*Argia moesta*	32	
Eastern forktail	*Ischnura verticalis*	33	
Variable dancer	*Argia fumipennis*	49	
Ebony jewelwing	*Calopteryx maculata*	67	

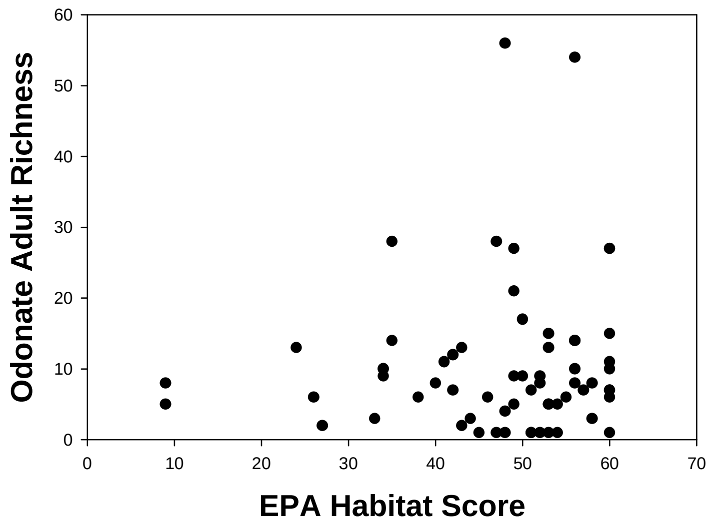


Figure 3.7.1. Relationship between odonate adult species richness and EPA Habitat score.

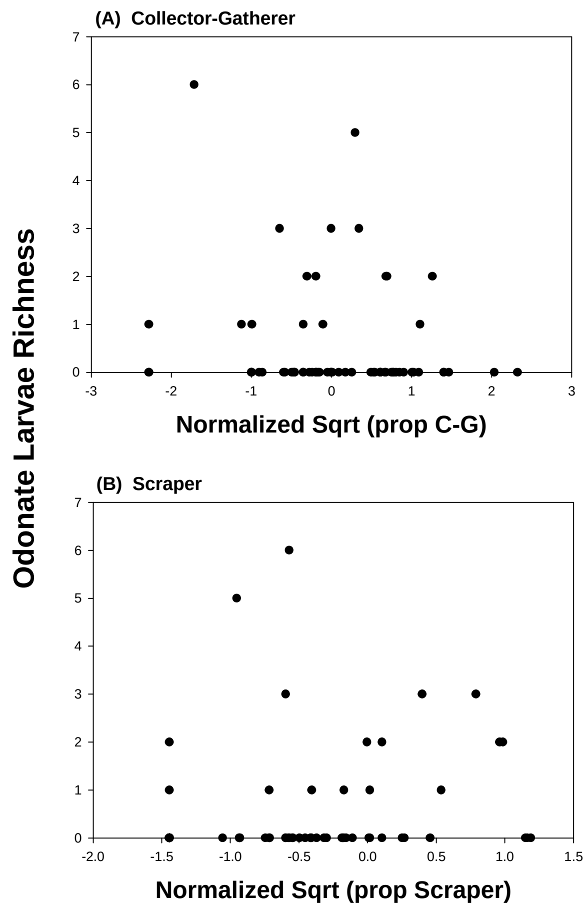


Figure 3.7.2. Relationship between odonate larval species richness and (A) normalized sqrt (proportion Collector-Gatherers) and (B) normalized sqrt (proportion Scrapers).

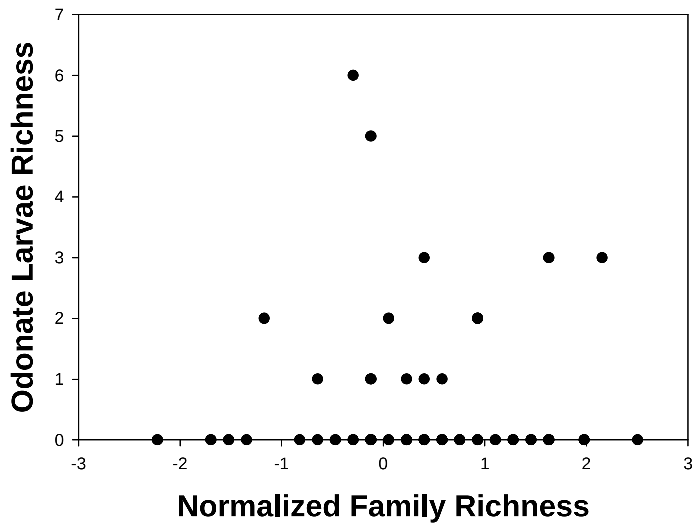


Figure 3.7.3. Relationship between odonate larval species richness and normalized Family Richness.

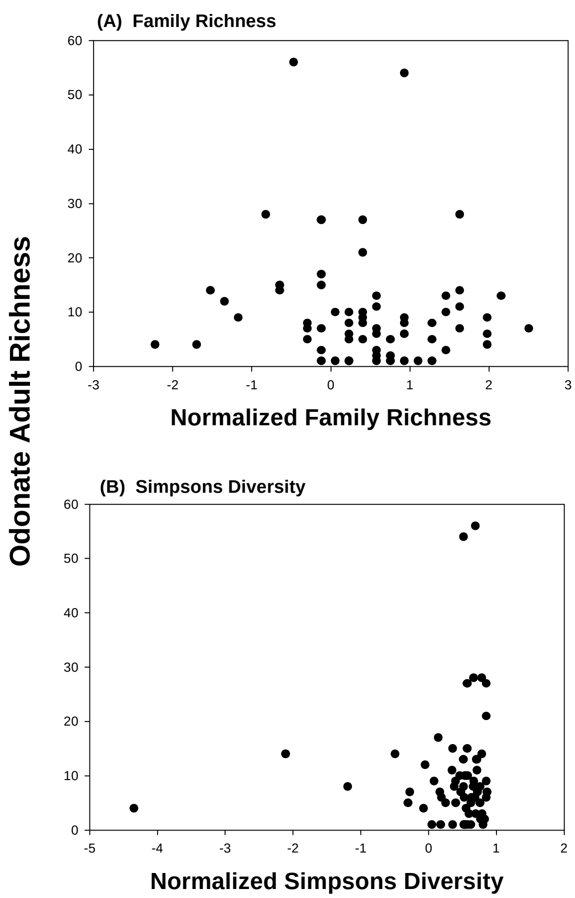


Figure 3.7.4. Relationship between odonate adult species richness and (A) normalized Family Richness and (B) normalized Simpsons Diversity.

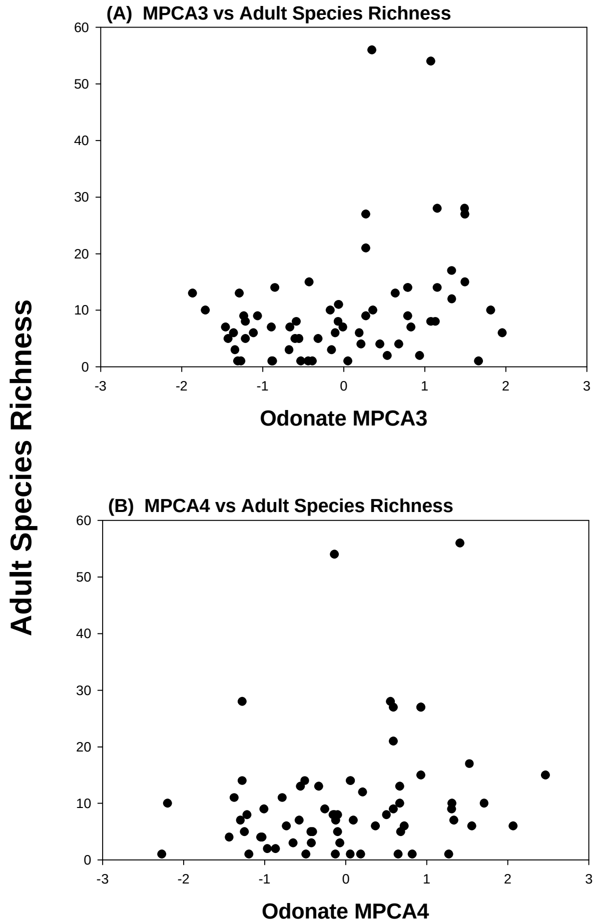


Figure 3.7.5. Relationship between odonate adult species richness and (A) MPCA3 and (B) MPCA4.

4. DISCUSSION

4.1 Relationships between metrics and indices from different taxa

There are three approaches to the use of indices of biotic assemblages for bioassessment. All three posit a relationship between anthropogenic effects and assemblage structure, but differ in the use of assemblage data. The first approach establishes explicit relationships between stressors and assemblage structure. These stressor-assemblage relationships can be used to develop models for inferring one variable from the other. Specifically, assemblage data can be used to estimate stressor levels. This can be used where direct estimates of stressor levels are not available or where they would be too expensive or time-consuming to measure. Assemblage data may be useful in providing an integrated measure of conditions over time, which may be more accurate than single point measurements of stressor levels. The second approach uses assemblage structure as a predictor of other aspects of system condition. As with the first approach, this may be useful where the system variable of primary interest is not as easily measured. The approach may also be useful as a warning signal to provide evidence of change in system conditions which anticipate significant ecological problems. For example, the analyses of algae data and metrics done for NJDEP by ANSP related algal metrics to nutrient concentrations (i.e., stressor levels) and system conditions (nuisance algal growths). These represent applications of the first and second approaches. The third approach treats assemblage structure as the variable of interest, so that changes in assemblage structure due to anthropogenic effects directly measure ecological impairment. IBIs use metrics to define ecologically relevant and important aspects of assemblage structure. For this approach, definition of reference conditions is important to determine the magnitude of change represented by current conditions. Direct measurement of stressor levels is not inherently necessary for this approach, though stressor information is often important in developing and interpreting IBIs. Note that using this third approach, indices for different taxa need not be similar. If a taxon is insensitive to a stressor, it may show no impairment at levels where another taxon shows high impairment. The NJFIBI and AMNET indices are examples of the third kind of index. In developing these indices, sites across a range of stressor levels were studied, and the range of values was used to define critical levels for the various metrics.

Changes in indices are commonly considered to be biological impairment. Since impairment is a value-based idea, this equivalence involves a tacit assumption that change from “undisturbed” condition is equivalent to a loss in value (Karr, 2004 [NE game talk]). In the remainder of the discussion, impairment will be used to describe change in condition from undisturbed condition. However, it is also necessary to define levels of change that would be considered moderately or severely impaired.

The measurement of condition of a stream reach is subject to several sources of variation:

1) Sampling error in collections will affect estimation of metrics, creating sampling variation;

2) Since condition of a reach is typically inferred from condition within a sample from a relatively small sample area within the reach, there may be spatial variability in condition;

3) since condition is typically inferred from a sample taken at one time, there may be temporal variability in condition.

Measures of condition at a site before disturbance are almost never available, so the measurement of change is typically done by several forms of inference:

1) Measurement of condition within reference sites. These measurements are subject to several kinds of bias and imprecision:

> a) Few or no truly undisturbed sites may be available for sampling, so that quality of available reference sites is under-estimated, therefore under-estimating the amount of change.
>
> b) Available reference sites are biased toward geological and topographical conditions which discouraged development. For example, high-gradient streams with rocky, low-nutrient soils may be over-represented, since these would have been subject to less agricultural use.
>
> c) Reference conditions typically vary among streams. In many cases, it is possible to relate this variation to differences among streams, e.g., differences in stream size, gradient, and habitat complexity. This variation may be accounted for by calibrating metrics for these variations (e.g., for stream size), by defining different indices for different conditions (e.g., high-gradient and low-gradient streams), by defining sampling protocols to minimize variation (e.g., choosing sampling areas with a mix of habitat types), or by choosing metrics which are relatively insensitive to such variation.
>
> d) Even after adjustment for known sources of variation, there is typically much variation in reference conditions. It is often difficult to apportion this residual variability to natural or anthropogenic influence. Sites with apparently “low” condition may be considered to reflect natural variability or effects of an unknown and unmeasured anthropogenic influence.

2) Use of historical records to reconstruct historic assemblage characteristics. These are subject to several biases, as well:

> a) The earliest detailed historic data typically post-date significant anthopogenic disturbance.
>
> b) Species may be under-represented in historical records, because they were difficult to capture by techniques in use, poorly known or diagnosed taxonomically, of lower recreational interest, or sufficiently common that specific documentation was not made.
>
> c) Uncommon species may be over-represented, since collectors tended to preserve these and may have targeted sites where certain species were previously known or would be suspected to occur; thus, occurrence in historic records doesn’t necessarily indicate ubiquity of occurrence.

3) Analyses of stressor-metric relationships may be used to indicate directions of response. The extremes of these relationships may be useful in assigning reference levels.

Since metrics and calibration of metrics are derived from the definition of reference conditions, these biases will affect the accuracy of measures of impairment.

Correlations among indices for different taxa will depend on the variation within each index, as discussed above, as well as inherent differences in the responses of the taxa to various factors. Indices are defined to receive maximum values under regional reference conditions. If indices are defined and calibrated along a similar gradient sites, minimum values of each index would be expected at the most degraded sites. There may be some differences among indices created by different endpoints. No index can be calculated for sites with no fish, while algal indices might still be defined, so that minimum values for fish indices would occur at sites with low, but above minimum values for algal indices. However, while this affects absolute values of indices, this is likely to have only a small effect on correlations among indices. Thus, an inherent positive correlation among indices is expected, with the magnitude dependent on the similarity of response to stressors among the taxa (as well as the variation within each index, as discussed above). For example, different points of inflection for sigmoid responses to stressors weaken correlations among indices (Fig. __). Different thresholds for different stressors will further reduce correlations among indices.

Therefore, interpreting relationships among indices to discern impairment involves three related tasks:

1) Estimation of relationships among indices. This is discussed in the following section.

2) Distinguishing different responses of taxa from inherent variability within single-taxon indices. Where inherent variability is high, a difference in measured values of different indices may reflect this variability rather than differences in responses among taxa. These issues are discussed in two subsequent sections on variability within indices and differential responses between taxa.

3) Defining levels of impairment corresponding to different levels of impairment within taxa. If differences in responses can be established, there is frequently a need (often regulatory) to place the reach on a single scale of impairment. For example, if fish and algal indices showed no impairment, while the macroinvertebrate index showed high impairment, would the reach be considered highly impaired? This question is largely a policy issue. However, it will be discussed within a subsequent section on implications for integrated assessment.

Relationships among indices

We examined metric patterns between taxonomic groups and found that each of the pairwise comparisons of measures for different taxonomic groups showed some level of correlation. For full data sets (i.e., all samples for which joint data are available), the correlations are generally weak, as indicated by low r^2 values. However, the relationships are highly significant. Correlations are higher, though less significant for some subsets of the data, such as the NJ FIBI fish samples and associated AMNET samples, or macroinvertebrate-algal samples from single projects or drainages. This pattern is expected from the nature of the analyses. The larger data set includes data from different parts of region and collected by different groups, so a greater variability is expected within taxonomic groups and in comparisons among taxonomic groups. However, the larger number of samples for the full analyses produces greater significance levels. The comparison of the AMNET and FIBI scores for the NJ FIBI sites found an r^2 of 0.15. Much of the variability was due to relatively low fish IBI scores at sites with moderate to high AMNET scores.

Relatively low correlations among indices of different taxa have been found in other studies. In a study of Coastal Plain streams in South Carolina, Paller (2001) compared fish IBI scores with a benthic macroinvertebrate IBI score (HDMI) derived from Hester-Dendy artificial substrate samplers. He found an r^2 of 0.39 with the greatest difference between the indices observed at sites with high or moderate disturbance. However, some sites with high fish IBI scores had low HDMI scores, a pattern opposite that seen in the NJ FIBI data. Bryce and Hughes (2003) calculated IBI's for fish, macroinvertebrates, and diatoms along a disturbance gradient (a mix of agriculture, mining, acidic deposition and logging) at a series of sites in the mid-Atlantic Highlands. Over all sites, they found decreasing integrity in all three groups with increasing disturbance, but high variability, especially among disturbed sites. They divided their data into two gradients: a mining and an agriculture gradient (some sites were in both gradients). Graphs of the data allow estimates of correlation among the metrics (Table 4.1.1). Correlations were high for the agriculture gradient, but lower for the mining gradient.

Table 4.1.1 Pearson correlation coefficients (r) calculated for fish, macroinvertebrate and diatom IBI's, estimated from graphs in Bryce and Hughes 2003. Data above the diagonal are for the agriculture gradient, and data below the diagonal are for the mining gradient.

