MUNCIE SANITARY DISTRICT'S BUREAU OF WATER QUALITY

ANNUAL FISH COMMUNITY REPORT FOR 2010

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March 2011

BUREAU OF WATER QUALITY

LOCAL WATER POLLUTION CONTROL

"WE HAVE ONLY ONE EARTH, LET'S ALL WORK FOR ITS PROTECTION"
- John M. Craddock

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EXECUTIVE SUMMARY

- The objectives of this study were to assess the biological integrity of the fish communities within the WFWR and its tributaries within Delaware County in order to 1) evaluate the health of these aquatic communities, 2) supplement chemical assessments by evaluating overall water quality, and 3) report the results in a manner that is useful to both the public and professionals.

- Fish were collected with a Smith-Root backpack, tote-barge, or boat mounted electrofishing unit.

- Fish communities were evaluated for general health using the Index of Biotic Integrity (IBI).

- Habitat was evaluated with the Qualitative Habitat Evaluation Index (QHEI).

- IBI scores were found to be correlated with QHEI scores. High quality habitat promotes more resilient fish communities and habitat appears to have a different affect on high quality, average quality, and low quality fish communities.

- IBI scores are generally lower in tributaries as opposed to White River.

- No significant differences were found in total IBI scores upstream, within, or downstream of Muncie city limits. However, a slight decrease was observed downstream of Muncie.

- Individual metrics of the IBI show significant influence from urbanization as well as agricultural related stressors.

- Green sunfish increased in abundance from 2009 to 2010 at all Buck Creek sites. Green sunfish are a tolerant species that can survive in highly perturbed conditions. Buck Creek will be closely monitored the next few years to determine if 2010 was an anomaly due to one strong year class or a decline in biological integrity is occurring in Buck Creek.

- Smallmouth bass population estimates ranged between 618 per mile to 767 per mile and averaged 670 per mile within Muncie city limits. This equates to one fish every 8 linear feet of White River. Length frequencies indicate a substantial proportion of individuals are between 279 mm (11 inches) and 356 mm (14 inches), suggesting fishing fish this species should be above average for the next few years.

- Improvements in the fish community will likely occur with continued improvements in the Muncie Water Pollution Control Facility, reduction in Combined Sewer Overflow events, and improved land use practices at the headwaters of smaller tributaries.

INTRODUCTION

Delaware County encompasses nearly 250 miles of streams which provide habitat for 65 species of fish, 13 species of mussels, and numerous birds and mammals. These public waterways offer recreational opportunities such as fishing, canoeing, and swimming to Delaware County residence. Additionally, White River provides a source of drinking water for Muncie residence as well as residence of downstream cities such as Anderson and Indianapolis.

Prior to passage of the Clean Water Act (CWA) and its amendments in the early 1970s White River was the receiving stream for several point source stressors such as; wastewater treatment facilities, combined sewer overflows (CSOs), battery and transmission plants, and tool and die shops. These point sources were unregulated and lead to massive amounts of pollutants entering the river severely degrading water quality. Toxic pollutants that hindered all but the most tolerant species included ammonia, cyanide, lead, zinc, and chromium (Craddock 1975). In addition to these point source pollutants, nonpoint source pollutants were also contributing to the degraded water quality. Originating from agriculture and urbanization, runoff including sediment, fertilizers, insecticides, and herbicides are one of the top sources of impairment. Currently agriculture and hydromodification such as dredging, channalization, and impoundments by dams are listed as the source for over 60% of the reported impaired rivers and streams in the U.S. (U.S. EPA 2009).

Historically these threats to water quality have been evaluated with a single faceted chemistry approach. Chemical testing and bioassays provide empirical and legal validity to assessments but can not accurately provide a holistic representation of water quality. The main deficiencies of this approach include (Hughes 1990); 1) failure to account for naturally occurring differences in conventional water quality parameters, 2) failure to consider combined chemical effects, 3) toxicity tests may not be representative of indigenous species or the most sensitive species, 4) chemical testing is expensive, and 5) factors that prevent attainment of biological integrity are not limited to toxins. Finally, a chemical representation of water quality by itself fails to meet all of the fundamental goals of the CWA.

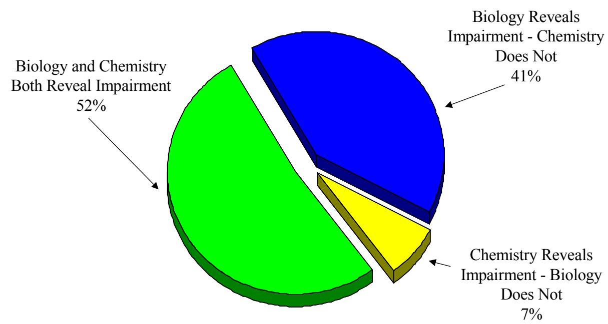

Figure 1.—Efficacy of chemical and biological assessments in detecting stream impairment.

The CWA's principal objective is to restore and maintain the physical, chemical, biological, and radiological integrity of the nation's surface water. In response to the CWA, biological criteria have been incorporated into the monitoring programs of regulatory agencies to evaluate impaired waterways (Craddock 1975; OEPA 1989; Simon & Dufour 1997; Dufour 2000). The first quantitative measure of biological integrity to address the entire fish assemblage was developed by James Karr (Karr 1981). Karr's original Index of Biotic Integrity (IBI) was composed of 12 metrics that measure species richness, trophic composition, and fish abundance and condition. Karr's original IBI has been modified for use in a variety of ecosystems and has been extended to evaluate other taxa (Simon 1991, 1994; Scott 1999; Weigel et al. 2002; Mebane et al. 2003).

Biological indicators provide many benefits to a water quality program. Biological communities reflect the cumulative impacts of the watershed condition. Fish are long lived and disturbances in their environment can be reflected at the community or individual level (e.g. DELT anomalies, % tolerant species and age and growth). Fish represent a variety of trophic levels; omnivores, herbivores, insectivores, planktivores, and piscivores. Fish are ubiquitous and found in even the smallest of streams. Biological sampling is also relatively inexpensive compared to chemical analysis. In addition, descriptors of the fish community are more easily related to the public.

While the benefits of biological criteria are widely known they are not intended to replace chemical sampling. Implementation of the two in concert provides the most holistic representation of water quality. It has been found that 40% of impaired streams in Ohio were detected by biological assessments and missed by chemical sampling (OEPA 1994) (Figure 1). While 7% was found only with chemical sampling. In addition, chemical testing is sometimes necessary as a follow up to pinpoint the exact cause of disturbances found by biological testing. A single approach or a single statistical framework (e.g. Shannon Diversity Index) is insufficient at describing every variable that affects water quality. Multiple sampling approaches coupled with multiple analyses which take into account the nuances of the relationship at hand is necessary to formulate a holistic conclusion on water quality.

The Bureau of Water Quality (BWQ) began supplementing its chemical sampling with biological assessments of fish and macroinvertebrates in 1973 (Craddock 1975). The combination of monitoring data along with the cooperative efforts of local industries has accounted for an enormous reduction of toxic pollutants in White River. However, they have also begun to highlight the extent of nonpoint source (NPS) stressors. Today, the recently unmasked effects of NPS pollution have become the leading cause of water quality impairment in the Midwest, demanding greater emphasis on the broad sensitivity of biological assessments (IDEM 1998, OEPA 2000).

The objectives of this study are to assess the biological integrity of the fish communities within the WFWR and its tributaries within Delaware County in order to 1) evaluate the health of these aquatic communities, 2) supplement chemical assessments by evaluating overall water quality, 3) evaluate the smallmouth bass population and 4) report the results in a manner that is useful to both the public and professionals.

METHODS

Assessment of the biological integrity of the fish communities and habitat of the WFWR and its tributaries.—Prior to 1990, fish sampling was sporadic, using a backpack electrofishing unit, electric seine, or kick seine. In 1990, the BWQ began a standardized annual sampling program. Variation in sampling design prior to 1990 precludes the use of some statistical analysis. Fish sampling methods were based on the electrofishing guidelines provided by Simon and Dufour (1997) and the Ohio Environmental Protection Agency for assessment of streams within the Eastern Corn Belt Plains ecoregion (OEPA 1989).

Beginning in 1990, fish were sampled using one of three types of Smith-Root Inc. electrofishing gear. Each unit emits a pulsed direct current of electricity that temporarily stuns fish so they can be netted and placed in a live well. Wadeable sites were sampled with a tote-barge electrofisher (TBS). In extremely small tributaries where a TBS unit was too bulky to be hauled by one person, a lightweight, battery-powered backpack unit (BPS) was used. At sample sites too deep to wade, a boat mounted electrofishing unit was used.

From 1980 through 2010, the BWQ conducted 1,089 sampling events at 186 synoptically selected sites from the WFWR and its tributaries, as well as reference sites in the Mississinewa River. Annual stations were chosen based on historical baseline sample stations, presence of a riffle-run-pool complex, proximity to potential stressors, and site accessibility. Variables that most significantly affect electrofishing efficiency and aquatic community condition are measured at each sample location prior to sampling. Conductivity, water temperature, and dissolved oxygen were measured with a portable YSI Inc. meter following standard methods (4500-O G, 4500-H B, and 2510-B respectively).

Sample sites were classified as headwater (those with drainage areas $\leq$ 20 mi.^{2}), wading (drainage areas > 20 mi.^{2} and shallow enough to wade) and boat sites (those sites that are too deep to wade). Each stream category was evaluated with a unique set of metrics specifically calibrated by drainage area (Appendix B). Headwater and wading sites were sampled for distances of 50 to 200 m, and boat site lengths were sampled for distances of 450 to 1050 m.

Fish were processed according to Ohio EPA (1989) and Simon and Dufour (1997) methods for determination of IBI and MIwb scores at all sample sites from 1990 to 2008. Fish were identified following Trautman (1981) and sorted by species and measured in one of two ways. Game fish (ex. basses, bluegill, and catfish) were individually measured for length (millimeters) and weight (grams). Non-game species (ex. minnows, suckers, and darters) were bulk weighed and measured for a single minimum and maximum length. Fish under 25 mm were not included to reduce the bias of young-of-the-year fish. Museum vouchers are kept of all fish species collected by the BWQ. One representative of each species from each subwatershed is taken as a voucher every five years. Vouchered specimens are cataloged and maintained by the BWQ for verification of identification and as historical representatives of species characteristics. All other fish are released.

The Index of Biotic Integrity (IBI), originally developed by James Karr, and the Modified Index of Well-being (MIwb) (Gammon 1976) provide sensitive and reproducible measurements of the integrity of fish communities (OEPA 1989). These indices have been calibrated for use in specific ecoregions defined by the mutual presence of geographic variables pertinent to biological potential. Streams within the same ecoregion and with comparable drainage will contain similar structural communities that have predictable and measurable responses to perturbation.

The IBI is composed of twelve metrics that measure functional aspects of fish communities including species composition, trophic composition, and fish condition. Each metric is scored according to the degree of deviation from a "healthy" or least impacted stream of comparable size (1 = severe deviation, 3 = moderate deviation, and 5 = little or no deviation). The total score of 12 to 60 is used to assign a narrative description of *very poor, poor*, *fair*, *good*, or *excellent* to the biological integrity of the community within the sampled stream segment (Appendix B).

The MIwb, used primarily as a supplement to the IBI, consists of four measures of fish community structure based in part on the Shannon diversity index. Healthy communities are defined in part by the presence of diverse assemblages, making MIwb scores a reliable measure of general water quality. Scores of 0 to 12 reflect community homogeneity based on relative species numbers and weights. As with the IBI, narrative descriptions of *very poor* to *excellent* are then assigned to the stream segments (Appendix B).

Beginning in 2002, QHEI measurements were taken in conjunction with each sampling event according to the guidelines provided by Rankin (1989). Habitat assessments allow a preliminary estimation of the potential contribution of habitat alterations (as opposed to chemical pollution) as the cause of impairment. The Qualitative Habitat Evaluation Index (QHEI) measures variables that are pertinent to biological potential including the quality of substrate, cover, channel morphology, riparian zone, and riffle-run-pool complexes. Habitat quality is scored from 0 (poor quality) to 100 (high quality).