	Macroinvertebrates	Diatoms	Fish
Macroinvertebrates		0.62	0.58
Diatoms	0.42		0.63
Fish	0.23	0.07	

Griffith, et al. (2005) compared fish, macroinvertebrate and periphyton indices in 86 sites in the Rocky Mountains in Colorado. Fish metrics were species richness, fish abundance, relative abundance of Salmonidae, relative abundance of native fish, and *Onchorhnnchus*/Salmonidae ratio (proportion of Salmonidae that are rainbow or cutthroat trout). Macroinvertebrate metrics were Ephemeroptera (E) and Plecoptera (P) genera richness, Trichoptera (T) genera richness, relative abundance of crustacea and mollusca, Chironomidae (Ch) genera richness, relative abundance of 5 most dominant genera, macroinvertebrate density, relative abundance of EPT, Orthocladiinae/Ch ratio, Tanytarsini/Ch ratio and Hilsenhoff's biotic richness. Periphyton metrics were algal division richness, non-diatom general richness, relative abundance of diatoms, diatom tolerance value, relative abundance of *Acnnanthes minutissima*, relative abundance of nitrogen-heterotrophic diatoms, diatom abundance, algal cell abundance, chlorophyll and biomass. These fish metrics are rather different from the FIBI metrics, reflecting the major differences in assemblage richness and taxonomic composition of the two regions. The macroinvertebrate metrics include metrics similar to the AMNET indices, as well as metrics reflecting abundance, importance of some non-insect taxa, and Chironomid assemblage structure. The Rocky Mountain periphyton metrics include some that are parallel to the diatom metrics used in this study, but include a larger variety of metrics. The three metrics tended to have relatively small, but positive correlations (Table 4.1.2), similar to results of this study and Bryce and Hughes (2003).

Table 4.1.2 Pearson correlation coefficients (r) of fish, macroinvertebrate and periphyton indices from Griffith, et al. (2005). Values above the diagonal are based on use of all 86 samples, and values below the diagonal are based on exclusion of 18 sites where no fish were found.

	Macroinvertebrates	Periphyton	Fish
Macroinvertebrates		0.325	0.419
Periphyton	0.091		0.338
Fish	0.006	0.338	

Variability within indices

As noted above, individual indices are subject to several sources of variability:

A. sampling error and precision of links between different samples

B. assessment issues such as definition and calibration of metrics and indices, and

A. *Sources of error*– All metrics are estimates of assemblage properties from samples, with concomitant sampling error. In general, a sample is used to estimate condition in a stream reach and for a period of time. Thus, sampling error can be treated as having several components: true sampling error (related to estimating a metric for that specific site and time period), components related to spatial variation within the reach, and factors associated with temporal variation. Others have analyzed variability of metrics (e.g., Angermeier and Karr 1986, Karr and Chu 1999). For this study, we focused on correlations among samples. Therefore, the latter two components are important where linked samples of different taxa are taken at different places or time periods (i.e., if samples are taken at exactly the same time and place, the spatial and temporal variation would not affect correlations among taxa). However, the true sampling error will always affect inter-correlations.

Sampling error can be affected by protocol or site-specific biases (e.g., different probabilities of capturing, sorting, or identifying taxa or groups of taxa; sample-size dependent bias in estimating species richness measures; biased selection of specific sampling based on habitat or other factors), and by unbiased variance. For example, in the simplest sampling model, the coefficient of variation of a proportional metric in a sample or subsample is sqrt((1-p)/pn) where p is the true proportion and n is the number of individuals . Therefore, the coefficient of variation for a 100-count subsample is at least 10% for groups comprising less than half the sample and can be appreciable for rare groups.

Values for some of the primary NJ proportion metrics (e.g., %EPT, %salmonid/top carnivorous fish) as well as some of the other metrics analyzed (e.g., %Shredder macroinvertebrates, %Intolerant fish) are often low.

This variation (and consequently inter-correlation among metrics) can be affected by impairment. Under references conditions, communities are typically stable with lower diversity, and the probability of obtaining representative samples is high. Similarly, under conditions of severe stress, taxa diversity is low with only the most tolerant taxa present. It is at the intermediate levels of impairment where sensitive taxa are still present, but in low abundance that sampling variance can be at its greatest (Connel 1978, Townsend and Scarsbrook 1997). Although sampling error may still be significant, the use of multiple metrics reduces the effects of error in each metric by distributing it across measures.

Imperfect links between samples was an additional source of error in this study. Ideally, correlations would be assessed on samples of different taxa taken at the same time and place. Many samples in this study were obtained from programs designed as multi-taxonomic programs in which sampling was closely linked. For example, for several of the ANSP studies (e.g., the dam study and land use pattern study) involved fish, macroinvertebrate, and algae sampling. In most cases, macroinvertebrate and algal samples were usually collected on the same day or within a few days, with fish samples usually collected within 10 days. However, some of the linked data used in the study were from different programs. In particular, many of the NJ fish samples (e.g., from the EPA FIBI development and NJ FIBI study) were linked with AMNET data. These data were often taken at the same station, but sometimes samples were several km apart (in order to provide sufficient habitat for fish). Correlations among macroinvertebrate and fish components were higher when limited to samples taken at the same site, showing that the linking of samples did affect correlation in this study. Additionally, samples were usually taken in different years. The AMNET data used were from samples collected between 1993 and 1998 (the 1998 data were the primary source of data used in comparisons). The EPA FIBI development data were collected 1990-1996 and the NJ FIBI data were collected 2000-2003. Associated land use information was predominantly 1995 data. These spatial and temporal differences in the linked samples introduces variation related to among-year differences in weather and hydrology, trends in land use (e.g., increased development), and spatial differences in ecological condition.

Variation in metric relationships is also introduced by differences in sampling programs. Pearson correlation analyses of metrics for the entire algal dataset (n=242) and the NJ Algal Indicators of Eutrophication project (n=62) both showed that the Shannon-Weiner Diversity index was highly positively correlated with Diatom Species Richness and highly negatively correlated with % Dominant Taxon and % Dominants metrics. However, in the NJ Algal Indicators of Eutrophication project the % Dominants metric was positive correlated with % Dominant taxon as well and removed from subsequent analyses. This variation translates into different relationships with macroinvertebrates (Sections 3.4.3 and 3.5.1) and contributes to the noise present in the larger datasets.

B. *Definition and calibration of metrics*– The development of a multimetric approach to stream assessment typically involves the selection and calibration of metrics to characterize reference conditions that will form the basis for assessment, and the assessment of biological condition at sites and judgment of impairment. Biological communities deviate predictably from reference sites with changes in specific classification variables. For example, biological communities have been shown to change predictably with changes in elevation (Barbour et al. 1992, 1994; Spindler 1996) and drainage area (Ohio EPA 1987).

Land use and watershed area data were available for the majority of sites and used as predictor variables for variation in metrics. There were large differences among relationships of the various algal, macroinvertebrate, and fish metrics to land use. Several of the metrics, such as %Dominant macroinvertebrate family and %White sucker, showed weak relationships with land use. For several metrics (e.g. Diatom Richness, %C:P), there was little relationship with drainage area. These findings don't prove lack of response since the land use is only a crude measure of stress, and metrics may be responding to aspects of disturbance which were not measured (Kennan 1999). Nonetheless, the poor relationships suggest that these metrics will contribute variance to indices which will reduce overall correlation with disturbance and with other metrics.

Differential response of taxa

Different taxa may respond differently to different types, levels, or scales of disturbance. For example, indices based on diatom communities have been shown to respond more directly to organic pollutants than protazoans or benthic macroinvertebrates and thus yield more accurate predictions of water quality (Stewart et al. 1985, Lowe and Pan 1996). Similarly, macroinvertebrates have been shown to be more sensitive than fish to organic pollution (Paller 2001). Differential responses to stress are partially due to physiology and physical capabilities of different groups. Benthic algae are sessile and cannot avoid potential pollutants through migration or other means so often show rapid responses to disturbance (Lowe and Pan 1996). Additionally, benthic algae have relatively short life cycles with cells of some taxa dividing more than twice daily (Eppley 1977). As a result, although these communities are often the first to respond to stresses, benthic algae are also typically the first group to recover. Although possessing greater motility than algae, benthic macroinvertebrate have similar vulnerabilities to physical changes in the benthos. For example, benthic macroinvertebrates have been shown to have a greater sensitivity to sedimentation than fish (Berkman et al. 1986). Because of their motility, fish can often seek refuge for certain stressors and thus show different responses to disturbance.

In this study, response to stress (as urbanization) differed among taxonomic groups. Catch per unit effort as a function of stream size and urbanization (section 3.3.1) shows an increase which affects changes in total species richness, richness of benthic invertivores, and % insectivorous cyprinids with urbanization of small streams. The increase of generalist fish in small coldwater or headwater streams (Steedman 1988, Scott and Helfman 2001) will generate different patterns of fish metrics and metric relationships for other groups in these streams. Macroinvertebrate metrics showed varying relationships with urbanization. Only the first (Mpca1) and fourth (Mpca4) PC showed relationships with urbanization. Mpca2 and Mpca3 were related to %wet. Similarly, as urbanization increased, APCA1, predominantly a siltation index axis, increased as well. However, no other algal PCs responded to urbanization.

Bryce and Hughes (2003, see above), showed different responses of fish, macroinvertebrate, and algae to stress (mining and agriculture). There was very low correlation among diatom and fish responses to mining, which could reflect different effects of acid mine drainage, sedimentation, and other mining impacts on the two groups. While urbanization has major impacts on different taxa through effects on hydrology, geomorphology and water quality, the taxa are likely to respond to different aspects of these effects. For example, water chemistry, shading, and scouring flows might affect diatoms; sedimentation and riffle degradation will likely impact macroinvertebrates; and base flow, riffle degradation, and habitat quality with have greater effects on fishes.

Griffith, et al. (2005) also looked at differential response to mining and agricultural gradients. They found differences in response to these two gradients and to other site differences (e.g., stream size and slope). The first two principal axes of fish metrics reflected two aspects of the mining gradient (associated with physical and riparian effects and sediment versus dissolved metal concentrations), while the third reflected stream size and slope. The first macroinvertebrate axis was correlated with aspects of the mining gradient (somewhat similar to those associated with the first fish axis, while the second macroinvertebrate axis was associated with stream size and agricultural gradients, and the third reflected aspects of the mining gradient not fully captured by the first axis. The first periphyton axis was associated with sediment coarseness, the second was associated with agricultural disturbance, and the third was related to riparian condition and water chemistry.

Van Sickle and Whittier (2005) looked at correspondence between levels of various stressors and assemblage data to develop models of relative risk of biotic impairment from single stressors. They analyzed such models for fish, macroinvertebrate and algal indices for the mid-Atlantic Highlands region. They found differences in response of different groups to different stressors. For example, fish impairment was related to habitat impairment, macroinvertebrate impairment was related to sedimentation, and algal impairment was related to nutrients. Some stressors, such as acidification, affected multiple groups.

The analyses of relationships among metrics from different indices provides information on similarity and differences of response of taxa to different stressors. For example, both the algal taxa richness and the macroinvertebrate taxa richness metrics demonstrate responses to water quality. However, the type and scale of response by each group may not necessarily be similar (see Section C, below) and the exact stressor is unknown. Because specific stressor data was not available, we relied on the literature for metric development and predicted response. Most of the metrics analyzed in this study are identical or closely analogous with metrics which have been used in other studies and there is a large literature showing that these metrics are correlated with disturbance, especially land use (e.g., Roth, et al. 1996, Kemp and Spotila 1997, Wang et al. 2000, 2001, McCormick, et al. 2001, Paul and Meyer 2001, Kennen and Ayers 2003, Wang and Lyons 2003).

Most of the statistical analyses in this study used principal components of assemblage structure. This avoids many of the problems in using composite indices which may be affected by uninformative metrics and also avoids problems with interpreting large numbers of correlations. The principal components reflected joint variation among metrics. Metrics which respond differently tend to be correlated with different components. “Uninformative” metrics would be expected to be poorly correlated with other metrics, so their variation would be reflected in different principal components. This pattern was seen for the analyses, where the first several principal components showed high correlations with a number of metrics, and lower components showed relationships with progressively fewer metrics. Some of these lower components were highly correlated with a single metric. The highly significant inter-correlations among the first few principal components of different taxonomic groups indicates that this procedure isolates such uninformative metrics. However, the high variability of these relationships shows that there are significant other sources of variation.

It is difficult to make any strong conclusions of algal- and algal-macroinvertebrate because only 7 metrics were available for algal analyses. Four of the metrics were diversity measures (Diatom Species Richness, %Dominants,% Dominant Taxa, and Shannon-Weiner Diversity Index) while the remaining 3 metrics (Siltation Index, Centrales:Pennates, and Percent of *Achnanthidium* *minutissimum*) were autecological measures. Generally, because periphyton are directly affected by physical and chemical environmental changes, and usually have brief life cycles and rapid rates of reproduction, they are valuable indicators of short-term impacts. Periphyton assemblages are also sensitive to some pollutants which may not visibly affect other taxa or only show affects at higher concentrations. Perhaps most importantly, periphyton has specific ecological requirements that correlate strongly with environmental conditions (Pan et al. 1996, Stevenson and Pan 1999, Pan et al. 2000, Potopova and Charles 2002). However, the data available for autecological analyses were unavailable. Metric development of this group and further analyses is needed to elucidate the variation and usefulness of algae metrics and between-group relationships.

Regressions of the fish principal components indicated significant relationships with macroinvertebrate principal components, watershed area, land use, and fish habitat scores. For some of these, the relationships with the macroinvertebrate principal components were better than those with land use, indicating that the Fpca-Mpca relationships were not just driven by joint relationships with land use.

Calibration of metrics involves adjusting metrics for covariation with factors other than disturbance (e.g., stream size as measured by watershed area) and setting cutpoints between metric scores. Inaccuracies in the adjustment may result in incomplete adjustment, so that the adjusted metric still varies with the covariate. The analyses of the fish metrics for the NJ FIBI data suggests that the regression of richness metrics on watershed size does not completely remove the variation, and that there is watershed-size related variation in other metrics (such as %Insectivorous cyprinids), which is not adjusted. Since macroinvertebrate metrics did not show the same degree of watershed-size dependence, the adjustment issues affect inter-correlation. Furthermore, the rating score regressions for the fish species richness metrics resulted in high scores for a high proportion of small streams, while larger streams more often received lower scores. This may reflect use of a rating regression which is too low for small streams. However, it may also reflect increases in species richness of small streams with disturbance (at least up to a point).

Implications for Integrated Bioassessment

Since all of the taxonomic groups are relevant to ecological condition, indices from all groups could potentially be used in bioassessment. Since different responses of different groups are possible, there are both technical (statistical and biological) and policy issues in such integration. Technical issues are involved in determining accuracy of each assessment. Policy issues arise in defining a single impairment status from the multiple assessments. For example, apparent impairment of two taxa could be considered significant from both the technical view as providing greater confidence in the accuracy of the determination and from the policy view as indicating more complete impairment of the site.

From the policy view, the number and severity of impairments could be used to define a single ranking. For example, if assessments are assumed to be accurate, impairment of any taxon could be considered as impairment of the reach. Severity of impairment and the range of groups affected could affect the ranking of impairment. For example, moderate impairment of one taxonomic group and no impairment of the other two could be considered as a low level of impairment, while moderate impairment of two groups could be considered moderate impairment. This type of ranking system becomes more difficult considering the full range of assessment outcomes, e.g., comparing impairment at a site with moderate impairment of all three taxa with that at a site with severe impairment of one taxon and no impairment of the other two. However, the analyses of correlation among metrics suggest that some intercorrelation is expected, so that cases with very different assessments of different taxa may be uncommon. Consideration of error in assessments will further complicate integrated ranking of sites.

Several possible approaches to integrating use of different taxonomic groups are possible. The best approach will depend on the needs of the bioassessment program as well as on the statistical and ecological properties of the indices. It is useful to consider assessment as testing hypothesis about impairment. At the simplest level, the null hypothesis is that a taxon at a site is unimpaired and the test hypothesis is that the taxon at that site is impaired (more complex tests would involve levels of impairment). As in any statistical testing procedure, any approach will have to balance type I and type II statistical errors, i.e., rejecting the null hypothesis when it is, in fact, true (type I error) and accepting the null hypothesis when it is false (type II error). For a given distribution of data, tests typically involve a tradeoff between the probabilities of these errors, e.g.., decreasing the probability level for rejecting the null hypothesis (decreasing type I error) will increase type II error. Both error probabilities can be decreased by increasing the accuracy and precision of the data. Analogously, use of multiple taxonomic indices needs to be considered with respect to these errors. A procedure which makes it easier to reject the null hypothesis will increase the probability of false positives. Conversely, a more stringent test will result in fewer false positives but more incorrect assessment of no impairment. However, improvement in the indices will reduce both false negatives and false positives. The bases for determining how to balance these error types depends on factors related to relative consequences and costs (in the broad sense) of different errors.

As noted above, increasing accuracy and precision of indices will decrease probabilities of both types of statistical errors. Thus, evaluation of indices is an important part of defining protocols for integrated assessment. In this study, analyses of individual metrics and scoring systems suggest several areas of possible improvement in indices. These include:

A. Down-weighting or eliminating metrics which are less closely linked to impairment, either because of variability in estimation of these metrics or variable response of metrics to disturbance. For example, in the FIBI, % white sucker was not clearly related to impairment, possibly because of estimation and response issues. The salmonid-centrarchid metric may show weak response to disturbance because the metric is a composite of salmonid and centrarchid assemblages, which probably respond differently to disturbance. Among macroinvertebrate metrics, % Dominant Family was not clearly related to impairment. Like % White sucker, the apparent response of this metric may reflect both estimation and response issues.

B. Improved adjustment for relationships between metrics and covariates unrelated to impairment. In particular, several fish metrics are adjusted for stream size (by using different scoring thresholds for watersheds of different size). Analyses suggested that the adjustment was incomplete for some metrics, and that adjustment should be considered for some metrics (e.g., % insectivorous cyprinids). In addition, stream gradient may have an important effect on fish assemblages, so that measurement and adjustment for gradient may improve metrics.

C. Inclusion of additional metrics.

D. Refinement in definition of metrics and assignment of taxa to different classes, based on more precise understanding of taxonomic responses to impairment. For example, analyses of the FIBI scores for different streams suggested that occurrence of sensitive large stream and river species was not well-reflected in scoring, resulting in overestimates of impairment for some rivers. In constrast, the occurrence of small numbers of hatchery-derived salmonids in some streams may lead to underestimation of impairment in some streams. For the macroinvertebrate metrics, use of greater taxonomic resolution may improve accuracy and precision of metrics.

F. Increased sampling or laboratory effort. Since this would greatly increase processing time and costs, this may be difficult to implement.

In discussions to follow, we assume that some improvements have been evaluated and made. Thus, these scenarios do not necessarily apply to the existing data set of ratings. To some extent, comments about integrated assessment depend on assumptions about eventual precision of indices/metrics.

Approaches to using multiple indices include:

1 Joint use of multiple indices, e.g., by a ranking scheme as discussed above. The definition of the ranking scheme would determine the relative error rates. At one extreme, impairment by most or all indices would be necessary to assign impairment. This will be the most stringent criterion, with fewest sites judged as impaired. Given high levels of variability among indices demonstrated in this and other studies, differences in ratings among indices are expected to be common. Several studies have found that variability is highest at higher levels of impairment. Sampling error may also be relatively large at moderately impaired sites, because taxa which influence indices may be uncommon at these sites and subject to greater sampling variation. As a result, this approach is expected to greatly underestimate impairment. At the other extreme, impairment by few indices would be required to assign impairment. It may be argued that loss of integrity of any system component is evidence of impairment, so impairment of any index is sufficient to demonstrate impairment. As argued above, the analyses of multiple indices in this study and elsewhere indicate that differences in indices at any single site will be common. Since some of the differences reflect sampling variance, this approach will lead to overestimation of impairment.

2) Use of average values of indices to determine overall rating. This approach is intermediate to the first two and would minimize effects of extreme values of individual metrics which are unrelated to impairment (e.g., due to sampling error, etc.). More complicated averaging rules may also be considered. Indices from different taxa might be weighted differently, e.g., to put more weight on indices which are less variable. In practice, it is likely that this approach would end up being similar to the discrete integration protocol discussed above. However, extreme values in a single index would probably have less weight in determining impairment than in the first alternative. Relative error rates would depend on the exact averaging, weighting and scoring system used.

3) Sequential or phased use of metrics. Primary assignments could be based on a single index, and other indices could be used mainly in cases where the first index is ambiguous or to help assign causes of impairment. For example, the macroinvertebrate index may be selected, based on the relatively clear relationship to development. This protocol would be the most consistent with current practice, which bases impairment solely on the AMNET. This approach would be less sensitive to certain types of stressors which have relatively little effect on macroinvertebrates.

A variant of this approach could take into account typical triangular shape of stressor-response plots, i.e., a high variance of scores is seen for sites with low apparent stress, with a decrease in mean and variance with increasing stress. Thus, there is more confidence in a high score as evidence of a lack of impairment than in a low score indicating impairment An initial assessment could be based on one index (e.g., AMNET). If this shows a high value the site would be rated as unimpaired. If the primary index shows a low score, the other indices would be used.

4) Use of professional judgement to weigh evidence from the various indices. In addition to basic index values, evidence could include the values of individual metrics of the indices, raw species data, data on site characteristics (e.g., habitat), watershed characteristics (e.g., land use), point sources, or unusual conditions. Site visits might be determined to indicate unusual conditions or presence of unknown stressors. Additional sampling or lab analyses of existing samples of provisional sites could be done, either sampling all taxa or only the discordant taxa. For example, if macroinvertebrate indices are discrepant with other indices, larger sample counts and calculation of auxiliary metrics could provide more precise information on macroinvertebrate assemblages. Similarities and differences among indices could be examined with respect to metrics within the indices which are expected to respond similarly to the same stressors (e.g., intolerant fish metrics and several of the AMNET metrics). Unusual habitat conditions, land uses, or point sources which might affect single indices could be taken into account. Consistency of index and metric values with known stressors could also be used. This would provide the most thorough use of the biological data and would provide the greatest opportunity for resolving apparent discrepancies. However, this approach may be time-consuming. In practice, it may also be cumbersome to coordinate assessments by experts in each of the taxa. It may also be more difficult to defend, and other parties could devise alternative explanations and rankings for the sites.

5) Use of provisional or special categories based on lack of agreement among indices, with a decision system for resolving questionable sites. For example, if two indices showed similar ratings and the third index was similar or the same, that rating would be assigned to the site. With lack of such agreement, the site would be placed in a provisional category, indicating that more analysis or information would be needed to rate the site. In these cases, the site may be unrated until further analysis is done or a provisional rating may be given, depending on regulatory needs. A series of steps would be defined to resolve the site. These steps might involve examination of site characteristics (e.g., habitat) or watershed characteristics (e.g., land use). Site visits might be determined to indicate unusual conditions or presence of unknown stressors. Additional sampling or lab analyses of existing samples of provisional sites could be done, either sampling all taxa or only the discordant taxa. This approach would be difficult to implement, because it would be difficult to formulate the variety of situations which would be encountered.

6) Use of provisional or special categories based on lack of agreement among indices, with professional judgement used to rate sites in provisional categories. This is a hybrid between the previous two approaches. It has the advantage of an up-front protocol which would provide a rating for a number of sites, with a flexible approach to resolving discordant evidence. As in the previous approaches, additional site visits, sampling or lab analysis might be used to resolve questionable sites. A hypothetical example of this approach is shown in Table 4.1.3, in which overall ratings are derived from macroinvertebrate, fish and algal metrics. An overall rating score is developed based on the individual indices where two or three of the indices produce similar ratings, and no index

differs substantially from the others. Where these conditions are not met, a site-by-site analysis is done. While a number of metric combinations might require this provisional rating, some of these combinations are expected to be relatively rare (see frequency column in Table 4.1.3). The case-by-case analysis might conclude that the discordant index results from a sampling artifact or some natural factor which specifically affects that taxon. In that case, the results of the other indices would be the basis of the overall rating. Alternatively, the analysis might conclude that the discordant index reflects taxon-specific disturbance. Table 4.1.3 lists some specific possibilities which might be considered in the case-by-case analysis. For example, a discordant fish rating might be due to poor habitat quality (fish index lower than others) or due to presence of unusual habitats or proximity to areas with high diversity (fish index higher than others). A discordant algal rating might reflect a algae-specific toxicant (e.g., herbicide), recent flushing flows, or low habitat quality which would affect both fish and macroinvertebrates but not algae. Cases in which the macroinvertebrate index is much higher than the other indices might reflect unusual conditions such as sedimentation. Cases with a much lower macroinvertebrate index are apt to be rare, since impairment of macroinvertebrates is likely to be reflected in fish or algal indices. Additional sampling and analyses may be warranted in this case to determine the robustness of the macroinvertebrate rating.

7) Formulation of new indices or metrics which use metrics from different taxa. The PCA indicated joint variation of different combinations of metrics, with indication of relationships between these combinations and individual stressors. The macroinvertebrate metrics used in AMNET were generally highly inter-correlated, with joint variation indicating a major impairment gradient. Other metrics contributed to additional variation, although the bases of these components were not clear. These could include responses to different types of stress (e.g., nutrient enrichment) or could reflect differences in streams related to presence of lakes, reservoirs or wetlands in the watershed or other factors. The fish metrics were designed to assess a variety of aspects of assemblage structure, so that individual metrics are not as highly correlated. Components of the fish metrics were correlated with components of the macroinvertebrate metrics. Thus, it may be possible to use related metrics from different taxa to assess different aspects of impairment. Use of principal components, as done in this report, would be difficult, but simpler subindices might be developed. In some cases, individual metrics will be useful; for example, fish metrics relating to richness of intolerant species correlated with the AMNET metrics. Other metrics show complex relationships with watershed and site characteristics, making them harder to relate to other indices. For example, proportion of insectivorous cyprinids shows complex interactions between watershed size, watershed development and presence of lakes and impoundments, so that correlations with macroinvertebrate metrics will be variable. In the PCAs, individual metrics were often correlated with several axes, i.e., they responded to variation in a variety of stressors. As a result, compound indices might use the same metrics in different indices, making assessments very sensitive to these metrics. Griffith, et al. (2005) suggested a compound index of this type, using a total of 12 fish, macroinvertebrate and periphyton metrics, using metrics which were found to have the clearest relationships to individual stressors. This approach is attractive in explicitly integrating data from different taxa to indicate specific stressors. However, given the complexity of the relationships, confidence in the calibration and rating of each metric would be essential in providing robust assessment.

The reliability of any of the procedures will ultimately depend on the accuracy and precision of the individual metrics. This study indicates several ways in which the individual indices may be made more reliable. The analyses of the NJ FIBI data indicated that recalibration of metrics may significantly improve reliability. In particular, better adjustment of metric values to account for watershed area is possible. In addition, the fish IBI does not adjust for inherent differences in stream morphology. Some aspects of morphology, such as stream gradient, are closely related to fish habitat and occurrence, so that assessment of metric-stream gradient relationships may improve metric performance. It may be worthwhile to re-evaluate metrics which are negatively correlated with other metrics (e.g., number of salmonid-centrarchid species) or uncorrelated with identified disturbance gradients (e.g., proportion white sucker). Most of the analyses in this study used 100-individual subsamples of the macroinvertebrate data, as does AMNET. Use of larger counts could increase precision, although the improvement would need to be evaluated with respect to increased processing costs. The AMNET metrics are based on relatively coarse taxonomic precision (family level), although identifications to much finer levels are routinely done. It is possible that use of finer resolution would also improve the performance of the macroinvertebrate metrics. For this study, relatively few algal metrics were available for analysis. It is likely that additional metrics may provide additional information. Improvement in metrics is expected to reduce frequency of conflicting information from different indices and enable use of observed differences to provide information on different aspects of impairment.

These analyses indicate that integrated assessment of multiple indices can provide better support for assessment and provide information on stressors not as well evaluated in existing procedures. A series of further steps will be important in providing the basis for integrated assessment. These include re-evaluation and calibration of fish metrics to provide better control for natural variation in stream size and stream morphology, increased resolution of macroinvertebrate analyses (e.g., higher taxonomic resolution and/or larger sample sizes) either for the primary metrics or as auxiliary metrics to help resolve discordant data, development of additional algal metrics, evaluation of intercorrelation among new data, test cases of integrated approaches on subsets of existing data, and development of efficient protocols for integration across taxa.

Table 4.1.3. Possible protocol for integrating macroinvertebrate (m), fish (f) and algal (a) indices, using a scoring system for sites with low discordance, and examination of data and possible additional analyses for sites with discordant ratings. Overall 1 and 2 give two alternative scoring systems for the overall rating, which differ in whether 1 or 2 indices are required to assign the more impaired rating. Freq is the expected frequency of different situations, assuming sampling sites with a range of conditions.

M	F	A	Agree	Freq	Overall		Specific Considerations
					1	2	
u	u	u	3	high	u	u	
u	u	m	2	mod	u	m	
u	u	s	2	low	prov	prov	examine algal data; look for evidence of nutrient, shading, herbicide, recent storms
u	m	u	2	mod	u	m	
u	m	m	2	mod	m	m	
u	m	s	0	low	prov	prov	examine all relevant data
u	s	u	2	low	prov	prov	examine fish data and habitat; resample fish
u	s	m	0	low	prov	prov	examine all relevant data
u	s	s	2	low	prov	prov	additional macroinvertebrate analyses
m	u	u	2	high	u	m	
m	u	m	2	high	m	m	
m	u	s	0	low	prov	prov	examine all relevant data
m	m	u	2	high	m	m	
m	m	m	3	high	m	s	
m	m	s	2	high	s	s	
m	s	u	0	low	prov	prov	examine all relevant data
m	s	m	2	mod	m	s	
m	s	s	2	low	s	s	
s	u	u	2	low	prov	prov	examine macroinvertebrate data; look for sedimentation, non-nutrient point source, etc.
s	u	m	0	low	prov	prov	examine all relevant data
s	u	s	2	low	prov	prov	look at fish data, stocking, proximity to large river
s	m	u	0	low	prov	prov	examine all relevant data
s	m	m	2	high	m	s	
s	m	s	2	high	s	s	
s	s	u	2	low	prov	prov	examine algal data; look at gradient; amount and quality of riffles
s	s	m	2	high	s	s	
s	s	s	3	high	s	s	

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APPENDIX A

Graphs of relationships between fish abundance ($e^{mean[ln(catch\ per\ 100\ m)+1]}$-1), watershed size classes and urbanization classes (see Section 3.1.1).

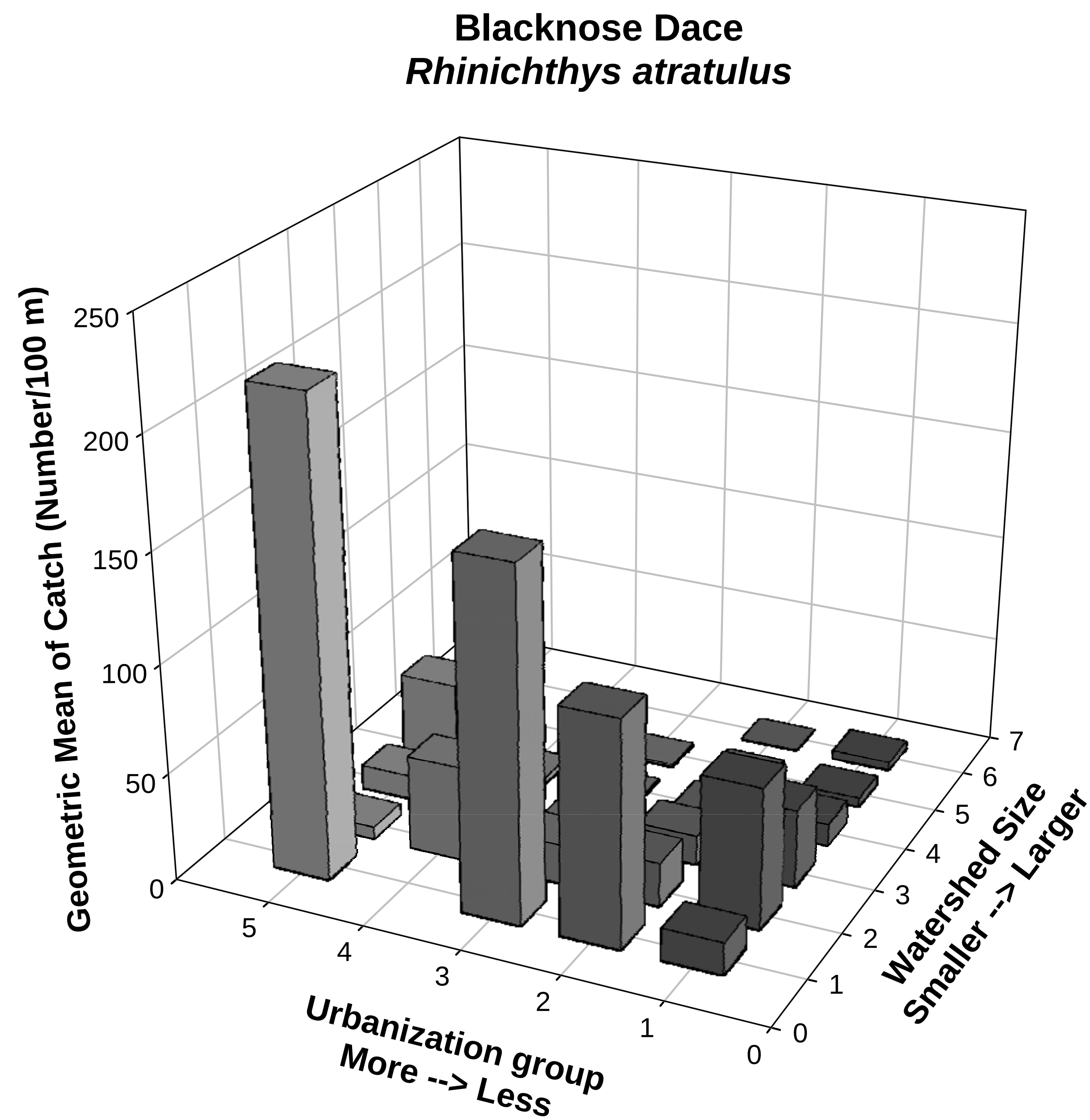

Figure A.1.1. Abundance-watershed size-urbanization relationship for blacknose dace.