Comparison of QHEI scores to biological index scores is a vital step in determining potential sources of impaired biological communities. Habitat quality is often the limiting factor of biological integrity; therefore, the quality of a fish community rarely exceeds the quality of habitat in which they live (Wang et al. 2001). Sites that have severely altered habitats due to channelization or dredging, for example, would not be expected to hold high quality fish communities. In these cases, the source of the disturbance is described clearly by the habitat assessment. Conversely, high quality habitat and poor biological integrity may be an indication of point source pollution. In addition, spatial differences in IBI, QHEI, and the fish community composition are analyzed.

Two additional studies were conducted in 2010; a smallmouth bass *Micropterus dolomieu* and rock bass *Ambloplites rupestris* population estimates and evaluation of potential estrogenic compounds in White River as determined by morphological measurements of bluntnose minnows. These studies are currently being completed and will be submitted for publication in the peer reviewed literature.

Fish assemblage statistical analysis.— Correlation analysis was used to determine if a relationship exists between average IBI, QHEI, and drainage area from 2004 - 2010. Following correlation analysis more specific cause-effect models are typically created using linear regression and/or Analysis of Variance (ANOVA) for multiple co-variate (i.e. influence of land use variables). As such, these parametric techniques and the less powerful non-parametric Kruskal Wallis test for categorical variables are typically used when relating IBI and QHEI (Simon and Sanders 1999; Smogor and Angermeier 1999; Santucci et al. 2005; Morris et al. 2006). These parametric procedures require important assumptions to be met and include normality of the dependant variable and homogeneity of variance (i.e. heteroscedasticity). Normality of the dependant variable states the variable follows a normal distribution throughout the values of the independent variable. Violation of this assumption can lead to inaccurate confidence intervals and has the potential to completely distort the true relationship. The second assumption states the variance around the regression line is equal for all values of the independent variable. If this assumption is violated the consequence can be overestimating the goodness of fit or concluding the relationship is stronger than it actually is. However, violation of this assumption does not undermine the relationship that may be present. Authors have argued violating either assumption will not affect the outcome if the data were collected from random samples (Bohrnstedt & Carter 1971; Havlicek & Peterson 1977; Edgell & Noon 1984). In contrast, others have argued strict adherence to both assumption must be followed or alternative non-parametric statistics be used (Cane 2001; Alexander & DeShon 1994). There are valid arguments on both sides. Data that do not meet these assumptions can and often are transformed or weights applied to each observation. While these manipulations assist in meeting assumptions they make the final relationship much more difficult to interpret and relate to ecological reality. To reduce the chances of committing type-II error we use the conservative approach and attempt to adhere to the underlying assumptions of the statistical test. The normality assumption is evaluated with the Shapiro-Wilks test and the heteroscedasticity assumption is evaluated with White's Test. When these assumptions are violated quantile regression will be used to evaluate the relationship of the variables. Quantile regression has been used to evaluate conditional responses with central tendency, variance, and the shape of the response variable (Koenker & Machado 1999). While not often used in ecology, quantile regression provides a more complete picture of conditional relationships (Cade & Noon 2003). This method originated and is used quite extensively in economics (Koenker & Bassett 1978; Koenker & Hallock 2001). Quantile regression is not bound by the strict assumptions of linear regression. Furthermore it can be used to describe the response of multiple quantiles (e.g. 25th and 75th quantile) which can help determine how habitat differentially relates to biological integrity, potentially assisting with feasibility studies which address expected improvements in the biological community.

In ecology quantile regression has been used to address the analytical issues of unmeasured limiting factors (Kaiser et al. 1994; Terrell et al. 1996; Thomson et al. 1996; Huston 2002; Doll, in press). The "limiting factor" concept, also known as Liebig's Law of the Minimum, states that the response of a biological process will be determined by the factor in least supply (Liebig 1840). Quite often closed systems (e.g. lotic water bodies) are consistently affected by the same group of factors in a similar intensity and the factor in least supply can more easily be determined by linear regression. However, when a data set spans several bodies of water or are collected from an extraordinarily large geographic area different groups of data points may be influenced by different limiting factors. Therefore, making it difficult to accurately determine the limiting factor. This later scenario can lead to unequal variances and variable central tendency along the response measurement. Which inherently violates the assumptions of linear regression or linear regression of the mean response may not be significant. To account for these unmeasured limiting factors quantile regression is used in the present analysis (Cade et al. 1999). Quantile regression was conducted using PROC QUANTREG (SAS 9.2).

Spatial variability of IBI scores in White River were evaluated from data collected between 2004 and 2010 (data are presented from all sites in Figure 10 while only sites that were sampled a minimum of 3 times within the time frame of interest were included in the analysis). Total IBI scores and individual IBI metrics were analyzed using a mixed linear model to evaluate the relationship with QHEI, spatial location in reference to city limits (upstream, within, and downstream), and measurement year. Total QHEI score, spatial location, and an interaction between total QHEI score and spatial location were modeled as fixed effects and measurement year was modeled as a repeated effect to evaluate variability within sampling stations. The mixed model was selected due to it permitting the analysis of unbalanced designs with missing data, which is the case in our dataset

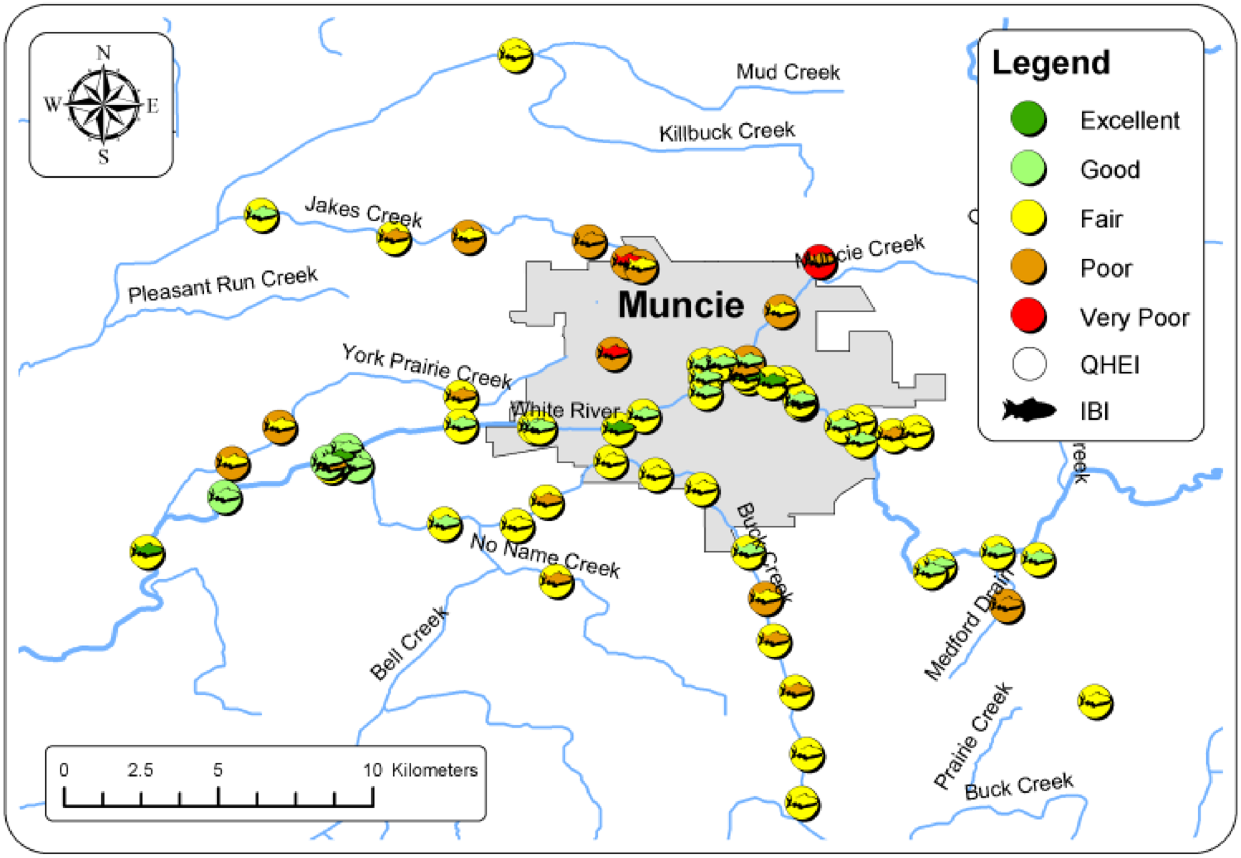

Figure 2.—Fish community and habitat health at sites sampled by the BWQ in 2010.

where not every site was sampled each year. Additionally, the mixed model allows for the independent data to be correlated unlike repeated measures ANOVA. In our dataset it is reasonable to assume measurements at the same site among years are correlated. To evaluate this assumption I applied five different covariance structures to the year effect. Each covariance structure assumes different correlative relationships between adjacent years. For example the variance components (VC) structure assumes no correlation between annual measurements at the same site, which is analogous to a standard repeated measures ANOVA, while the first order autoregressive (AR(1)) structure assumes correlations among adjacent years are equal and decline exponentially the further apart the measurement year is. For detailed descriptions of the five covariance structures see Kincaid (2005). The final covariance structure used for interpretation was selected based on the change in Akaike's Information Criterion (ΔAIC). The lowest AIC value is considered the best. The ΔAIC is calculated by taking the observed AIC for the model and subtract the smallest AIC from all the competing models. The model with the lowest AIC value will have a ΔAIC of 0.0. If the interaction term was not significant at the $P < 0.05$ level it was dropped for the final model. Least squared adjusted means were calculated in relation to city limits (i.e. upstream, within, and downstream). Differences between adjusted means were evaluated with Tukey-Kramer adjusted pair-wise differences. Adjusted means presented in text have been converted back to their original scale unless otherwise stated. All models were fit using PROC MIXED (SAS 9.2).

Spatial variability of IBI scores among watersheds were assessed using a similar approach as above. Spatial trends were evaluated by 14-digit HUC watershed unit rather than reference to city limits. As above, sample years were from 2004 to 2010 and only sites that were sampled at least three times were included. Some metrics were evaluated based on drainage area to facilitate dif-

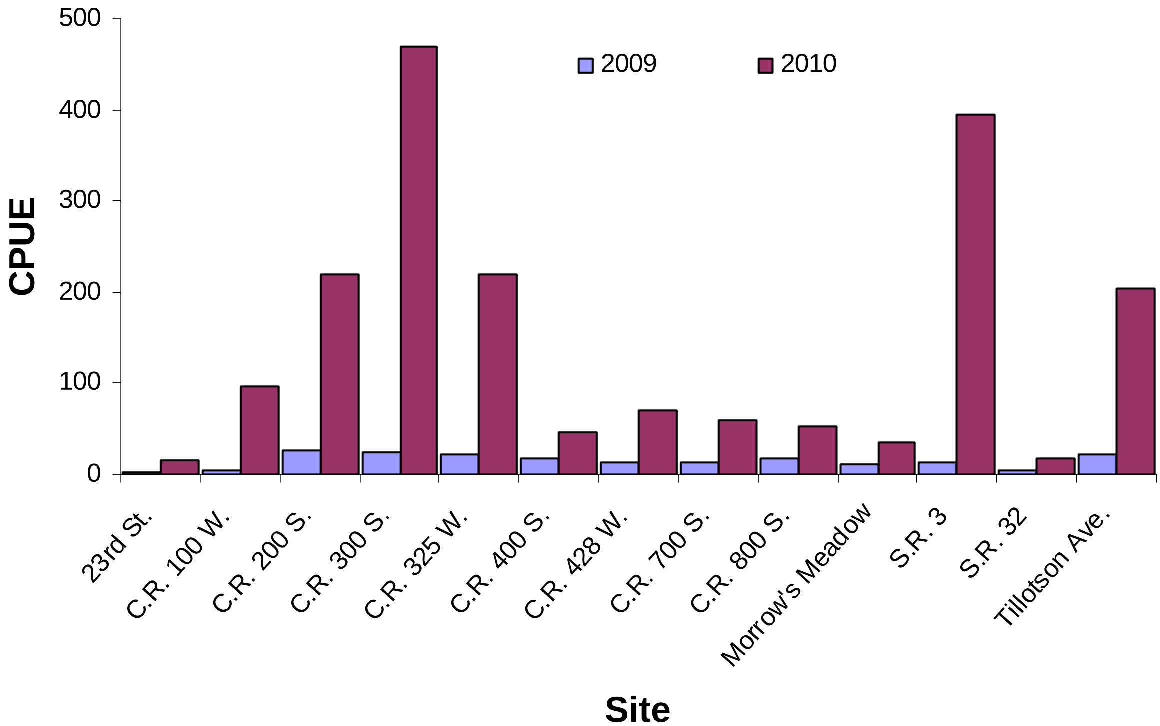

Figure 3.—Green sunfish catch per unit effort (CPUE) at Buck Creek sites from 2009 and 2010.

ferences in IBI scoring criteria (i.e. headwater <=20 m^2, and wadeing >20m^2). GIS statistics were obtained from Indiana's GAP program (Appendix G-1)All models were fit using PROC MIXED (SAS 9.2).

Prior to analysis count metrics were square root transformed, percentage metrics were arcsine-square root transformed, and catch per unit effort data were natural log transformed. All analysis was conducted in SAS 9.2 and significance was set at $P < 0.05$.

Smallmouth Bass population estimate.—A multi-pass depletion survey was conducted to estimate stream wide abundance of smallmouth bass. Smallmouth bass were sampled using a tote-barge electrofishing unit at six sites selected at random within Muncie city limits. Block nets were used at each end of the sample sites to prevent fish from moving in or out of the sampling area during the survey. Between 4 and 6 passes were conducted at each site. After each pass fish were counted and measured for length and weight. Fish were immediately released downstream of the sampling area outside of the blocked area. Only fish age $\geq$ 1 were included in the analysis. Young of the year fish were determined by gaps in the length frequency. Stream wide abundance estimates were calculated following the hierarchical model described by Royle & Dorazio (2008). Bayesian inference is used to fit the hierarchical model. This method is different than standard frequentists inference in that the results produce a direct probability statement about the parameters estimated. For example, each parameter (e.g. population abundance) has an associated distribution which is used to estimate the 95% credible intervals (95% CI). The credible intervals state that the true unobserved value has a 95% probability of falling between this range. The model is fit using the OpenBUGS 3.1.1 software.

Proportional stock density (PSD) and relative stock density (RSD) is used to describe the length frequency distribution of the fish population. PSD is the percent of individuals that are longer than stock size that are also longer than preferred size. Each fish species has a different stock size and preferred size designation. Smallmouth bass stock size is 178 mm (7 inches) and preferred size is 279 mm (11 inches). For example, if there were 75 fish ≥ stock length and 25 fish ≥ preferred length then the PSD is 33 (25/75 *100=33). RSD is the percent of individuals that are longer than stock size and are also longer than a different specified length. Smallmouth bass RSD is calculated with a specified length of 305 mm (12 inches) and 356 mm (14 inches) in this report.

RESULTS

Index of Biotic Integrity and Modified Index of Well-Being.—The BWQ sampled 62 sites from the WFWR and its tributaries in Delaware County in 2010 in order to evaluate the health and integrity of their fish communities. Complete lists of metric scores, sample collections, and precise site locations are available in the Appendices.

IBI scores for 2010 ranged from a low of 18 *very poor* at York Prairie Creek near Maddox Drive (YPC-9.0), to a high of 58 *excellent* at White River near the West Side Park (WHI-313.4) (Figure 2). A significant difference was found between IBI scores on White River and tributary sites (Wilcoxon test; Z = 6.14, P < 0.001). Wadeing sites on White River had a mean score of 50.9 (SE = 0.681) *good* while the mean score for sites on tributaries was 37.1 (SE = 1.282) *fair*. The tributary mean is similar to 2009 (36 *fair*) but lower than 2008 (41 *fair*). The higher average in 2008 was due to the sampling on Cabin Creek (IBI average = 50 *good*) and Stoney Creek (IBI average = 48 *good*) in 2008.

Ohio EPA suggests MIwb scores should be used only when replicate samples are taken, therefore MIwb scores are reported in the appendices merely to supplement IBI scores. MIwb scores for 2010 ranged from a low of 5.5 *poor* at Buck Creek near S.R. 3. (BUC-11.3) to 10.3 *excellent* at White River directly downstream of the High Street Dam (WHI-314.8). A comparison of IBI and MIwb scores for 2010 indicate a close similarity between the two matrices (r = 0.857, N = 39, P < 0.001). Narrative descriptions of the MIwb and IBI were in agreement for 15 out of the 39 sites included in MIwb calculations.

In 2010 an unusually high number of green sunfish Lepomis cyanellus were sampled. In fact green sunfish increased in abundance from 2009 to 2010 at every Buck Creek site sampled (Figure 3). Comparisons of the natural log transformed CPUE values indicated an overall significant increase from 2009 to 2010 (t(12)=1.78, P < 0.001). Green sunfish are a tolerant species that typically indicate a shift from fair to poor quality. They can successfully reproduce and outcompete in the most perturbed conditions. While the green sunfish is relatively common in Indiana and moderate numbers are typical, the large increase in 2010 is of concern. It is possible that one strong year class has been recruited into the population and their success was an anomaly. Nevertheless Buck Creek will be closely monitored over the next few years to see if the 2010 sample is an anomaly or a true shift in biological integrity.

Qualitative Habitat Evaluation Index.—QHEI scores for 2010 ranged from a low of 19 poor at Hamilton Ditch near C.R. 300 N. (HAM-0.2) to a high of 72.5 good at White River near C.R. 575 W. (WHI-308.5 & WHI-308.7) (Figure 2). As with IBI scores, QHEI scores were significantly lower in White River tributaries (Wilcoxon test; Z = -3.53, P < 0.001). Agriculturally related hydromodifications such as channelization and riparian removal on smaller streams were noted as the primary causes of low QHEI scores. Of the QHEI metrics, Channel Morphol-

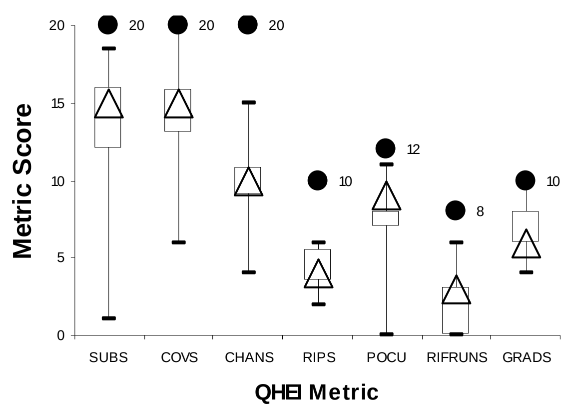

Figure 4.—Box and whisker plot of QHEI Metrics for 62 sites sampled in 2010. SUBS=Substrate Score, COVS=Cover Score, CHANS=Channel Score, RIPS=Riparian Score, POCU=Pool/Current Score, RIFRUNS=Riffle/Run Score, and GRADS=Gradient Score.
Δ = Median QHEI metric score
● = Maximum metric score

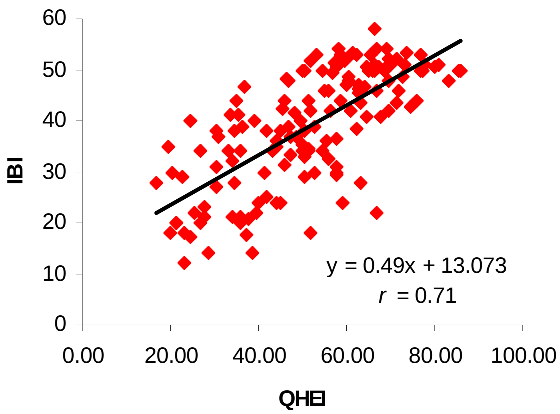

Figure 5.—Correlation between IBI scores and QHEI scores from 2004 to 2010.

ogy, Riparian, and Riffle/Run Quality had the poorest overall quality when compared to expected maximum score (Figure 4), and functional riffle/run/pool complexes were absent from 36% of all sites sampled. The majority of which were located in tributaries.

Relationships between IBI, QHEI, and drainage.—Since 2004, the BWQ has sampled 147 individual sites (many sampled more than once). During this time period, a significant positive relationship was detected between IBI scores and QHEI scores (r = 0.71, N = 147, P < 0.001) as would be expected given the dependency of biota on habitat (Figure 5). All QHEI metrics were found to be significantly correlated to IBI scores

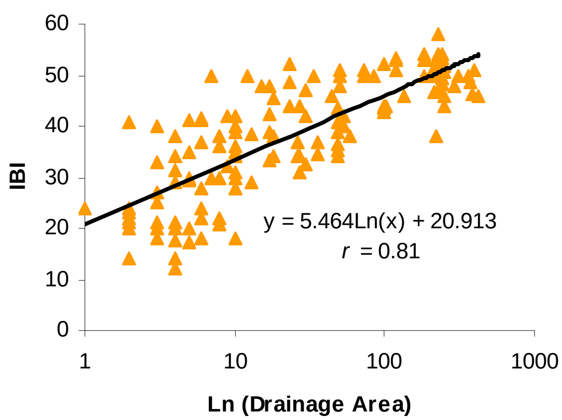

Figure 6.—Correlation between IBI scores and Ln(Drainage Area) from 2004 to 2010.

(Appendix C). Additionally, IBI metric #4, the number of sucker/minnow species, appeared to have the weakest correlation to QHEI metrics.

In addition to examining the relationship between IBI and QHEI scores, IBI and QHEI scores were compared with drainage area. Drainage area had a significant positive relationship with IBI (r = 0.81, N = 147, P < 0.001) and QHEI scores (r = 0.64, N = 147, P < 0.001) (Figure 6 and 7). Each index is designed to assess streams irrespective of drainage area; therefore, the implication is that smaller streams are either more likely to be altered or are more susceptible to equivalent alterations than larger streams.

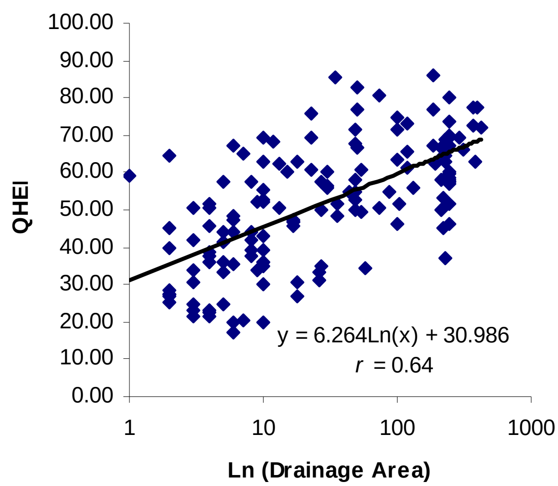

Figure 7.—Correlation between QHEI scores and Ln(Drainage Area) from 2004 to 2010.