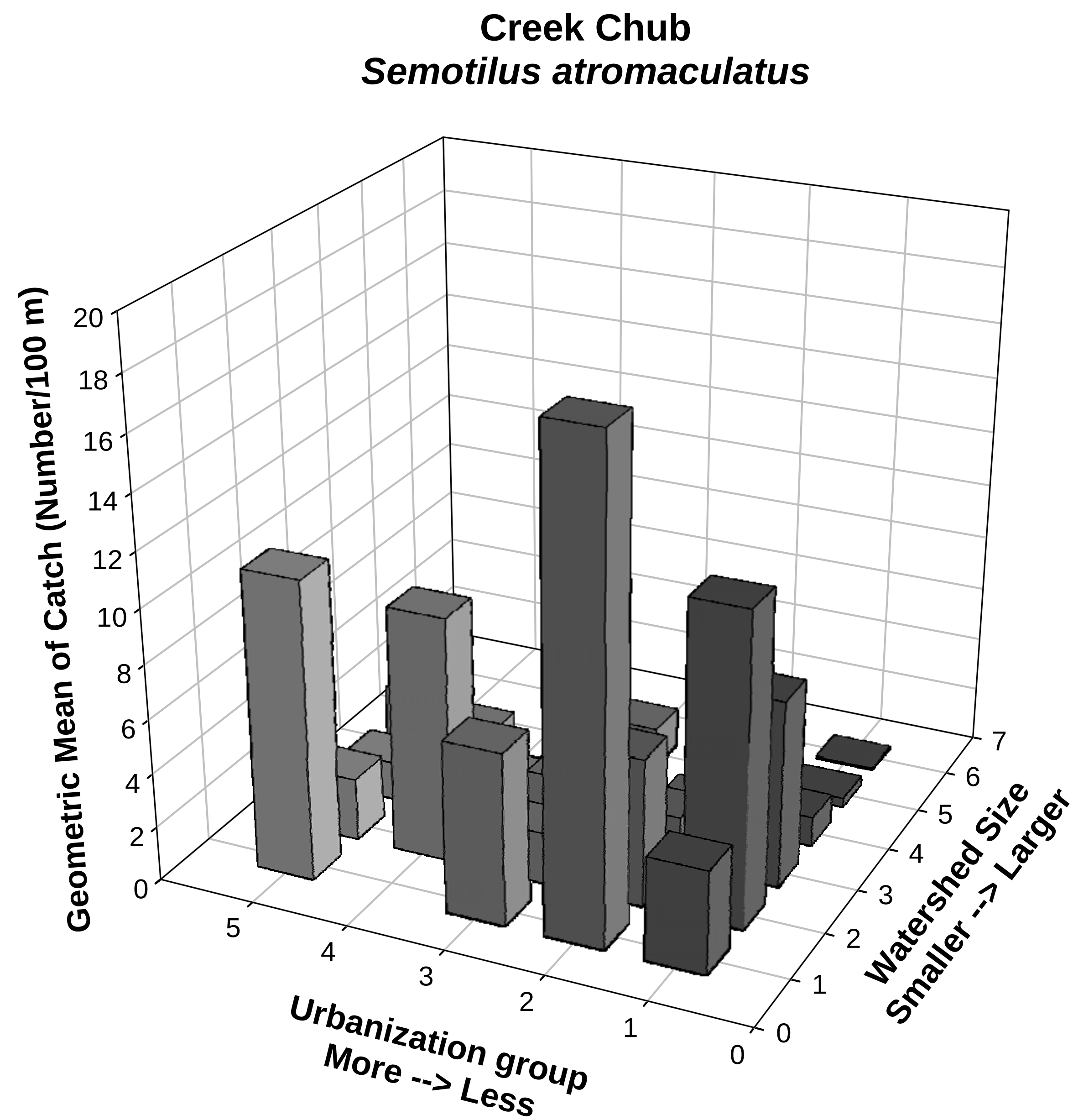

Figure A.1.2. Abundance-watershed size-urbanization relationship for creek chub.

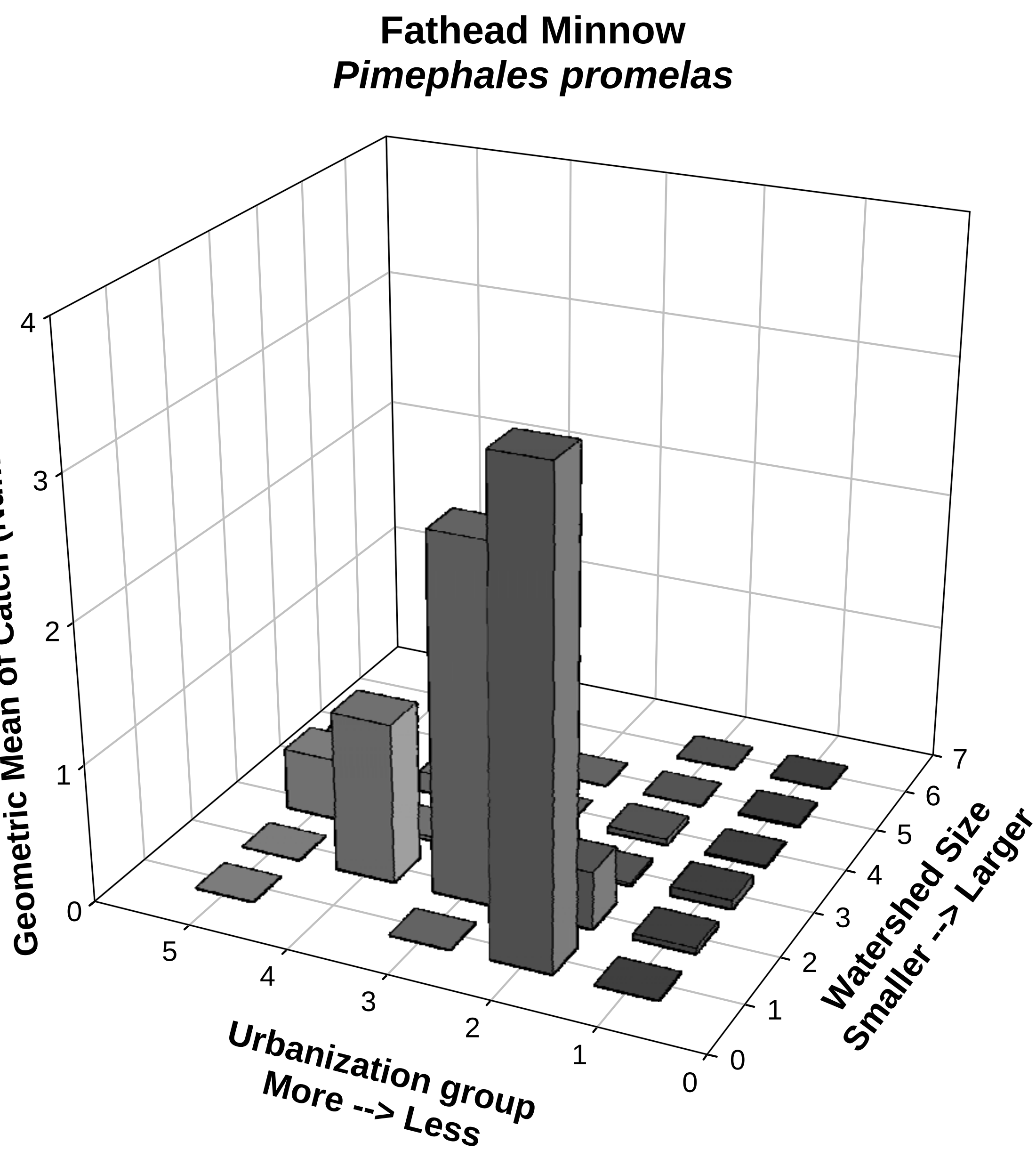

Figure A.1.3. Abundance-watershed size-urbanization relationship for fathead minnow.

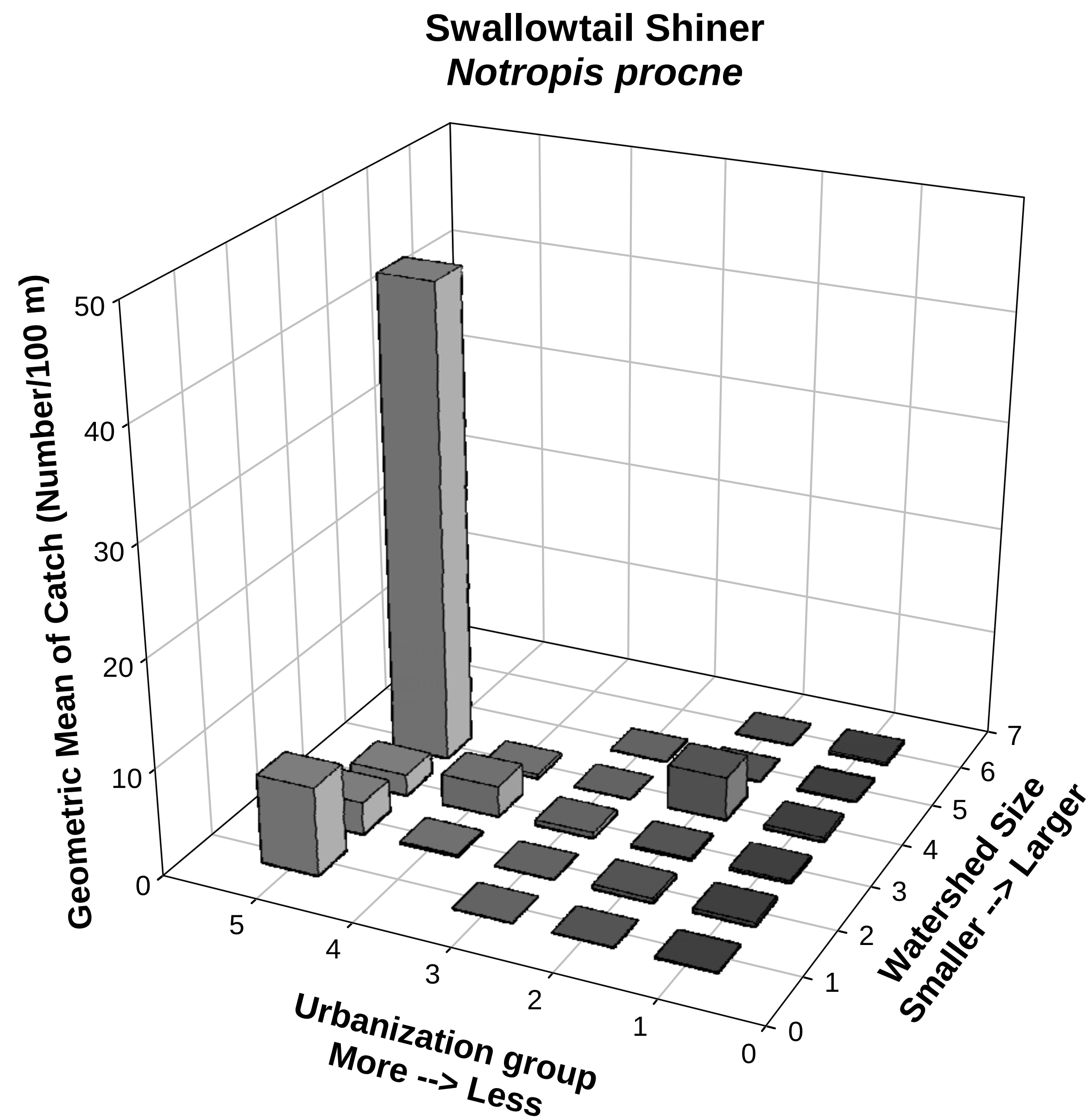

Figure A.1.4. Abundance-watershed size-urbanization relationship for swallowtail shiner.

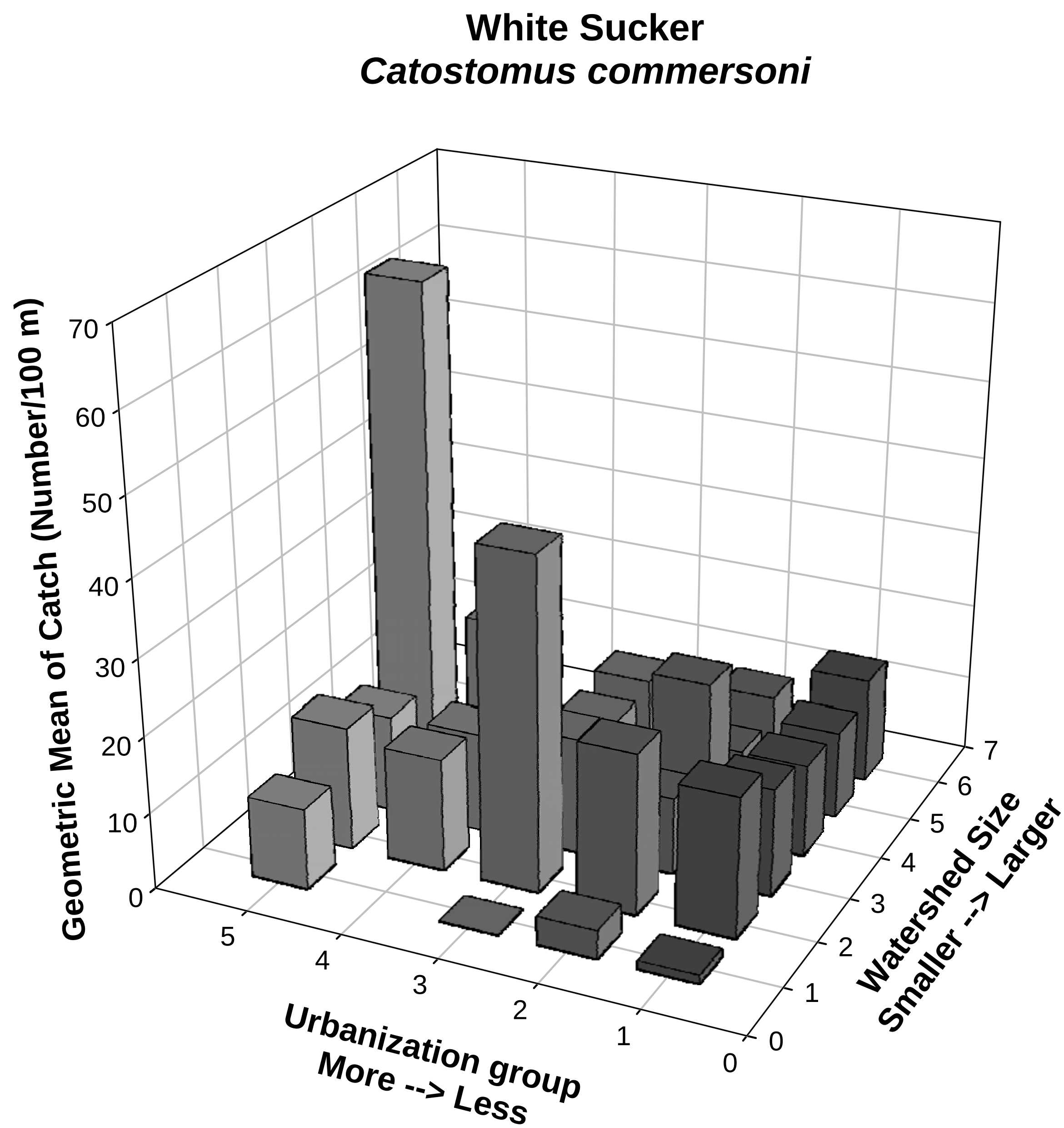

Figure A.1.5. Abundance-watershed size-urbanization relationship for white sucker.