The normality and heteroscedasticity assumptions were violated in the relationship between IBI and QHEI scores (Shapiro-Wilks; W = 0.981; N = 147; P = 0.038) (White's Test: X^2 = 9.99; df = 2; P = 0.007). Considering the strong correlation and theoretical justification for the relationship the two datasets are still considered associated; however, the magnitude and precise estimate of this relationship can not be described with linear regression. Therefore, quantile regression was used to evaluate the conditional response of QHEI on IBI. Five quantiles were evaluated (0.10, 0.25, 0.50, 0.75, & 0.90) and indicated IBI scores are much more variable in poor habitat than they are in high quality habitat (Figure 8). Suggesting fish communities are more resilient with high quality habitat. Several covariates which were not evaluated for their influence including the proximity to high quality habitat and temporary point source disturbances can be contributing to the wide fluctuation of IBI scores with poor habitat. In addition, biological integrity in the upper quantiles exhibit a different relationship with habitat than fish communities in the lower quantiles. A comparison of slopes from all quantiles (0.10 to 1.00 in 0.05 increments) shows the slope is relatively constant for quantiles at and below 0.50 while there is a precipitous decline for quantiles above 0.50 (Figure 9). Therefore it is concluded that habitat has a different affect on high quality, average quality, and low quality fish communities. This variable relationship has implications when attempting to predict improvements in IBI due to QHEI improvements. For example, a 20 point in-

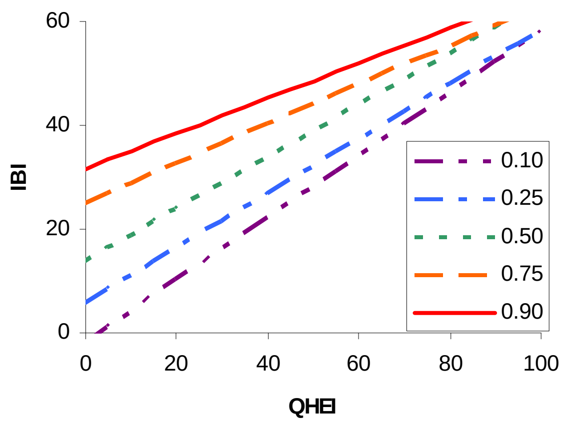

Figure 8.—Quantile Regression graph for select quantiles of QHEI vs IBI for years 2004 to 2010.

crease in QHEI scores should increase the IBI score by 6.8, 10.0, and 12.0 points for high quality, average and low quality fish communities, respectively. This relationship is likely due to other limiting factors being more important as some sites.

The upper quantiles are a representation of the biological integrity that is only influenced by habitat as summarized by QHEI scores. The decrease in slopes with increasing quantiles suggests other limiting factors are affecting the lower quantiles. More specifically QHEI does not appear to influence IBI scores as strongly as standard linear regression suggests (Table 1). It is important to

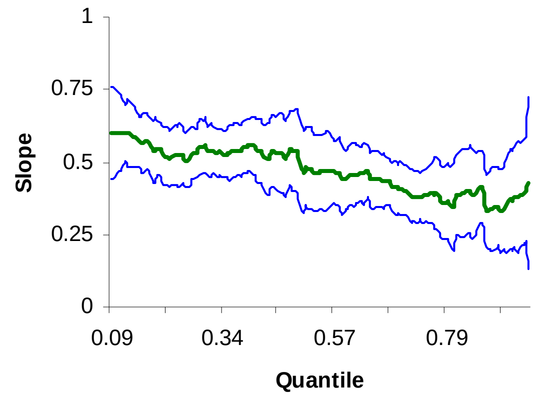

Figure 9.—Slope comparison for all quantiles of QHEI vs IBI for years 2004 to 2010. Solid green line represents slope values and dashed blue lines are 95% Confidence Limits.

note that this analysis does not suggest habitat is not influencing the fish community at the other sites. The analysis is merely suggesting that there are limiting factors at those sites other than reach scale instream habitat. It is also possible that individual metrics within the QHEI are having conflicting or differential influences on the fish community.

In general, studies that evaluate the relationship between individual or several covariates on fish communities as represented by some form of IBI, multivariate ordination, or measure of abundances will use standard parametric and nonparametric test (Simon & Sanders 1999; Sullivan et al 2004). While this yields valuable information more specific and uncorrelated relationships could be found by applying quantile regression (Cade & Noon 2003). This analysis suggests that the IBI can not be compared with the QHEI using standard linear models. Transformation of the data

Table 1.—Intercept and slope values for select quantiles and linear regression. Significance (*P* < 0.05) indicated by *

Quantile	Intercept	Slope
0.10	-1.78	0.600*
0.25	5.94	0.526*
0.50	14.00*	0.500*
0.75	12.05*	0.381*
0.90	31.72*	0.338*
Linear Reg.	13.73*	0.490*

could be conducted and may allow assumptions to be met for parametric statistics. However, transformations would make interpretation of the data more difficult. The discrete scoring method used in the IBI may also be contributing to the heteroscedasticity nature of the relationship. Continuous scoring methods have been shown to improve performance of a macroinvertebrate IBI and lead to increased precision in the number of condition classes the index could distinguish (Blocksom 2003).

White River spatial variability.—Spatial biotic integrity and habitat index score trends through Muncie reflect the cumulative impact the city imparts on the water quality of White River. Index of Biotic Integrity scores fluctuate along White River as it flows from sites above Muncie's influence to within the city where the impact of urban land use, CSOs, and the Muncie Water Pollution Control Facilities (MWPCF) and Yorktown (YWPCF) are present (Figure 10).

The preferred covariance structure for the model describing total IBI score and metrics are in Appendix E. The different structures account for year-to-year variability that would be lost by analyzing each year separately. Additionally, including each year individually allows for a more robust analysis to detect potential trends. Only met-

Table 2.—Results of mixed model to determine differences in IBI scores in reference to Muncie City limits. Significance ($P < 0.05$) indicated by *

Metric	F-value for Fixed Effects QHEI	City Limits
IBI	2.44	2.31
One	0.63	3.21
Two	0.00	0.92
Three	2.65	0.81
Four	0.38	2.84
Five	4.75*	0.24
Six	4.86*	3.03
Seven	7.20*	18.58*
Eight	0.48	0.63
Nine	2.61	2.80
Ten	0.68	2.27
Eleven	0.74	0.85
Twelve	0.72	0.43

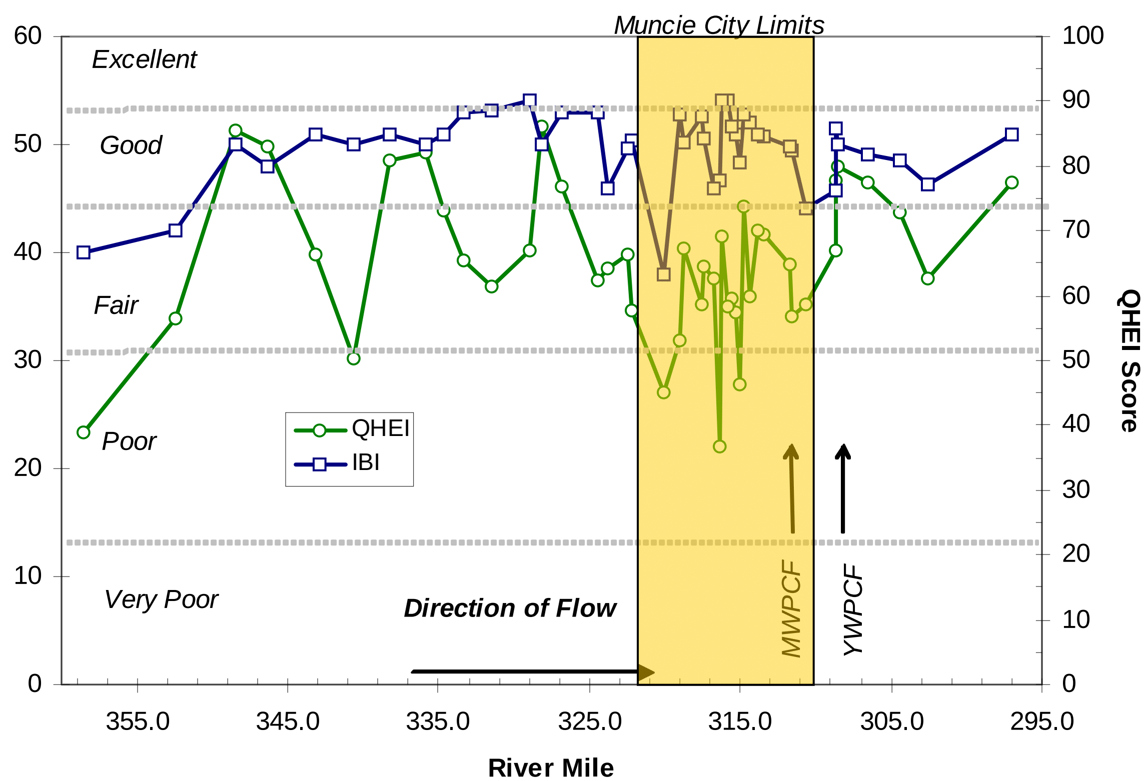

Figure 10.—Average IBI and QHEI scores from White River from 2004-2010.

Table 3.—HUC-14 Watershed.

14-Digit HUC	Watershed Name	Area (Ha)
05120201010120	White River - Truitt Ditch	4,769
05120201010130	White River - Muncie Creek	3,480
05120201020020	Buck Creek - Macedonia Creek	6,339
05120201020060	White River - Buck Creek	3,553
05120201030010	White River - York Prairie Creek	4,852
05120201040030	Jake's Creek - Eagle Branch	4,698

ric seven, percent omnivores, resulted in a model with both fixed effects significant (Table 2). Pair-wise comparisons indicated that sites downstream of Muncie are significantly greater than sites within (t(23) = -2.79, P < 0.001) or upstream of Muncie (t(23) = -6.05, P < 0.001). The percent omnivore metric evaluates the intermediate to low categories of environmental quality (Simon & Dufour 1997). The estimated least squared adjusted means are 10% (± 0.006 SE) downstream, 7% (± 0.003 SE) within, and 8% (± 0.021 SE) upstream. Therefore, while a significant difference exists the percent omnivores remains relatively low. The results suggest that both habitat and urbanization pressures are related to a higher percentage of omnivores while the actual percentage remains below a level for concern. However, if a noticeable (and significant) increase can be detected with only the influence of Muncie the combined influences of other municipalities downstream in addition to Muncie likely compound the effects.

While the total IBI scores were not significantly related to their reference to city limits at the P < 0.05 level, it was significant at the P < 0.10 level (P = 0.087). Pair-wise comparisons of this model suggest sites downstream of Muncie are significantly lower than sites within city limits (t(23) = 2.33, P = 0.029) but not significantly different than sites upstream (t(23) = 1.37, P = 0.184). These results suggest urbanization is imparting a marginally negative affect. The estimated least squared adjusted mean IBI score downstream of Muncie is 47.7 *good* (± 0.926 SE) while it is 50.4 *good* (± 0.645 SE) within city limits.

The analysis of White River sites in 2009 drew some different conclusions. It was found that the total number of species, number of darter species, number of sucker species, and percent carnivores were significantly different based on reference to city limits. None of which were determined to be significant in the present analysis. The discrepancies can be explained by the analysis employed. In 2009 yearly measurements were averaged by site which eliminated the ability to account for autocorrelation. The present analysis accounts for autocorrelation through the repeated measures design by incorporating each annual sample and applying a covariance structure that best fits the data. Additionally, the 2009 comparisons were made without factoring in habitat which was also found to be significantly different with reference to city limits. The present analysis incorporates habitat as a fixed effect in the model. Finally, the analysis used in 2009 was a non-parametric test which is not as conservative at detecting differences as is a parametric mixed model.

HUC-14 watershed comparisons.—Six HUC-14 watersheds were evaluated to determine differences in IBI scores (Table 3). The preferred covariance structure for the models describing IBI score and metrics are in Appendix F. Habitat (QHEI) was found to be a significant fixed effect

Table 4.—Results of mixed model to determine differences in IBI scores in reference to HUC-14 watersheds. Significance (P < 0.05) indicated by*

	F-value for Fixed Effects		
Metric	QHEI	Watershed	Inter-action
Total IBI	9.55*	7.96*	3.00*
One	8.85*	4.19*	
Two (Headwater)	4.29*	1.07	
Two (Wadeing)	4.26*	2.00	
Three (Headwater)	5.71*	2.55	
Three (Wadeing)	1.18	23.27*	
Four (Headwater)	4.02*	1.37	
Four (Wadeing)	0.42	5.12*	
Five	3.14	5.99*	2.61*
Six	14.18*	4.57*	
Seven	1.94	6.93*	4.72*
Eight	9.46*	6.53*	5.92*
Nine (Headwater)	7.54*	1.86	
Nine (Wadeing)	4.32*	14.54*	
Ten	0.06	4.36*	
Eleven	3.09	0.99	
Twelve	5.12*	5.08*	2.51*

in 11 out of the 17 models and watershed was a significant effect in 11 of the models (Table 4).

Both QHEI scores and watershed had a significant effect on total IBI scores. The interaction term was also significant indicating that the relationship between IBI and QHEI scores differ among watersheds. This outcome is not surprising since some watersheds are dominated by White River sites while others are dominated by headwater streams. As discussed in the previous section headwater streams are more susceptible to equivalent alterations than larger streams. Hence, display a different slope parameter as indicated in this model.

Pair-wise comparisons indicated the White River – York Prairie Creek (WRYPC) (Figure 11) watershed was the most unique as it was significantly different than all the other watersheds except the Jake's Creek – Eagle Branch (JCEB) watershed (Figure 12, Table 5). The WRYPC watershed primarily includes sites on York Prairie Creek which typically has the lowest scoring IBI sites. This watershed is also heavily influenced by urbanization pressures such as stormwater runoff. Likewise the JCEB watershed is also influenced by urbanization pressures. Adjusted means for the two watersheds were the lowest of the 6 analyzed (WRYPC = 30.8 ± 1.934, JCEB = 28.58 ± 1.552). Both being below the score the Indiana Department of Environmental Management considers "Impaired". However, it is important to note the adjusted mean is the estimated score after removing the influence of habitat and treating each watershed as if they all had the same quality of habitat. The raw means at these watersheds are 33.8 (WRYPC) and 29.4 (JCEB). This suggests that while habitat is playing a role in determining biological integrity, these two watersheds are notably different from the others when habitat (i.e. QHEI) is held equal among watersheds.

Similarly, the three highest scoring watersheds; White River – Buck Creek (WRBC) (Figure 13), White River – Truitt Ditch (WRTD)

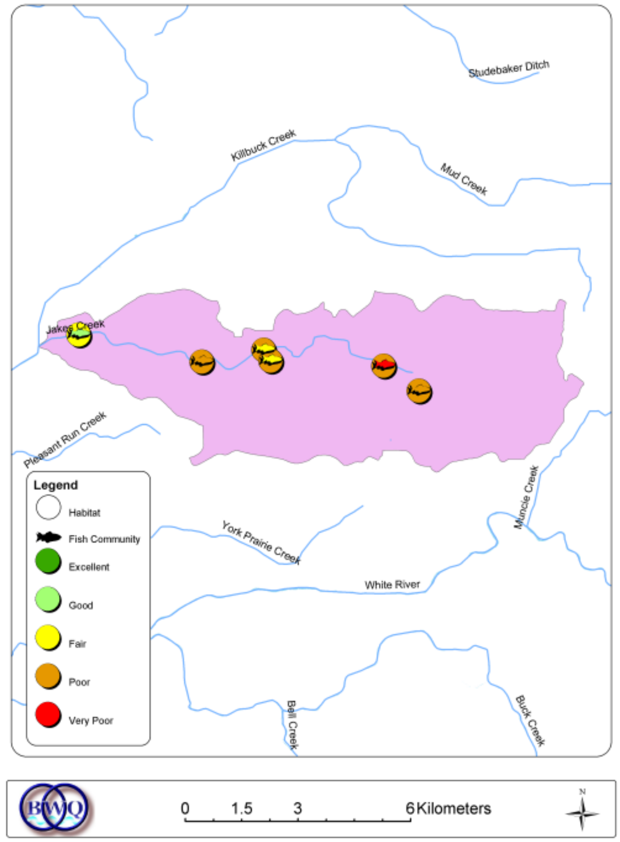

Figure 12.—Jake's Creek—Eagle Branch Watershed.

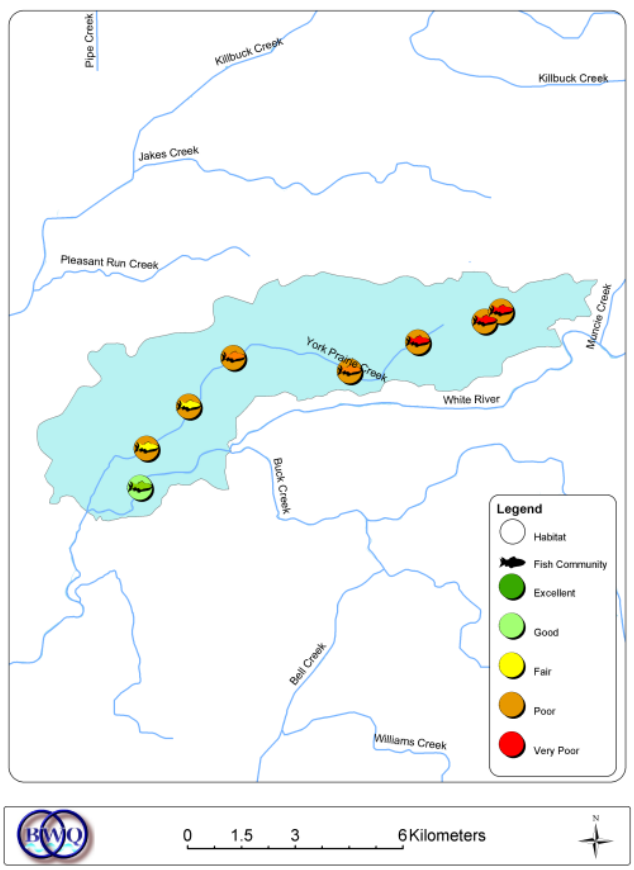

Figure 11.—White River—York Prairie Creek Watershed.

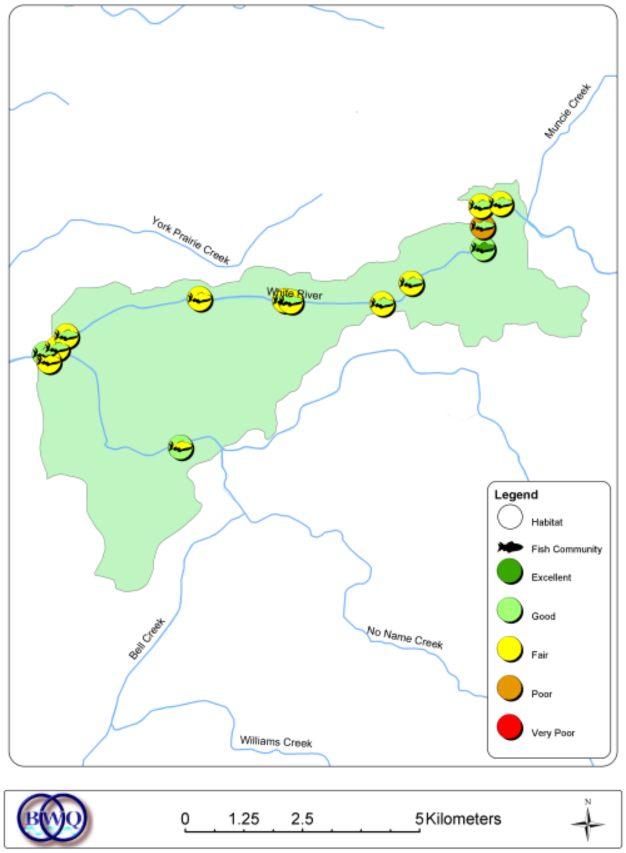

Figure 13.—White River—Buck Creek Watershed.

(Figure 14), and White River – Muncie Creek (WRMC) (Figure 15) were not significantly different from each other. These watersheds contain four or more White River sites each contributing to the similarities. The highest scoring watershed, White River – Buck Creek (raw mean IBI = 48.5) is made up of 13 sites from White River and 3 from Buck Creek. Even after accounting for habitat these sites are generally of *fair* quality (adjusted mean IBI = 44.8). The WRTD watershed has the second highest raw IBI score (45.1) and the second highest adjusted mean IBI score (42.1). This watershed contains the lowest percentage of total impervious cover (3.5%) and second lowest amount of agricultural row crop (29%). The third watershed of this group, WRMC, also has relatively low impervious cover (7.1%) but one of the highest agricultural row crop (43%) and agricultural pasture (17%).

Of the individual metric models, five resulted in a significant effect from both QHEI and watershed. Sites which did not have a significant effect from QHEI but were significantly different among watersheds suggests habitat is not having a major influence on the observed scores.

The total number of species metric (metric one) was significantly related to QHEI and significantly different among watersheds (Table 4). This metric has been considered the most powerful metric in determining biological integrity (Simon & Dufour 1997). As such you would expect it to follow a similar trend as the total IBI. This trend is apparent in this dataset. The WRYPC and Buck Creek – Macedonia Creek (BCMC) (Figure 16) watersheds had the lowest adjusted total species (10.2 and 12.3, respectively) and the WRBC, WRTD, and WRMC had the highest adjusted mean total species (19.4, 16.8, and 16.8, respectively). The WRBC, WRTD, and WRMC watersheds also had the highest IBI scores.

The percent tolerant individuals (metric six) were also significantly related to both QHEI and watersheds. This metric detects a decline in stream quality from *fair* to *poor* (Simon & Dufour

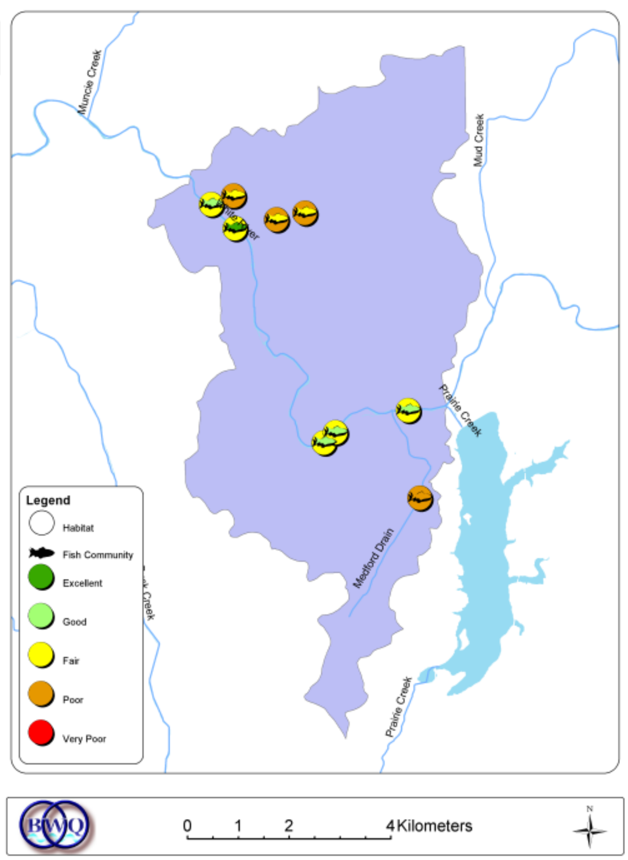

Figure 14.—White River—Truitt Ditch Watershed.

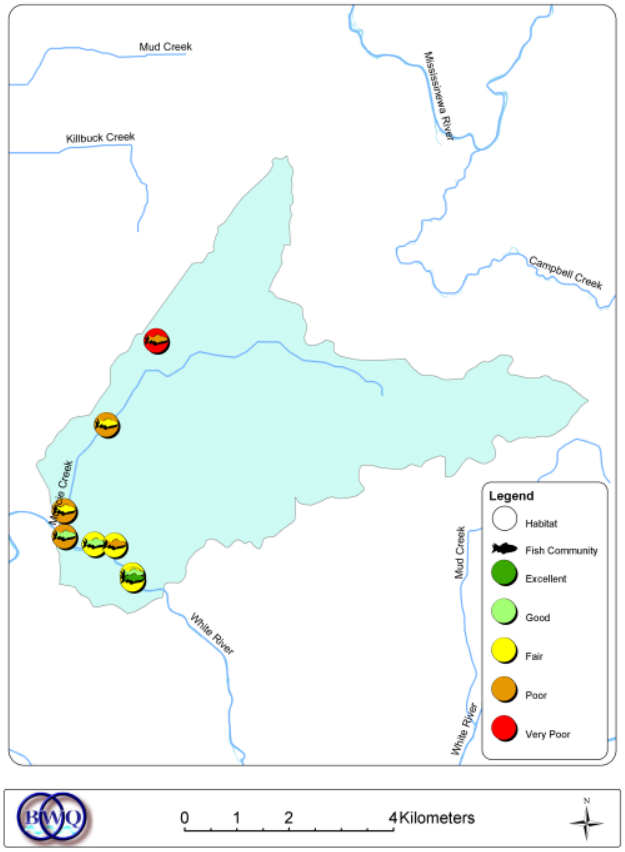

Figure 15.—White River—Muncie Creek Watershed.

insectivores decreases. The BCMC watershed had the lowest percent insectivore score (52%) and significantly differed from the WRMC and WRBC watersheds (Table 5). The WRYPC watershed had the second lowers insectivore score (58%). However, for the purposes of IBI calculations the adjusted average for both were not below the 25% threshold used to consider this metric a 1.