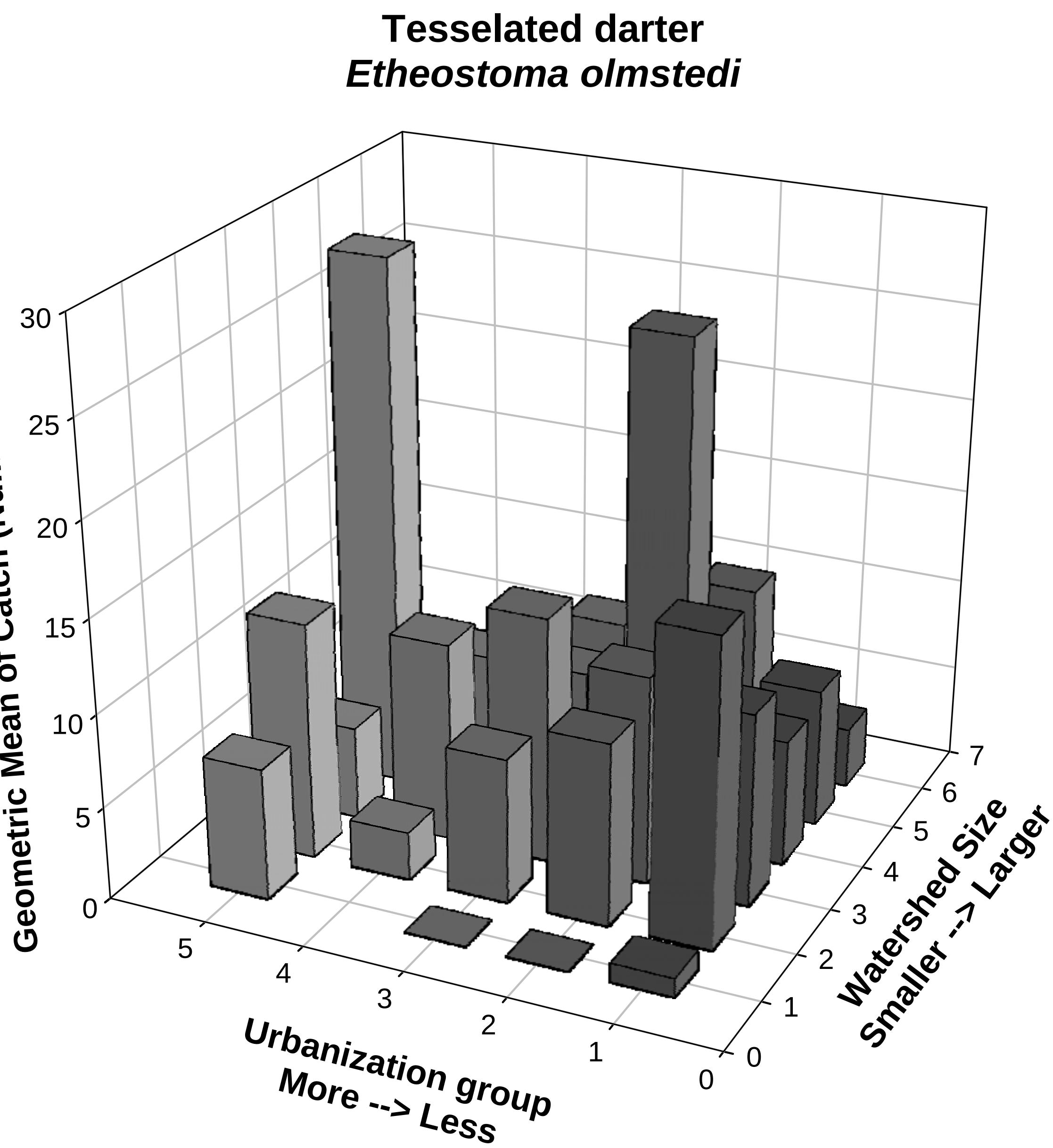

Figure A.1.6. Abundance-watershed size-urbanization relationship for tesselated darter.

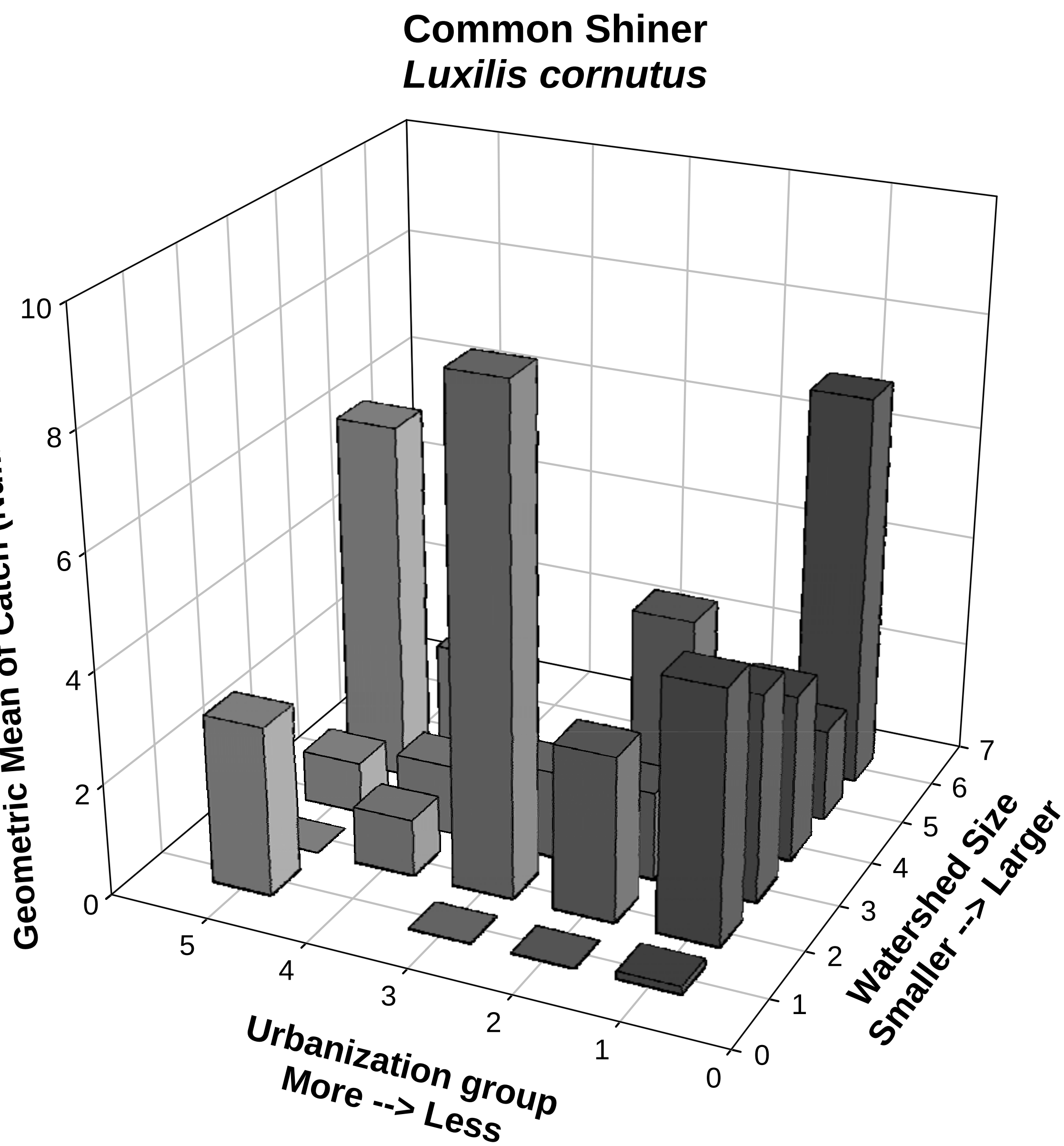

Figure A.1.7. Abundance-watershed size-urbanization relationship for common shiner.

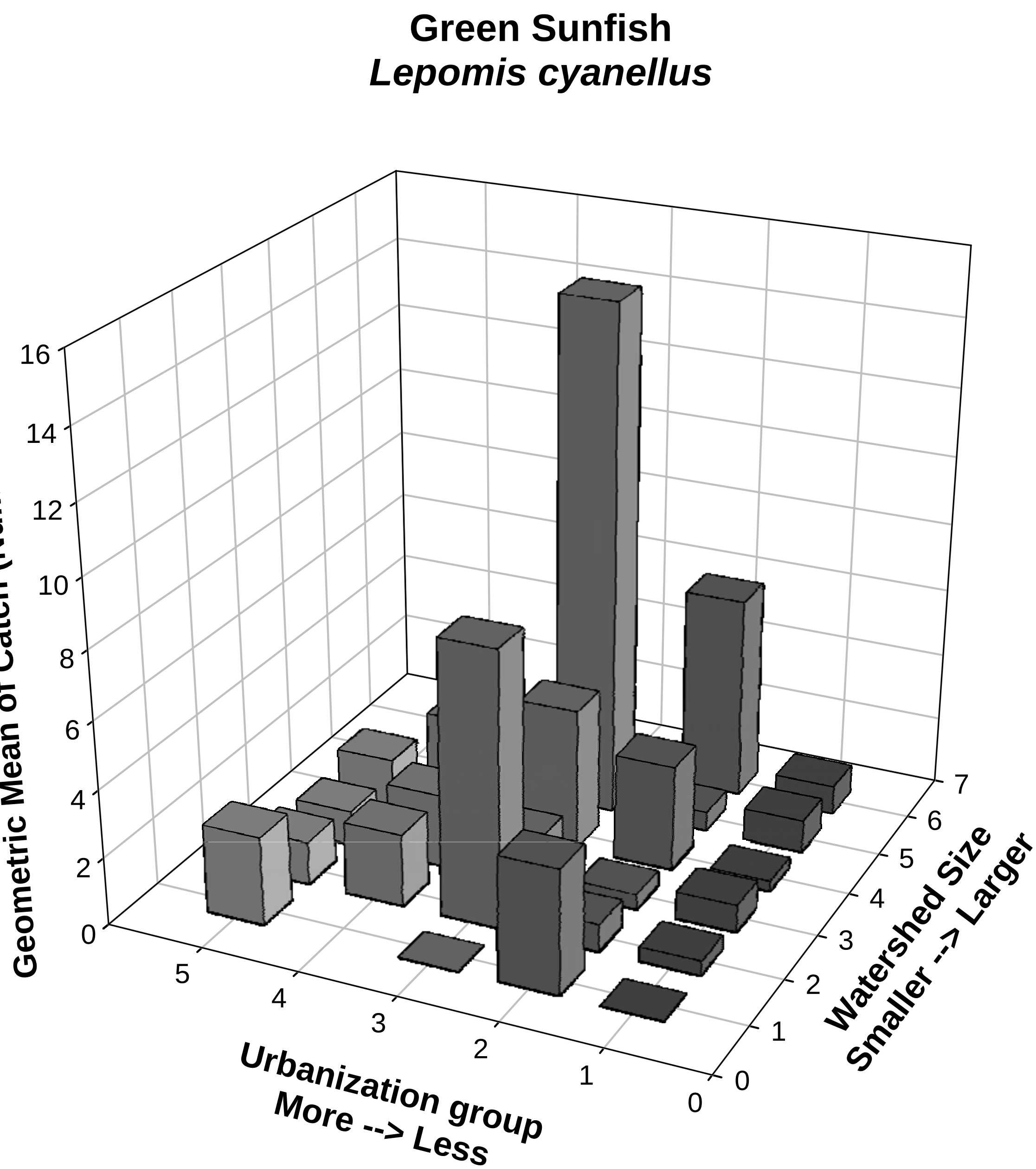

Figure A.1.8. Abundance-watershed size-urbanization relationship for green sunfish.

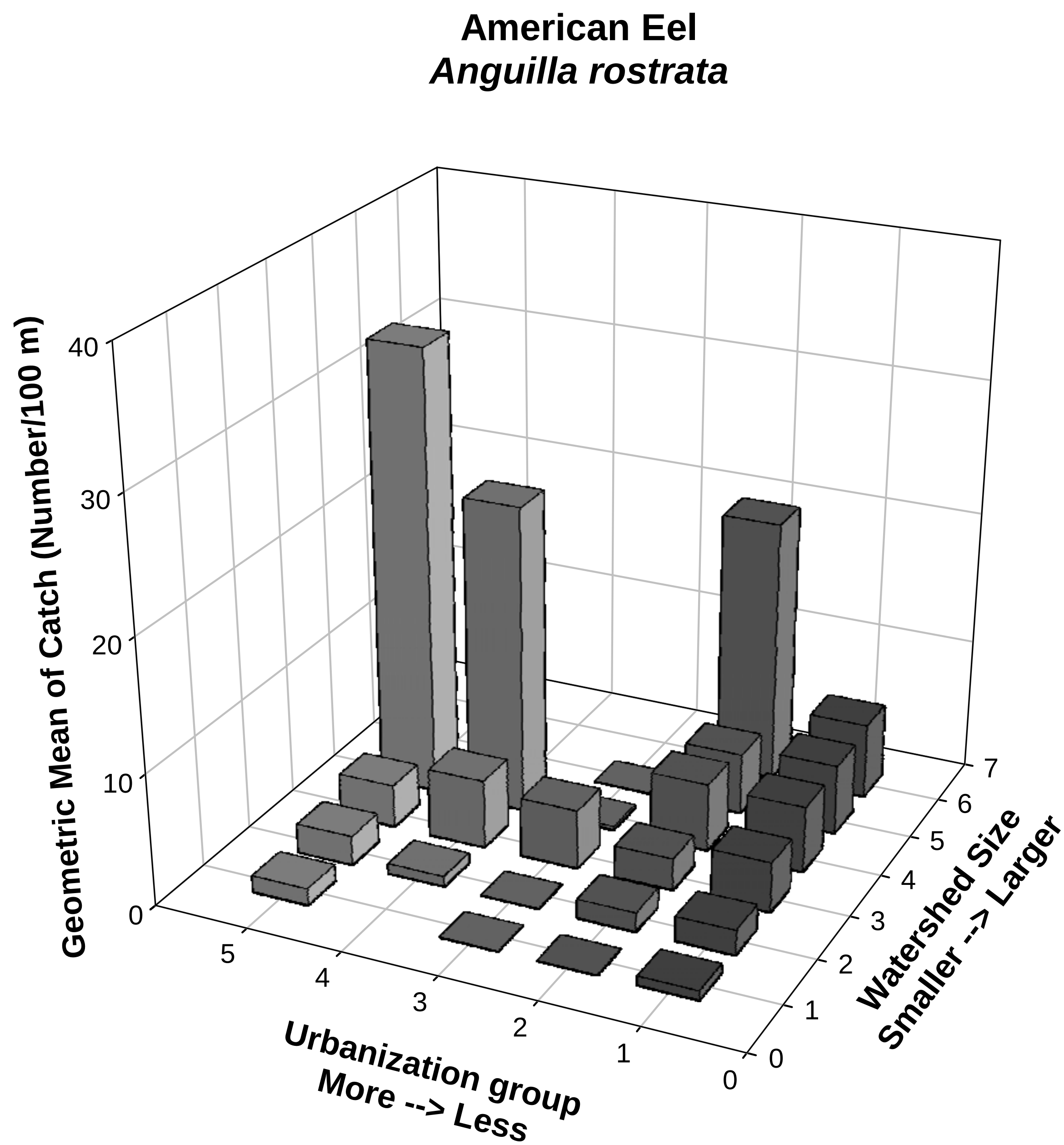

Figure A.1.9. Abundance-watershed size-urbanization relationship for American eel.

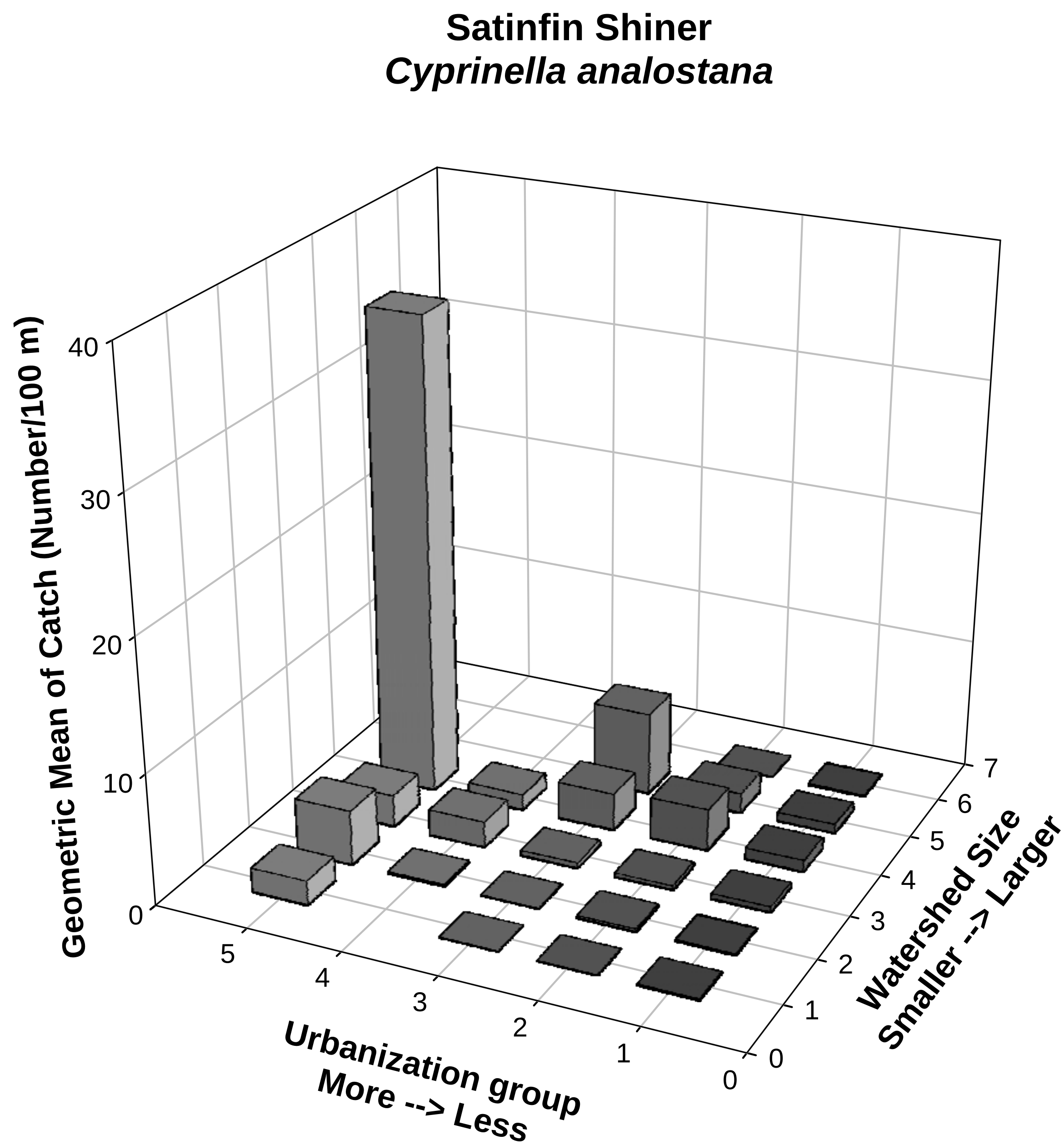

Figure A.1.10. Abundance-watershed size-urbanization relationship for satinfin shiner.