The percent top carnivore metric (wadeing metric nine) was also significantly related to QHEI and watersheds. This metric is only valid in wadeable streams (i.e. drainage area > 20 miles^2). This metric measures community integrity in the upper trophic levels of the fish community and it is typically only in high quality environments that upper trophic levels are able to flourish (Simon & Dufour 1997). Only five watersheds are used in this analysis. The JCEB watershed only contains sites where the drainage area is < 20 mi^2. The BCMC watershed had the lowest percent carnivore metric (< 1%) and also significantly different from the other 4 watersheds (Table 5). This water-

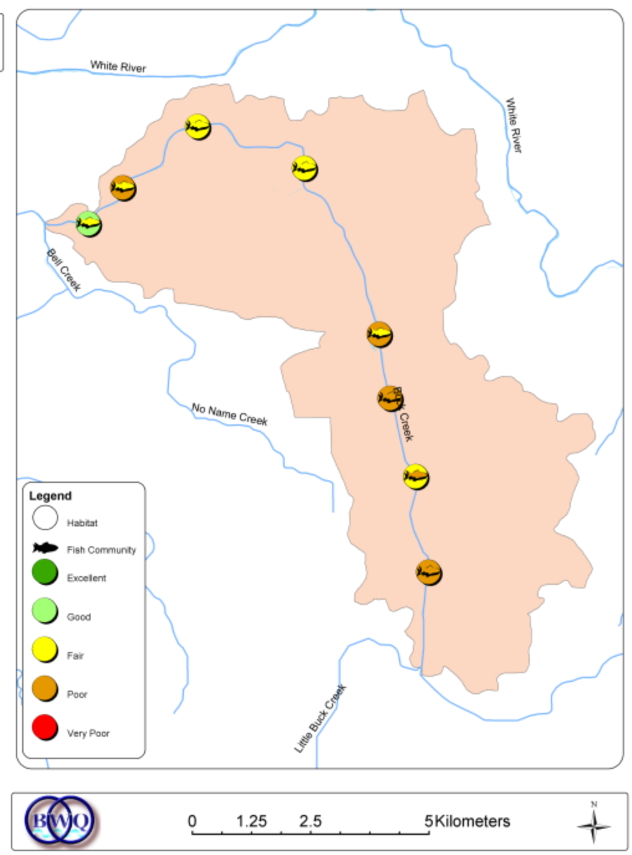

Figure 16.—Buck Creek—Macedonia Creek Watershed.

1997). Species included in this metric are those tolerant to thermal loadings, siltation, habitat degradation, and certain toxins (Ohio EPA 1989). The WRYPC and BCMC watershed had the highest adjusted mean percent tolerant metric (50% and 49% respectively) and were significantly different than the other 4 watersheds (Table 5). Metric scores reflect the poor habitat quality of WRYPC and the relatively highest amount of agricultural activity in the BCMC watershed (total 70%).

The percent insectivores (metric eight) model included a significant effect from QHEI, watershed, and the interaction of QHEI and watershed. Indicating QHEI has a different effect on IBI scores depending on watershed. This suggests other limiting factors that affect this metric are present such as stormwater runoff. The percent insectivore metric reflects the condition of the benthic macroinvertebrates that comprises the primary food base (Simon & Dufour 1997). This metric is inversely related to environmental degradation, that is as degradation increases the percent

Table 5.—Pair-wise comparisons of HUC-14 watersheds. “X” indicates significant pair-wise difference in total IBI scores and individual metrics between watersheds.

	White River - Truitt Ditch	White River - Muncie Creek	Buck Creek - Macedonia Creek	White River - Buck Creek	White River - York Prairie Cree	Jake's Creek - Eagle Branch
Total IBI Score						
White River - Truitt Ditch			X		X	X
White River - Muncie Creek			X		X	X
Buck Creek - Macedonia Creek	X	X		X	X	
White River - Buck Creek			X		X	X
White River - York Prairie Creek	X	X	X	X		
Jake's Creek - Eagle Branch	X	X		X		
% Tolerant Metric (Metric 6)						
White River - Truitt Ditch			X		X	
White River - Muncie Creek			X		X	
Buck Creek - Macedonia Creek	X	X		X		
White River - Buck Creek			X		X	
White River - York Prairie Creek	X	X		X		
Jake's Creek - Eagle Branch						
% Insectivore Metric (Metric 8)						
White River - Truitt Ditch				X		
White River - Muncie Creek			X			
Buck Creek - Macedonia Creek		X		X		
White River - Buck Creek	X		X		X	
White River - York Prairie Creek				X		
Jake's Creek - Eagle Branch						
% Carnivore Metric (Metric 9 wadeable only)						
White River - Truitt Ditch			X			
White River - Muncie Creek			X			
Buck Creek - Macedonia Creek	X	X		X	X	
White River - Buck Creek			X			
White River - York Prairie Creek			X			
Relative Number of Individuals (Metric 12)						
White River - Truitt Ditch					X	
White River - Muncie Creek					X	
Buck Creek - Macedonia Creek						
White River - Buck Creek					X	X
White River - York Prairie Creek	X	X		X		X
Jake's Creek - Eagle Branch				X	X	

shed is primarily Buck Creek sites and does not include any from White Rive. This suggests the cool water habitat is not conducive for a large population of carnivores.

The relative number of individuals (metric twelve) model included a significant effect from QHEI, watershed, and the interaction of QHEI and watershed. The same conclusions can be drawn from when the interaction variable was significant in the percent insectivore model. That is QHEI has a different effect on the relative # of individuals metric depending on watershed.

Ecoregional comparisons.— Underlying site-specific habitat variability is the broader effect of ecoregion differences (USGS 2007). Ecoregions are those areas with generally similar ecosystems. They can be described by characteristics including wildlife, physiography, geology, soil, climate and land use. Ecoregions have four levels of classification, from Level I to Level IV, with Level I encompassing the broadest description and Level IV being the most specific. Delaware County and all of the Upper West Fork White River lie within the Eastern Corn Belt Plains, a Level III ecoregion delineation. Within the Level IV delineation, three separate ecoregions can be found in Delaware County (Figure 17).

North of White River is the Clayey High Lime Till Plains (CHLTP), distinguished by turbid, low gradient streams that cross less productive, poorly-drained soils. Within Delaware and Randolph County, this ecoregion includes the Mississinewa River watershed and many smaller tributaries of White River. Most of the samples from this area were taken from headwater streams, with a small number coming from the wadeable Mississinewa River. Biotic integrity and habitat scores were *poor* at most sites sampled in this ecoregion. The mean IBI score was 34.0 *poor*, and the mean QHEI score was 47.5 *poor*. The most abundant taxa by number were Cyprinids (51%) followed by Catostomidae (22%), Centrarchidae (11%), and Percidae 9% (Figure 18). The bluntnose minnow *Pimephales notatus* was the dominant species by number (25%) and golden redhorse *Moxostoma erythrurum* (32%) were the dominant species by weight.

Through the middle of the county and bordering nearly the entire length of White River is the Loamy High Lime Till Plains (LHLTP). Soils here are typically better drained than those of the previous ecoregion and have slightly higher gradients. Mean IBI score for this region was 44.6 *fair* and the mean QHEI score was 62.3 *fair*. Sample site selection within the LHLTP was biased towards White River due to its proportional presence within the ecoregion. Cyprinids were the dominant family (41%) (Figure 18). Similar to the CHLTP, bluntnose minnow was the dominant species by number (13%) followed by golden redhorse (13%), spotfin shiner *Cyprinella spiloptera* (9%) and rock bass (6%). Golden redhorse were the dominant species by weight (33%) followed by common carp *Cyprinus carpio* (16%).

Further south encompassing most of Buck Creek and the Prairie Creek subwatershed is the Whitewater Interlobate Area (WIA). The coarse-bottomed streams in this region have moderate gradients and are supported by abundant ground water supplies leading to noticeably cooler water temperatures. The cooler temperatures have a discernable effect on the composition of fish communities in this ecoregion and on IBI scores. The mean IBI score from this region was a 39.9 *fair* and the mean QHEI score was a 60.1 *fair*. Centrarchidae was the dominant family (32%) followed by Cyprinidae (26%), Catostomidae (17%), and Cottidae (16%) (Figure 18). Green sunfish was the dominant taxon by number (30%) followed by mottled sculpin *Cottus bairdi* (16%) and creek chub *Semotilus atromaculatus* (14%). White suckers *Catostomus commersonii* were the dominant species by weight (37%) followed by creek chubs (15%) and northern hog suckers *Hypentelium nigricans* (12%). The thermal regime of Buck Creek is indicative of a coolwater stream (Conrad 2005). Therefore, the fish community is biased towards species that prefer coolwater and the Indiana IBI is not calibrated to adequately represent a coolwater fish community. This feature of Buck Creek has led to an artificially low IBI for the region. Catostomidae has typically been the dominant family in this ecoregion. As mentioned in a prevision section an unusually high abundance of green sunfish was collected in 2010. Green sunfish are considered a tolerant species and indicate degraded conditions. Sites which produced higher than normal green sunfish will be monitored in the coming years.

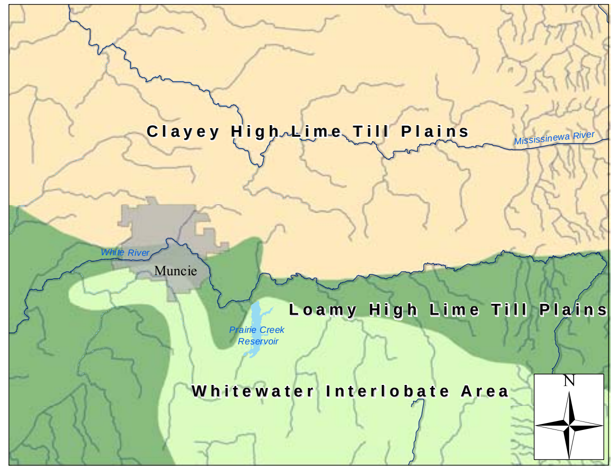

Figure 17.—Level IV ecoregions of Delaware County (USGS 2007).

The three Level IV ecoregions within Delaware County have significantly different IBI (Kruskal-Wallis; X^2 = 10.64, DF = 2, P = 0.005) and QHEI scores (Kruskal-Wallis; X^2 = 8.93, DF = 2, P = 0.014). The LHLTP has the highest mean IBI score (46.1 ± 1.5 SE) and the highest mean QHEI score (62.4 ± 1.7) while the CHLTP has the lowest mean IBI score (39.2 ± 2.6) and the lowest mean QHEI score (51.2 ± 3.6). Seven IBI metrics and 4 QHEI metrics were significantly different among ecoregions (Table 6). Three IBI metrics, the number of sunfish species (3), the number of sucker species (4), and percent individual top carnivores (9), were only significant for wading sites. Their corresponding headwater metrics were not significant. Metric 3 and4 differences reflect the dominance of Buck Creek sites in the WIA where the coolwater regime tends to favor sculpins over darters and tend to have a more diverse sucker assemblage. In contrast metric 9 differences reflect the dominance of White River sites in the LHTP. Due to its size, White River is more conducive to higher abundances of top carnivores particularly smallmouth bass and rock bass. Similarly differences in metrics 1, 5, and 10 are due in large part to White River being the predominant stream sampled in the LHLTP. These metrics are calibrated to reflect a positive relationship with drainage area. For example, collecting 10 species at a

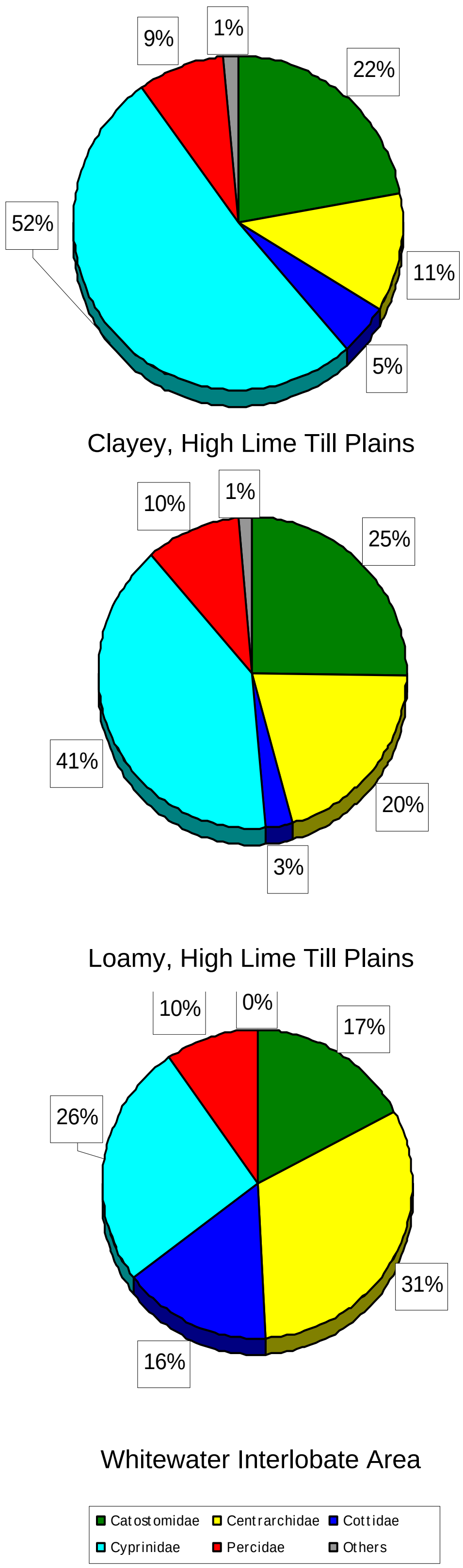

Figure 18.—Proportion of family level taxa present in each Level IV ecoregion

site with a drainage area of 10 mi^2 would yield an IBI metric rating of 3 while the same number of species at a site with a drainage area of 1000 mi^2 would yield an IBI metric rating of 1. The remaining metrics that were significantly different does not show the same relationship with drainage area. Metric 6, percent tolerant individuals were highest in the WIA (59.14) and CHLTP (49.21). This metric detects a decline in stream quality from *fair* to *poor* (Simon & Dufour 1997). The differences are likely due to poor habitat, stormwater, and agricultural pressures at the headwater streams in these ecoregions. Similarly metric 10, percent simple lithophilic spawners, reflects pressure from poor habitat, stormwater, and agricultural pressures. Lithophilic spawners require clean gravel or cobble for successful reproduction and have been shown to have a negative relationship with increased siltation (Berkman and Rabeni 1987). Siltation originates from stream bank erosion and row crop agriculture brought on by poor riparian zone practices.

Four QHEI metrics were significantly different among ecoregions (Table 6). Overall the CHLTP has the lowest QHEI score on average (47.5 ± 4.3) followed by the WIA (60.1 ± 1.5) and the LHLTP (62.3 ± 1.8). Of the significant metrics, Cover, Channel, Pool/Current, and Gradient Scores were lowest in the CHLTP. It is interesting that the Riparian Scores were not significantly different and the median values were the same for all three ecoregions. Considering the difference in lithophilic spawners you would assume varying degrees of Riparian Scores. This is explained by the difference in Substrate Scores. Lithiphiles need both high quality substrate and low to moderate amount of silt. Therefore it is concluded that while the Riparian Scores are relatively low and act as a negative influence on the fish communities the difference in Substrate Scores are driving the observed differences in lithophilic spawners.

Smallmouth Bass population estimate.—A total of 197 smallmouth bass was collected. Total lengths ranged from 99 to 457mm. Stream wide abundance of smallmouth bass in Muncie city limits was estimated at 670 fish per mile (95% credible intervals = 301 & 1,333). This equates to approximately one smallmouth bass > age-1 every 8 linear feet of White River. The site located near Ball Road (WHI-316.8) had the high-

Table 6.—IBI and QHEI metric median values by ecoregion from samples collected in 2010. * indicates significant differences based on Kruskal-Wallis Test

	Ecoregion		
	CHLTP	LHLTP	WIA
IBI Metrics			
ONE*	14.00	20.00	15.00
TWO (HEAD)	3.00	2.00	2.00
TWO (WADE)	5.00	5.00	3.00
THREE (HEAD)	0.00	4.22	14.51
THREE (WADE)*	4.00	4.00	2.00
FOUR (HEAD)	5.00	4.00	3.00
FOUR (WADE)*	5.00	4.00	1.19
FIVE*	3.50	9.00	4.00
SIX*	44.86	25.39	60.87
SEVEN	20.94	11.24	12.80
EIGHT	63.18	73.01	69.68
NINE (HEAD)	63.68	37.35	77.65
NINE (WADE)*	7.43	10.44	1.19
TEN*	11.33	22.98	25.03
ELEVEN	0.00	0.00	0.00
TWELVE	199.00	315.20	256.00
QHEI Metrics			
Substrate Score	12.0	16.0	14.0
Cover Score*	13.5	15.0	15.0
Channel Score*	8.0	10.0	9.0
Riparian Score	3.9	4.0	4.0
Pool/Current Score*	7.5	9.0	9.0
Riffle/Run Score	3.0	3.0	3.0
Gradient Score*	6.0	8.0	6.0

est estimated abundance (767, 95% CI = 591, 984) (Figure 19). The site located in West Side Park had the lowest estimated abundance (618, 95% CI = 113, 1,637).

Stream wide, the smallmouth bass PSD was 46. This is interpreted as 46% of the fish that are longer than 178 mm are also longer than 279 mm. The RSD-305 was 24 and RSD-356 was 13. These values indicate there is a relatively high abundance of individuals in the 279 to 305 mm length range. The site at Ball Road had the highest PSD of 57 and the site at West Side Park had a PSD of 41.

White River supports a relatively large population of smallmouth bass. Additionally, the population has many individuals that are of preferred size suggesting angling for this species will be above average for several years. Population estimates for smallmouth bass will continue in 2011. Additional sites will be added upstream and downstream of Muncie City limits to evaluate if there are any differences in their population due to urbanization.

DISCUSSION

Despite the presence of a wide range of negative human impacts, the overall health of the fish communities within the WFWR in and around Muncie is *good*. While some minor differences were identified, namely the slight drop in total IBI scores downstream of Muncie, White River meets the goal of maintaining good biological integrity. The stability of White River is due in large part to the strict permitting efforts of point source outfalls through the National Pollutant Discharge Elimination Systems. Muncie's Long Term Control Plan (LTCP) specifies a 96% reduction in CSO discharge and the eventual elimination of combined sewers. Additionally, the city of Muncie's effort to reduce the amount of road salt should have a positive influence on fish communities (Keiper 2009). This, together with the enhanced efficiency of industrial pretreatment facilities and the improvement of Water Pollution Control Facility effluents will continue to improve biological integrity within White River.

The presence of dams or impoundments typically has noticeable negative effects on water quality (Santucci et al. 2005); however, the five dams located along White River maintain uncommonly high IBI scores. Dams have the tendency to trap sediment, increase water temperatures, decrease dissolved oxygen, and inhibit breakdown of background pollutants such as ammonia (Baxter 1977). Their presence blocks fish passage and creates lentic habitats unsuitable for rheophilic (river dependent) species (Beasley & Hightower 2000). In spite of these chemical and physical challenges, integrity of fish communities above Muncie's dams remains strong.

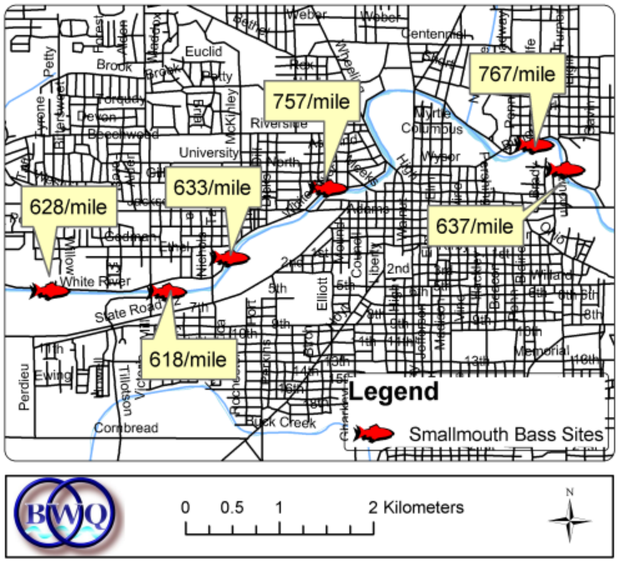

Figure 19.—Smallmouth bass population estimate sites. Numbers in balloons are the estimated number of fish per mile based on site specific samples.

In contrast to White River, its tributaries within Delaware County have consistently *poor* biological integrity ratings. Often, small streams and creeks are not maintained with consideration to water quality and aquatic life. Channelized, dredged, and denuded of riparian vegetation, they have been engineered for the sole purpose of rapidly draining water. Fish communities within these types of streams are dominated by pollution tolerant species. Under these conditions, biological integrity is often irretrievable (Yoder et al. 2000).

The watersheds in Delaware County show distinct differences in the fish community. As found in previous years the JCEP and WRYPC watersheds are the most impaired both biologically and physically. In contrast the three least impaired watersheds also contained predominantly White River sites and few tributary sites. Urbanization pressures appear to be having the most negative impact on the fish communities in the watersheds analyzed. While both Jake's Creek and York Prairie Creek are in the most impaired watersheds IBI scores on both creaks increase downstream outside of city limits.

Underlying ecoregion characteristics have led to a differentiation in habitat and fish communities. The CHLTP is described as having less productive soil with turbid, low gradient streams. These characteristics have led to more artificial drainage and clear cutting of the stream riparian zone to increase drainage efficiency, compounding anthropogenic influences on the fish communities. In contrast, the LHLTP are inherently more efficient in natural drainage reducing the amount of channelization and clear cutting that has been necessary to increase drainage. Lastly, the WIA contains distinctively cool water that is predominantly fed by groundwater. The unique thermal regime has led to a fish community that includes mottled sculpin, two species of dace, and native lampreys. When attempting to compare fish communities from these three ecoregions it is important to take into consideration the unique characteristics that are beyond the control of managers and inherently promote different fish communities.

Over the last thirty-five years, fish communities within White River in Muncie have dramatically improved; however, future improvements may depend on our ability to effect change in the tributaries which supply its water. In addition to efficiently conveying water, they simultaneously transport myriad nonpoint pollutants such as silt, fertilizers, pesticides, and many others which are discharged directly into White River. In Delaware County, these small streams account for greater than 80% of the county's stream miles and are capable of having a significant impact on the water quality of White River (Lowe & Likens 2005; Alexander et al. 2007). For example, effects of agricultural related run-off and stream bank erosion were found in the number of sucker species metric of the IBI. Often, the use of streams as drainage ditches is viewed as directly conflicting with the ability to support ecological integrity, but simple methods exist which can have dramatic improvements on water quality while still preserving the primary function of the stream. Headwater sites that typically receive *good* ratings, such as those along Stoney Creek, are bordered by wooded riparian zones, while those that typically receive *poor* ratings, such as those on Killbuck Creek and Mud Creek, were not. Streams bordered by a woody buffer strip 10 m wide may reduce the phosphorous load by 95% (Vought et al. 1995). Simpler vegetated borders such as filter strips and grassed waterways also provide significant benefits to water quality. They trap soil that would otherwise suffocate aquatic life and protect the natural structure and function of fish habitats. In addition to benefiting water quality, they can also increase farming profits by diverting efforts away from the naturally low-yield areas of buffer zones. Filter strips also supply increased access to fields, more forage for cattle, and improved aesthetics.