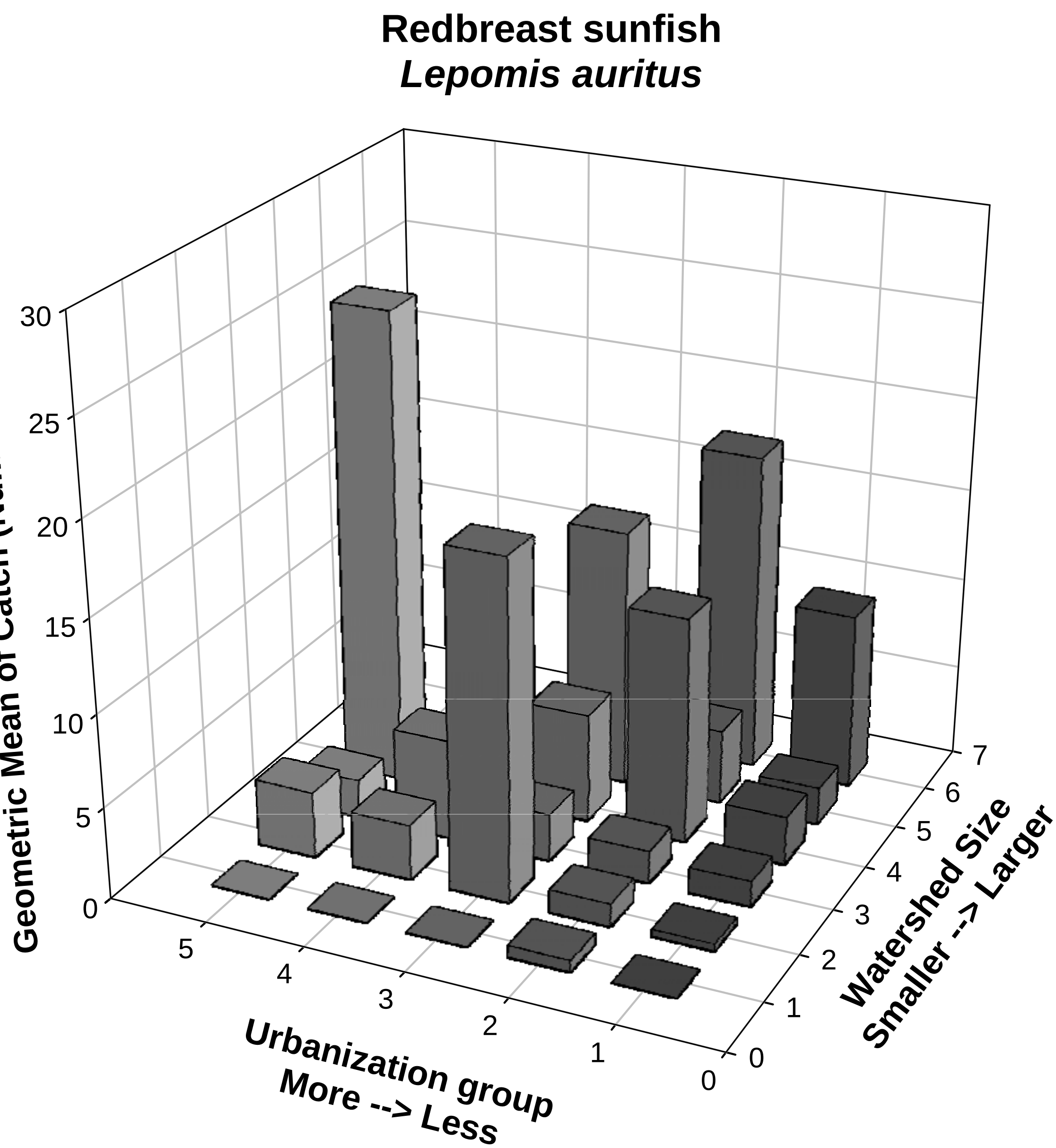

Figure A.1.11. Abundance-watershed size-urbanization relationship for redbreast sunfish.

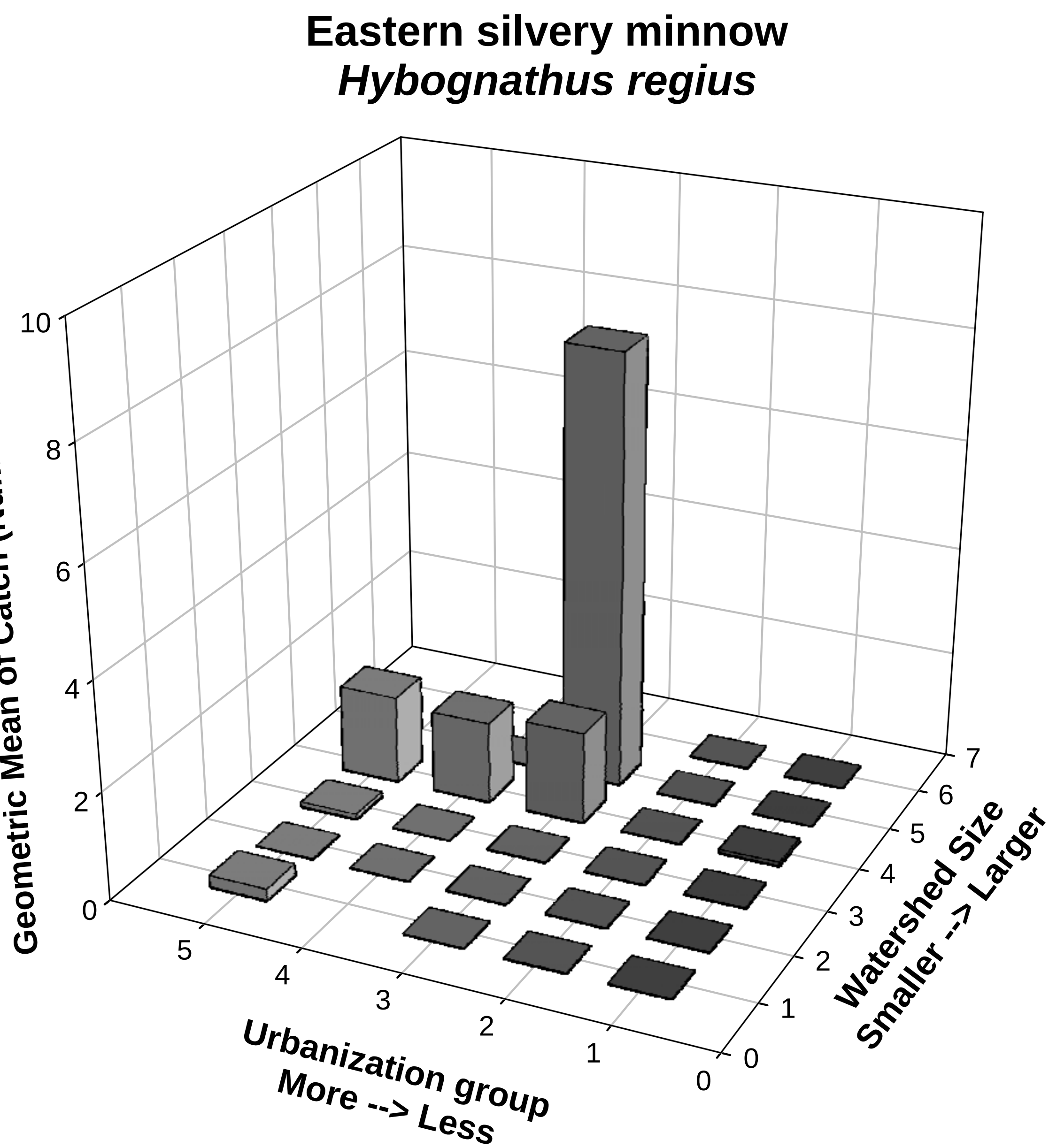

Figure A.1.12. Abundance-watershed size-urbanization relationship for eastern silvery minnow.

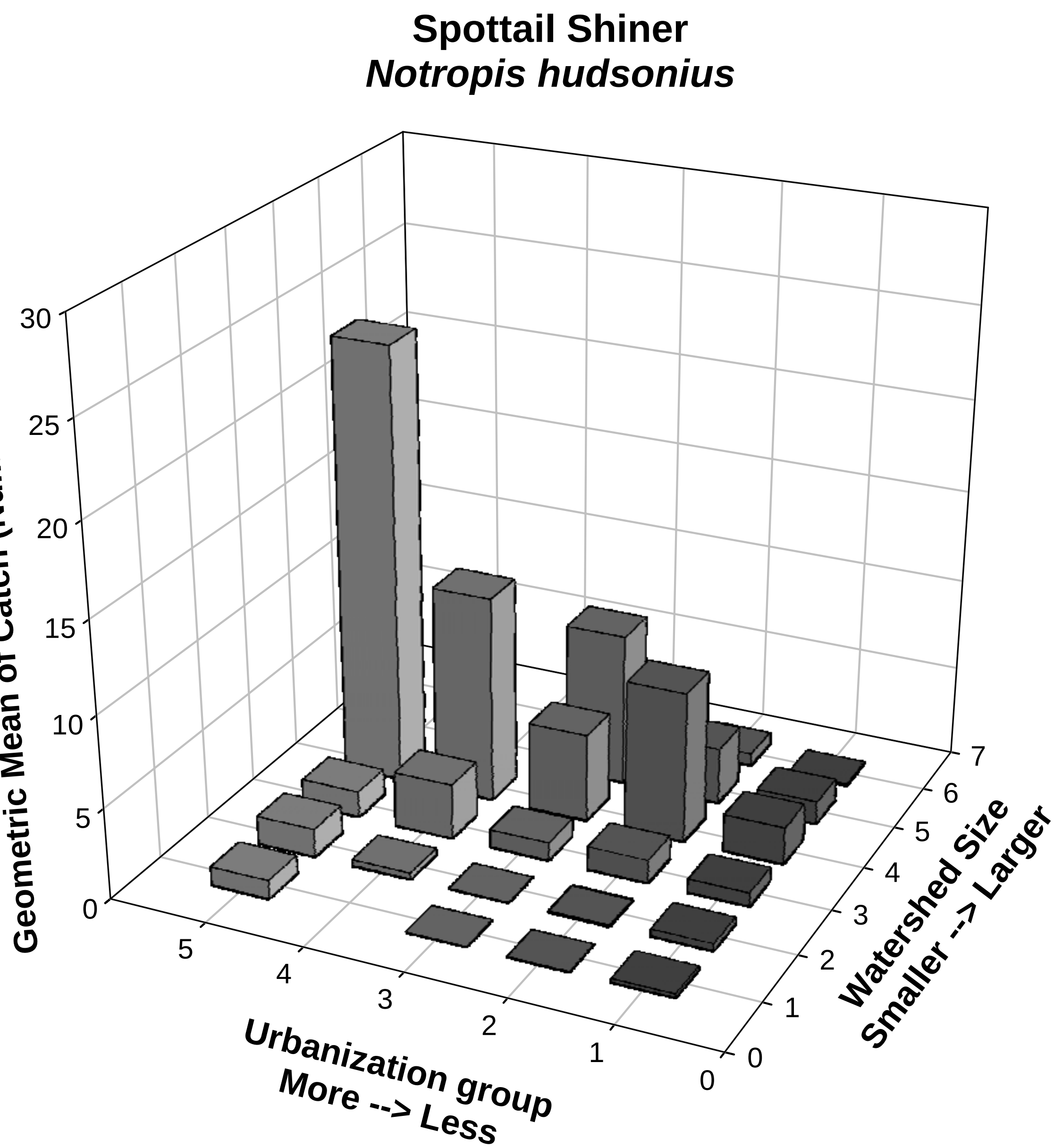

Figure A.1.13. Abundance-watershed size-urbanization relationship for spottail shiner.

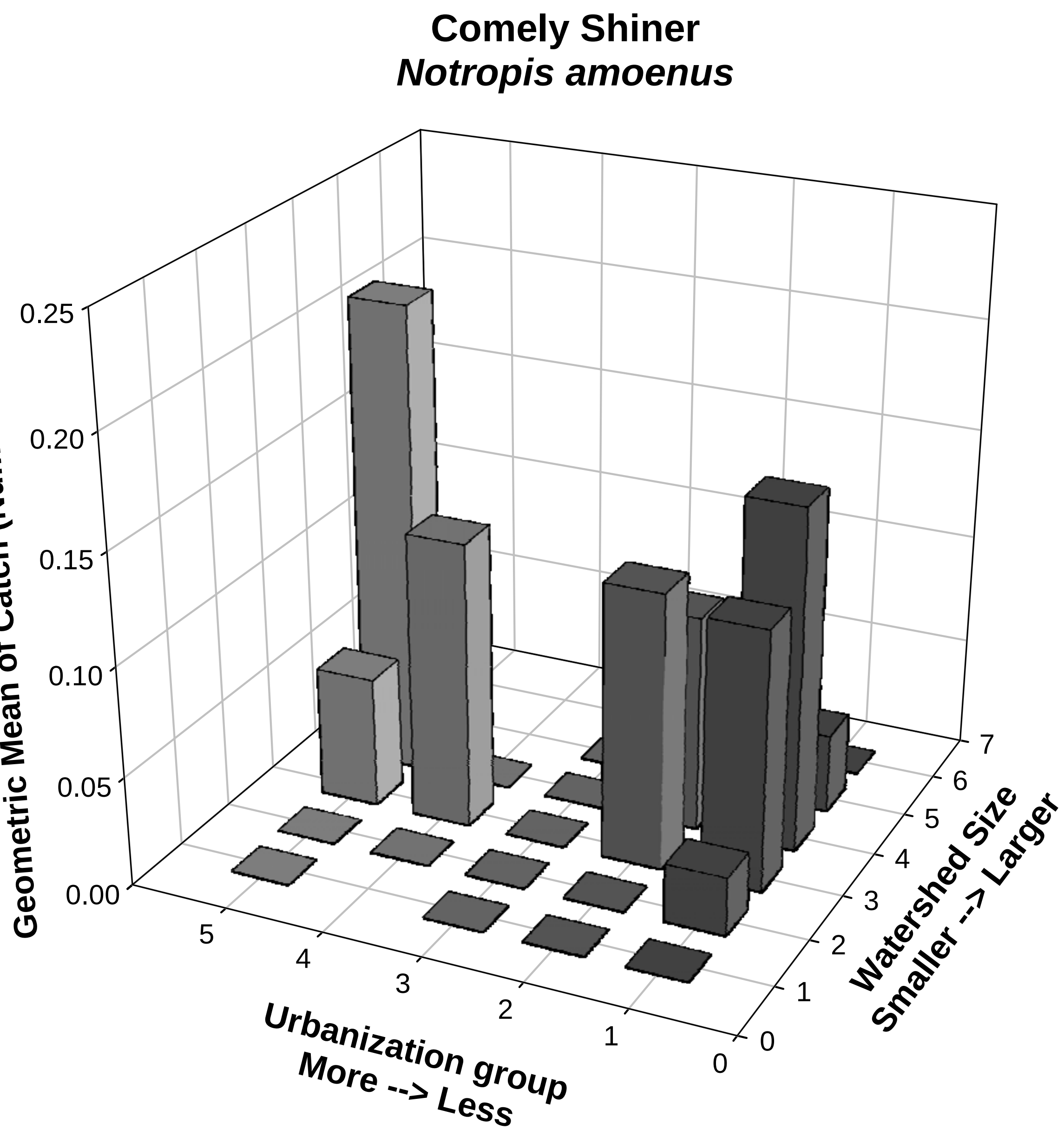

Figure A.1.14. Abundance-watershed size-urbanization relationship for comely shiner.

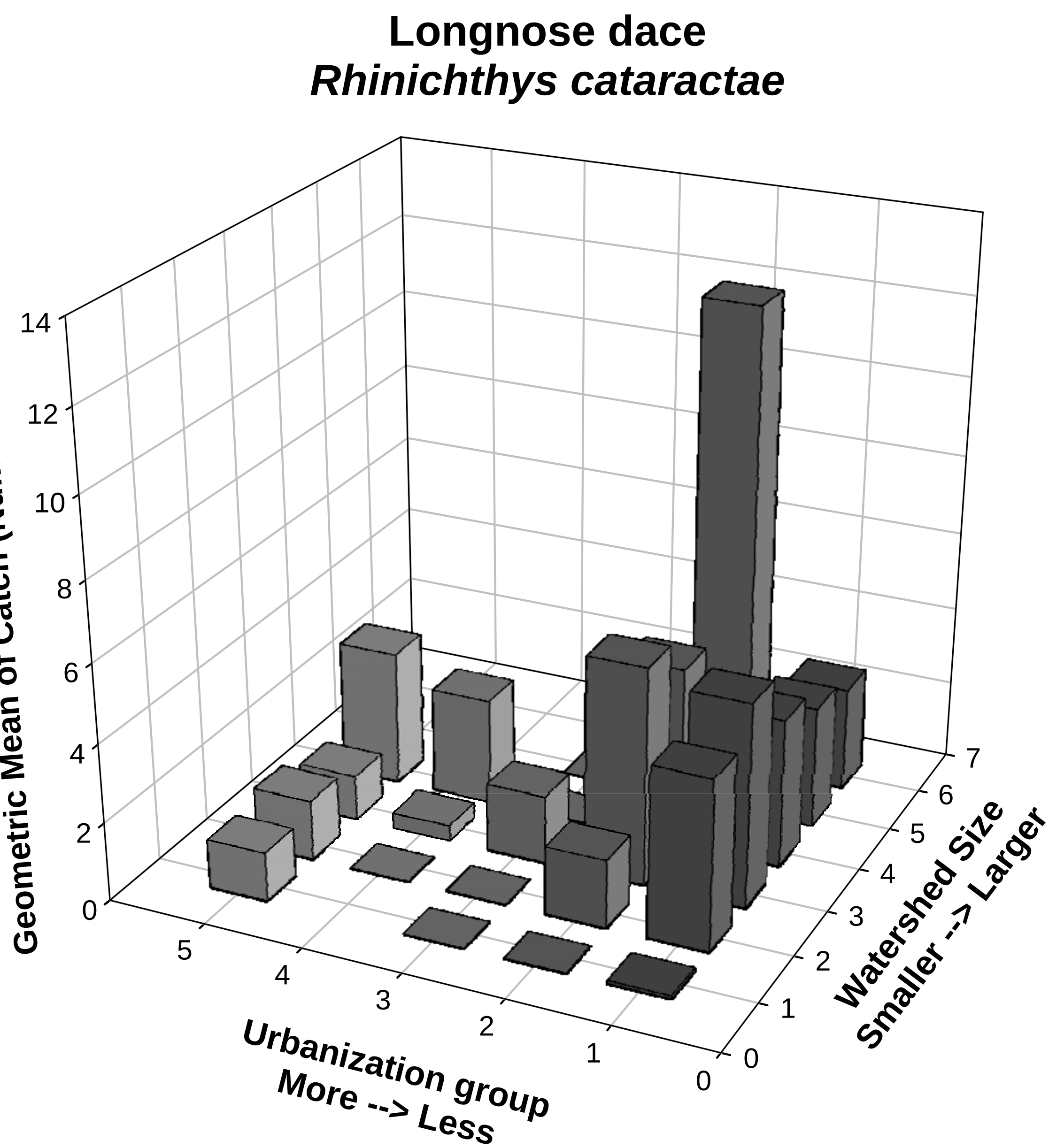

Figure A.1.15. Abundance-watershed size-urbanization relationship for longnose dace.

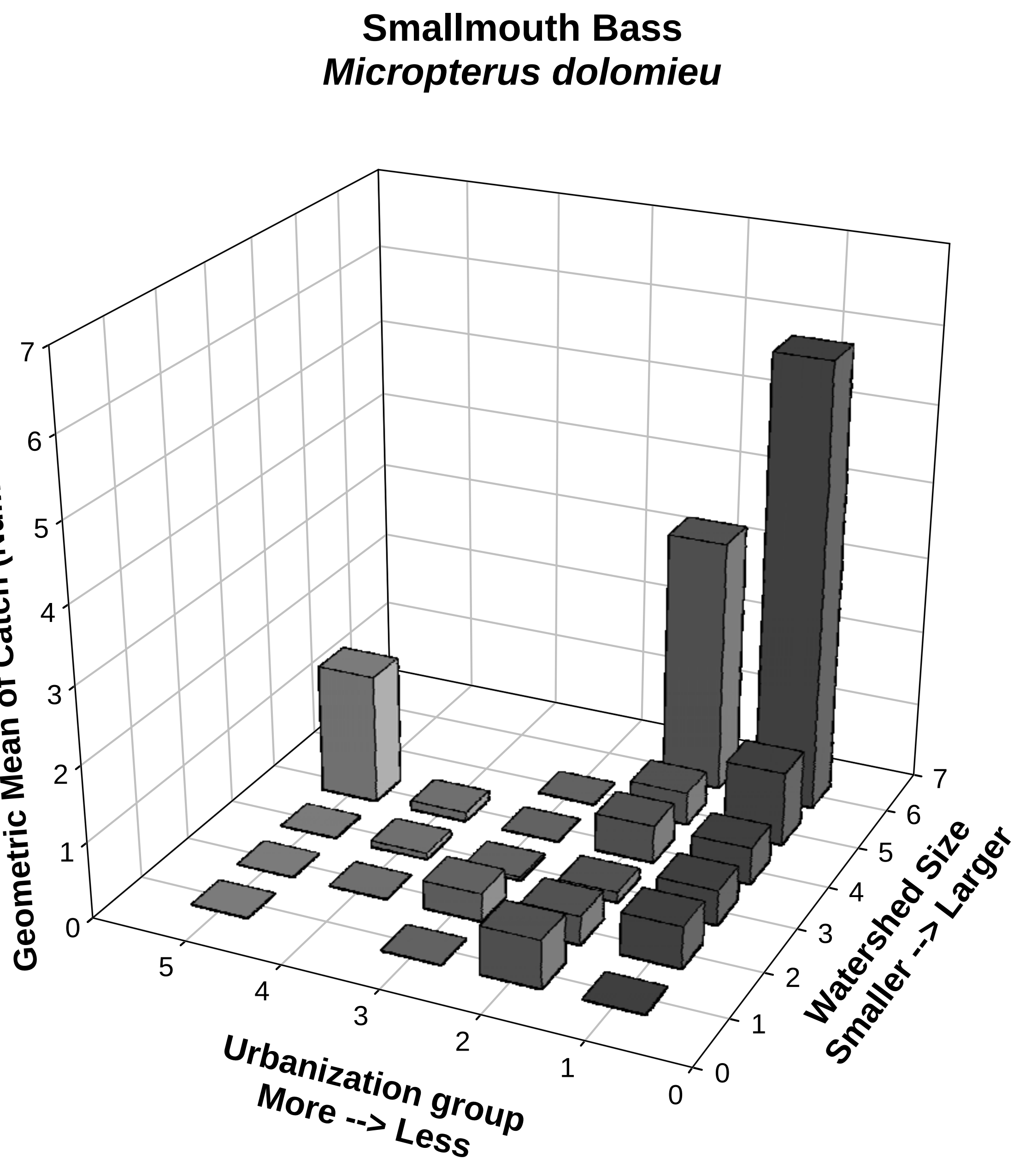

Figure A.1.16. Abundance-watershed size-urbanization relationship for smallmouth bass.

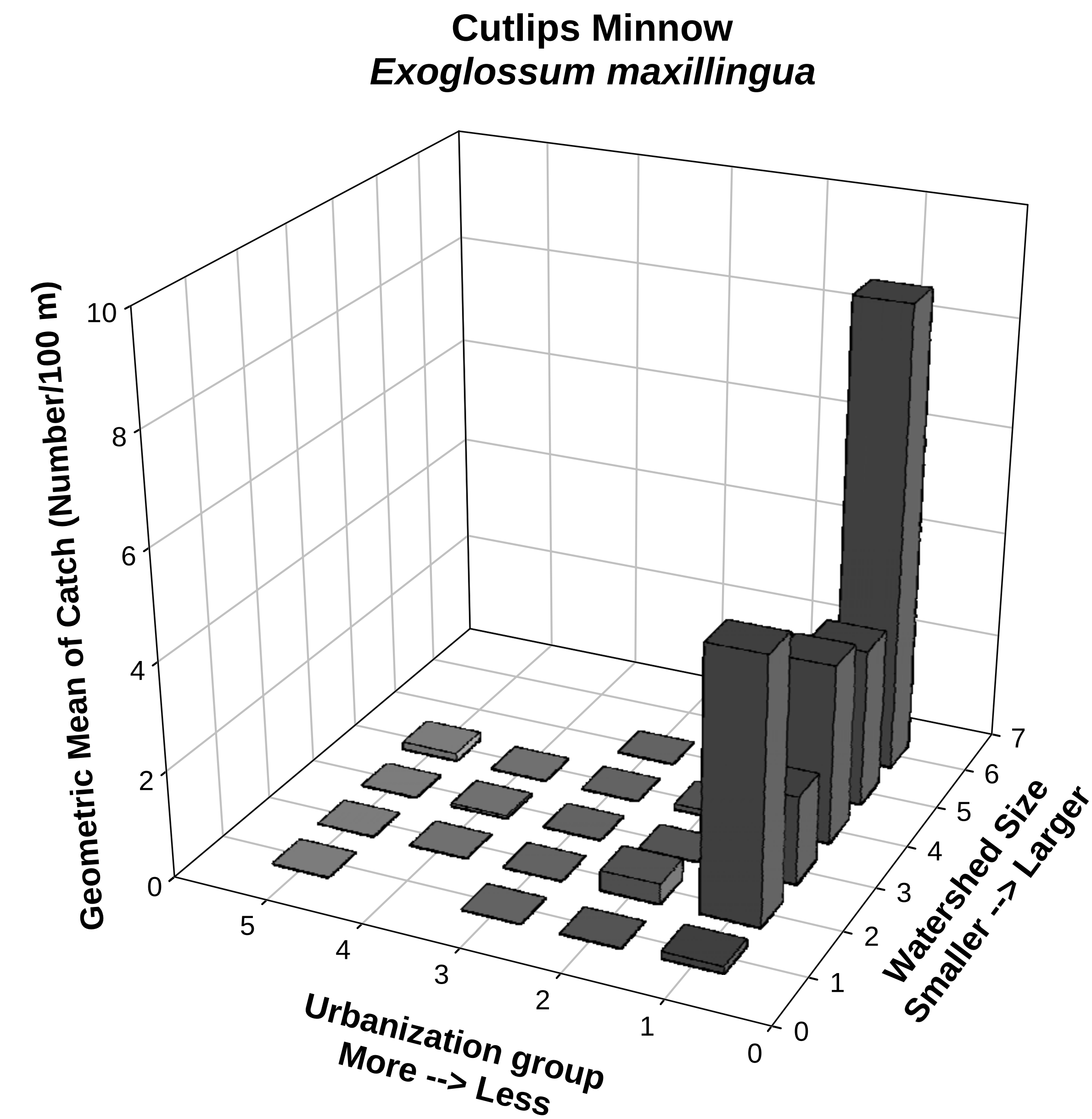

Figure A.1.17. Abundance-watershed size-urbanization relationship for cutlips minnow.

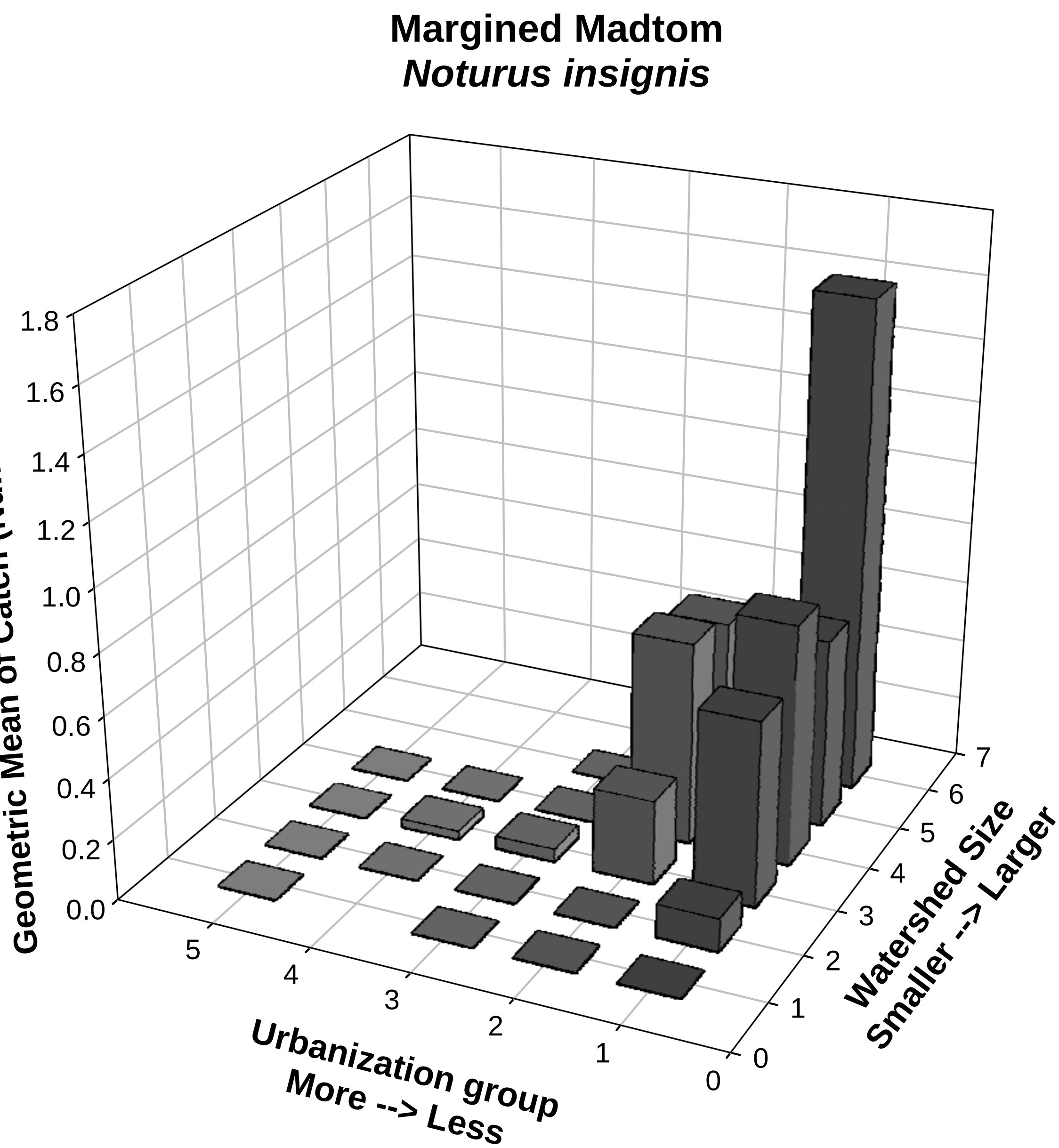

Figure A.1.18. Abundance-watershed size-urbanization relationship for margined madtom.

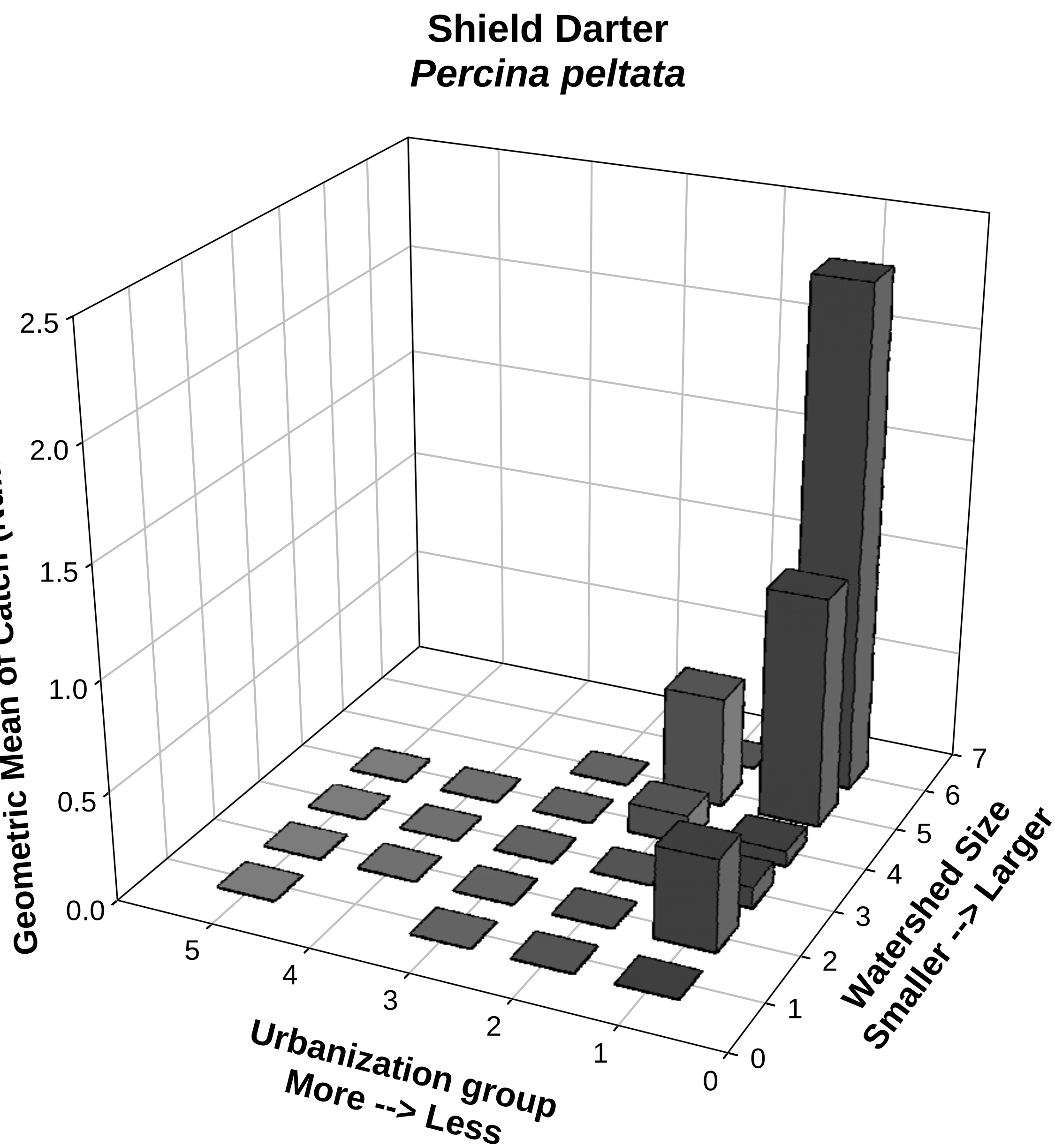

Figure A.1.19. Abundance-watershed size-urbanization relationship for shield darter.

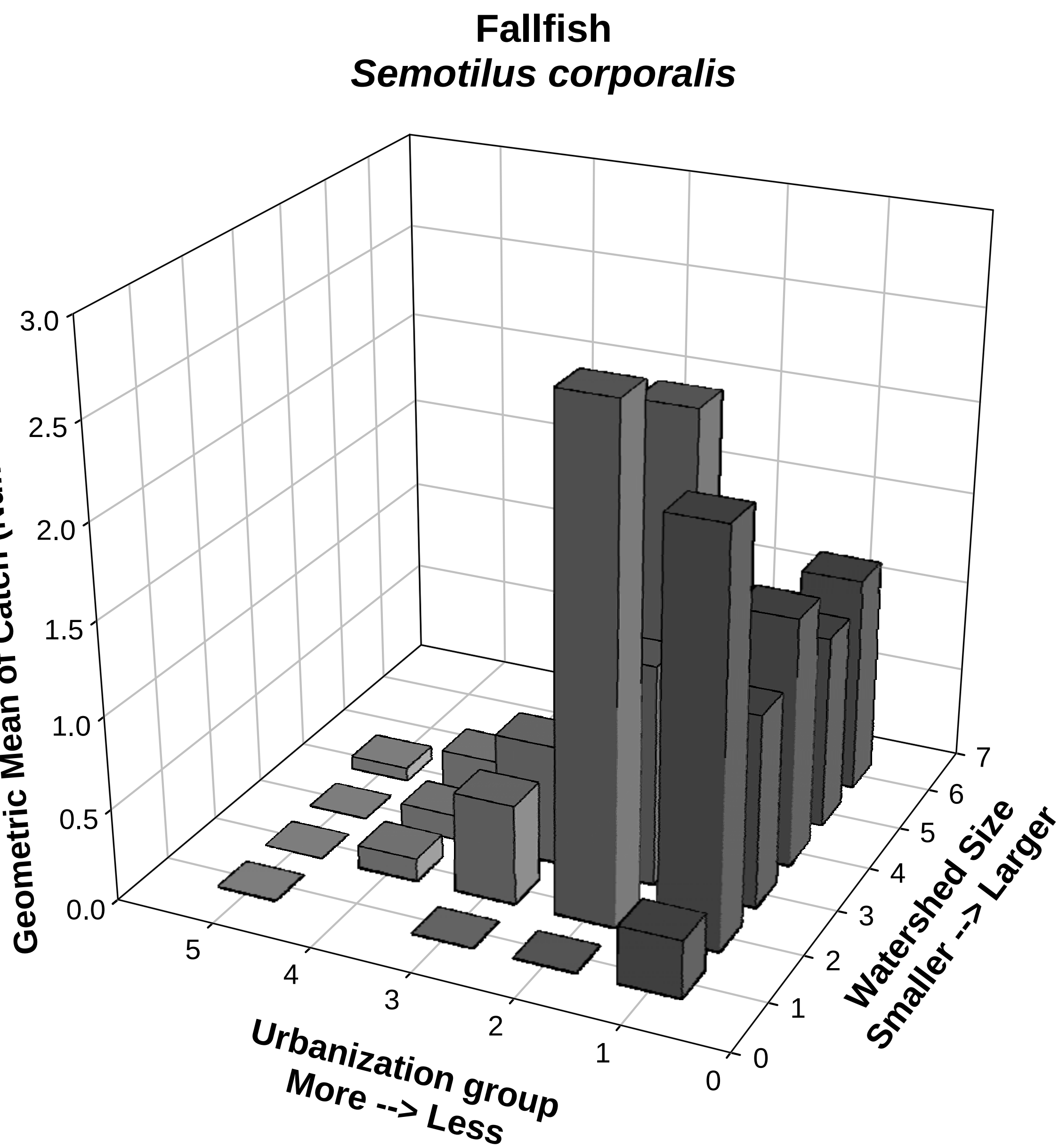

Figure A.1.20. Abundance-watershed size-urbanization relationship for fallfish.

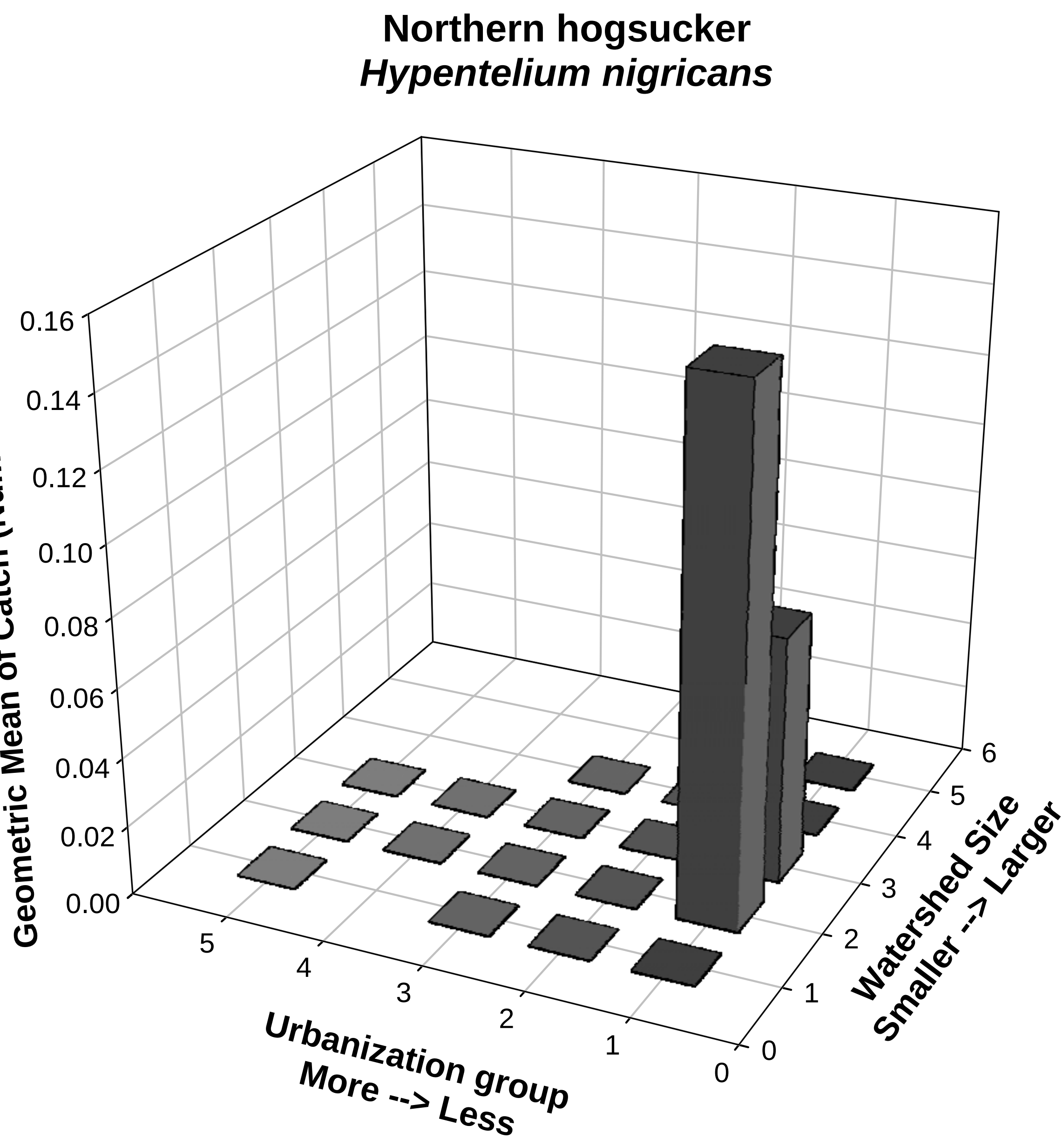

Figure A.1.21. Abundance-watershed size-urbanization relationship for northern hogsucker.

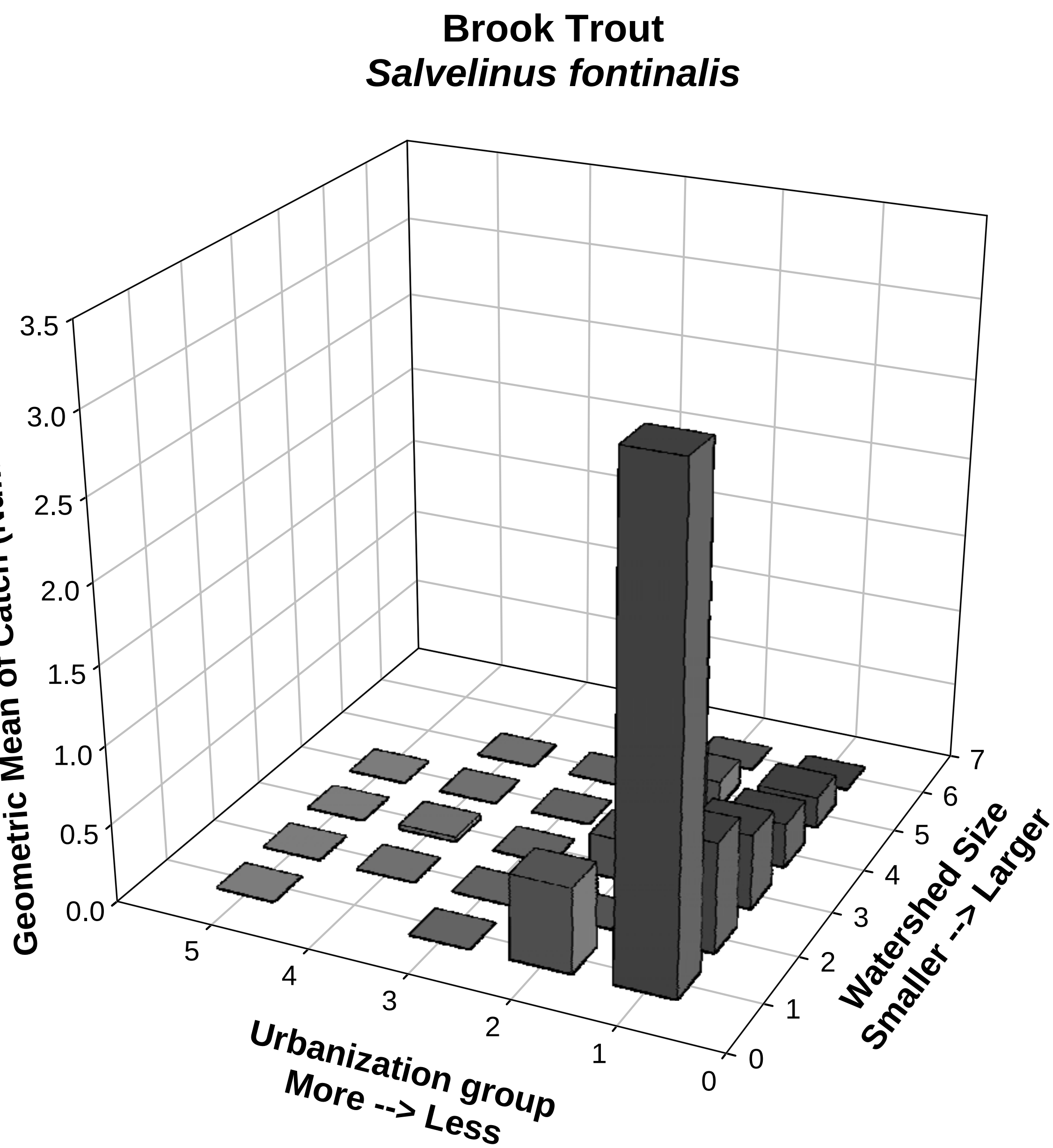

Figure A.1.22. Abundance-watershed size-urbanization relationship for brook trout.

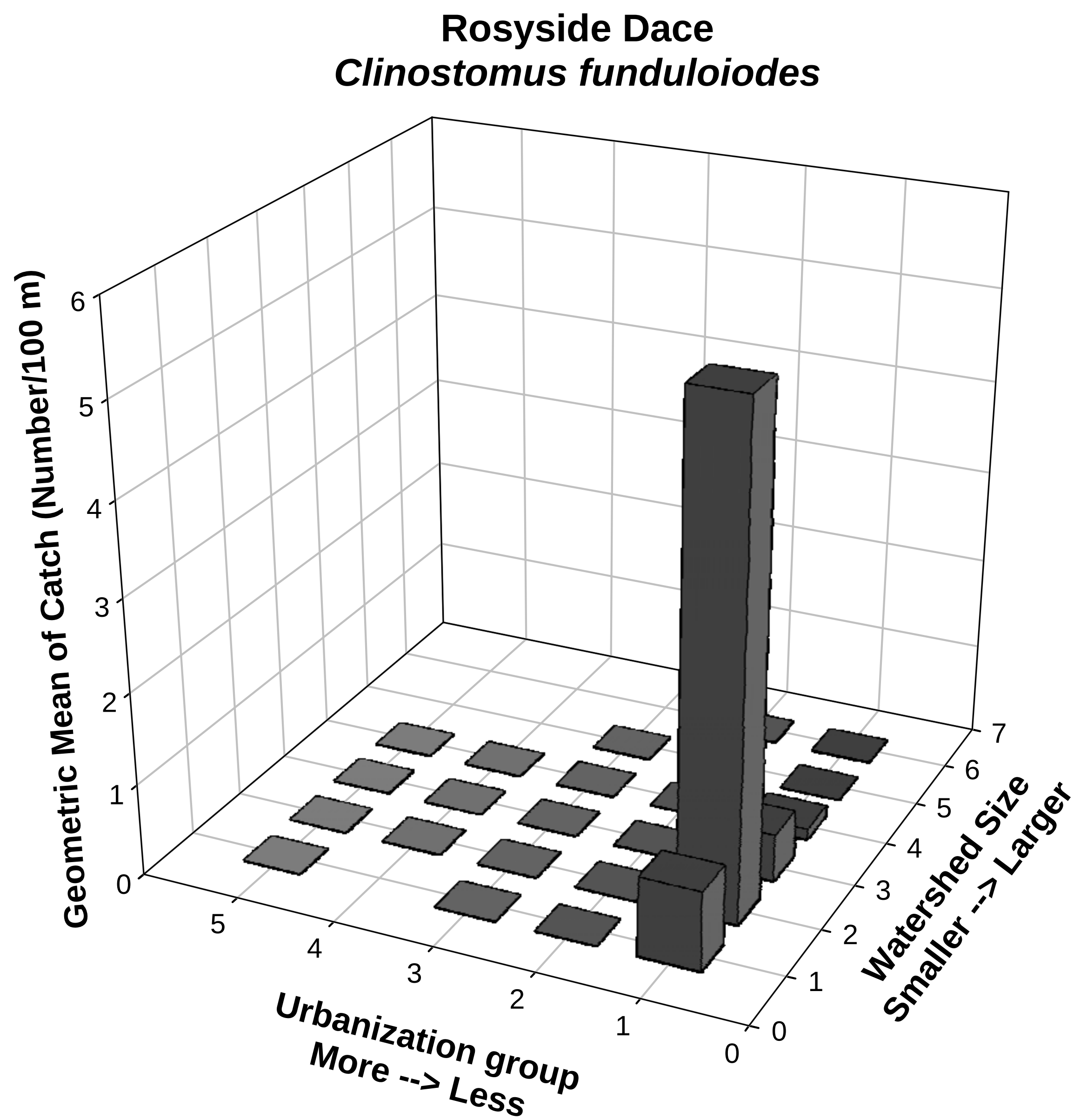

Figure A.1.23. Abundance-watershed size-urbanization relationship for rosyside dace.