Landowners that wish to implement riparian buffer strips can acquire funding through various programs from the Natural Resources Conservation Service (NRCS). The Farm Bill which funds these projects has been highly successfully. For example, the Wetlands Reserve Program alone has resulted in a total of 9,951 projects protecting 1,899,979 acres (NRCS 2004). Landowners are encouraged to contact their local NRCS office for more details on each program and information on how to apply. Additionally, state allocated 319 grants award money to counties to educate and involve local citizens in improving their watersheds. The BWQ assisted the Delaware County Soil and Water Conservation District through their White River Watershed Project that has focused its attention on NPS pollution within subwatersheds containing tributaries of White River. Future integrity of the fish community could be drastically affected by how we address these issues.

In 2011, the BWQ plans to continue sampling baseline sites to assess habitat and biological integrity of White River and its tributaries. As it has for the past thirty years, the BWQ will continue to work with industries and private citizens to see that Muncie continues to be a leader in water quality management by insuring that the resources of the White River remain healthy for the people of Muncie and Indiana.

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Appendix A-1: List of Species Collected From 2004-2010

Petromyzontidae (lampreys)

Lampetra aepyptera	least brook lamprey

Clupeidae (herrings)

Dorosoma cepedianum	gizzard shad

Cyprinidae (minnows)

Pimephales notatus	bluntnose minnow
Campostoma anomalum	central stoneroller
Semotilus atromaculatus	creek chub
Notropis ludibundus	sand shiner
Notropis rubellus	rosyface shiner
Ericymba buccata	silverjaw minnow
Cyprinella spiloptera	spotfin shiner
Luxilus chrysocephalus	striped shiner
Rhinichthys atratulus	blacknose dace
Notropis photogenis	silver shiner
Notropis volucellus	mimic shiner
Cyprinus carpio	common carp
Lythrurus umbratilis	redfin shiner
Cyprinella whipplei	steelcolor shiner
Phenacobius mirabilis	suckermouth minnow
Nocomis biguttatus	hornyhead chub
Nocomis micropogon	river chub
Carassius auratus	goldfish
Pimephales promelas	fathead minnow
Phoxinus erythrogaster	southern redbelly dace
Notemigonus crysoleucas	golden shiner
Hybopsis amblops	bigeye chub
Notropis blennius	river shiner
Ctenopharyngodon idella	grass carp

Catostomidae (suckers)

Moxostoma erythrurum	golden redhorse
Catostomus commersonii	white sucker
Hypentelium nigricans	northern hog sucker
Minytrema melanops	spotted sucker
Carpiodes cyprinus	quillback carpsucker
Moxostoma duquesnei	black redhorse
Carpiodes velifer	highfin carpsucker
Erimyzon oblongus	creek chubsucker
Ictiobus bubalus	smallmouth buffalo

Ictaluridae (catfishes and bullheads)

Ameiurus natalis	yellow bullhead
Noturus gyrinus	tadpole madtom
Noturus flavus	stonecat
Ictalurus punctatus	channel catfish
Ameiurus melas	black bullhead
Ameiurus nebulosus	brown bullhead
Noturus miurus	brindled madtom
Pylodictis olivaris	flathead catfish

Esocidae (pikes)

Esox americanus	redfin pickerel

Aphredoderidae (pirate perches)

Aphredoderus sayanus	pirate perch

Fundulidae (killfishes)

Fundulus notatus	blackstripe topmin.

Peociliidae (livebearers)

Gambusia affinis	mosquitofish

Atherinidae (silversides)

Labidesthes sicculus	brook silverside

Cottidae (sculpins)

Cottus bairdi	mottled sculpin

Percichthyidae (temperate basses)

Morone chrysops	white bass

Centrarchidae (sunfishes)

Lepomis cyanellus	green sunfish
Ambloplites rupestris	rock bass
Lepomis megalotis	longear sunfish
Lepomis macrochirus	bluegill
Micropterus dolomieu	smallmouth bass
Micropterus salmoides	largemouth bass
Pomoxis nigromaculatus	black crappie
Lepomis microlophus	redear sunfish
Pomoxis annularis	white crappie
Lepomis humilis	orangespotted sunfish
Lepomis gibbosus	pumpkinseed
Lepomis spp.	hybrid sunfish
Centrarchidae	sunfish Family
Micropterus punctatus	spotted bass

Percidae (perches)

Etheostoma nigrum	johnny darter
Etheostoma blennioides	greenside darter
Etheostoma spectabile	orangethroat darter
Etheostoma caeruleum	rainbow darter
Percina caprodes	logperch
Percina maculata	blackside darter
Percina phoxocephala	slenderhead darter
Etheostoma flabellare	fantail darter
Perca flavescens	yellow perch
Sander vitreus	walleye

Sciaenidae (drums)

Aplodinotus grunniens	freshwater drum

Appendix B-1: IBI Metrics

Site Type		Abbreviated in summary sheets as:
Wading Site Metrics:		
One:	Total number of species	# Total Species
Two:	Total number of darter species	# Darter Species
Three:	Number of sunfish species	# Sunfish Species
Four:	Number of sucker species	# Sucker Species
Five:	Number of sensitive species	# Sensitive Species
Six:	Percent of individual tolerants	% Tolerant
Seven:	Percent of individual omnivores	% Omnivores
Eight:	Percent of individual insectivores	% Insectivores
Nine:	Percent of individual top carnivores	% Top Carnivores
Ten:	Percent of individual simple lithophils	% Simple Lithophils
Eleven:	Percent of individuals with deformities, eroded fins, lesions, or tumors	% DELT
Twelve:	Relative number of individual fish per 15 times the wetted width	Relative Number

Headwater Site Metrics:		
One:	Total number of species	# Total Species
Two:	Total number of darter, madtom, and sculpin species	# Darter/Madtom/Sculpin
Three:	Percent of headwater species	% Headwater Species
Four:	Number of minnow species	# Minnow Species
Five:	Number of sensitive species	# Sensitive Species
Six:	Percent of individual tolerants	% Tolerant
Seven:	Percent of individual omnivores	% Omnivores
Eight:	Percent of individual insectivores	% Insectivores
Nine:	Percent of individual pioneering	% Pioneering
Ten:	Percent of individual simple lithophils	% Simple Lithophils
Eleven:	Percent of individuals with deformities, eroded fins, lesions, or tumors	% DELT
Twelve:	Relative number of individual fish per 15 times the wetted width	Relative Number

[NOTE: Refer to Simon and Dufour (1997) for exact calculation of metrics and description of guilds]

Appendix B-2: IBI, MIwb, and QHEI Ratings

Wading Sites:			
IBI Score	MIwb Score	QHEI Score	Rating
53-60	> 9.4	90-100	Excellent
45-52	8.3-9.3	71-89.9	Good
35-44	5.9-8.2	52-70.9	Fair
23-34	4.5-5.8	27-51.9	Poor
12-22	< 4.5	0-26.9	Very poor
<12	0		NO FISH FOUND

Headwater Sites:			
IBI Score	MIwb Score	QHEI Score	Rating
53-60	Not applicable to headwater sites	90-100	Excellent
45-52		71-89.9	Good
35-44		52-70.9	Fair
23-34		27-51.9	Poor
12-22		0-26.9	Very poor
<12			NO FISH FOUND

Appendix B-3: Breakdown of Index Scores from 2010

Sample Site	River Mile	Date Sampled		# Total Species	# Darter/ Madtom/ Sculpin	% Headwater Sp.	# Minnow Species	# Sensitive Species	% Tolerant	% Omnivores	% Insectivores	% Pioneering	% Simple Lithophils	% DELT	Relative Number	IBI Score	QHEI Score	MIwb Score
			IBI METRICS - HEADWATER SITES															
Huffman Creek C.R. 600 S.	0.2	7/23/10	Calc.	8	2	14.51	3	0	40.78	7.45	20.39	77.65	38.04	0.00	255.00	40	59.0	NA
			Score	3	5	5	3	1	3	5	1	1	3	5	5			
Medford Drain Burlington Dr.	1.3	6/29/10	Calc.	5	3	58.06	2	0	58.06	0.00	41.94	41.94	83.87	0.00	41.33	26	33.0	NA
			Score	3	5	5	3	1	1	1	1	3	1	1	1			
Greenfarm Ditch W. Riggin Rd.	0.3	6/17/10	Calc.	10	2	0.00	3	0	44.83	3.45	34.48	79.31	3.45	0.00	29.00	22	43.0	NA
			Score	3	5	1	3	1	3	1	1	1	1	1	1			
Greenfarm Ditch Wheeling Ave.	0.6	7/1/10	Calc.	12	3	0.65	4	0	67.97	11.76	62.75	87.58	5.88	0.00	153.00	38	28.0	NA
			Score	5	5	1	5	1	1	5	5	1	1	5	3			
Hamilton Ditch C.R. 300 N.	0.2	6/29/10	Calc.	14	2	0.00	5	2	81.13	71.70	17.45	63.68	10.85	0.00	212.00	34	19.0	NA
			Score	5	5	1	5	3	1	1	1	1	1	5	5			
Truitt Ditch C.R. 300 E.	1.6	6/17/10	Calc.	11	2	2.94	5	0	17.65	10.29	75.00	26.47	7.35	0.00	68.00	42	56.0	NA
			Score	5	5	1	5	1	5	5	5	3	1	5	1			
York Prairie Creek C.R. 400 W.	6.3	6/30/10	Calc.	11	3	3.45	3	1	43.10	8.62	34.48	68.97	8.62	0.00	58.00	34	53.0	NA
			Score	5	5	1	3	1	3	5	3	1	1	5	1			
York Prairie Creek Maddox Dr.	9.0	6/29/10	Calc.	8	1	0.00	3	1	75.00	40.00	40.00	65.00	2.50	0.00	33.80	18	46.0	NA
			Score	3	3	1	3	1	1	1	1	1	1	1	1			
Eagle Branch C.R. 350 N.	0.2	6/25/10	Calc.	12	3	22.22	4	0	49.21	7.94	74.60	55.56	7.94	0.00	63.00	40	41.0	NA
			Score	5	5	5	3	1	3	5	5	1	1	5	1			
Holt Ditch Ball Rd.	0.1	6/29/10	Calc.	18	3	1.55	8	7	29.53	29.02	66.32	27.46	5.18	0.00	165.43	44	64.5	NA
			Score	5	5	1	5	5	3	3	5	3	1	5	3			
Muncie Creek McGalliard Rd.	1.4	7/1/10	Calc.	19	2	0.00	8	4	68.13	64.29	32.42	65.38	26.37	0.00	364.00	38	33.0	NA
			Score	5	3	1	5	5	1	1	3	1	3	5	5			
Truitt Ditch Butterfield Rd.	0.8	7/12/10	Calc.	13	2	21.14	5	1	62.60	38.21	37.40	32.52	42.28	0.00	123.00	40	62.5	NA
			Score	5	3	3	5	1	1	3	3	3	5	5	3			
Truitt Ditch Highway 3	1.3	6/25/10	Calc.	10	2	17.65	4	0	38.24	20.59	23.53	60.00	18.24	0.00	170.00	32	64.0	NA
			Score	3	3	3	3	1	3	5	1	1	1	5	3			
Jake's Creek Everett Rd. Lift Station (A)	6.6	6/17/10	Calc.	15	2	0.00	4	1	44.90	14.29	75.51	42.86	4.08	0.00	49.00	24	44.8	NA
			Score	5	3	1	3	1	3	1	1	3	1	1	1			
No Name Creek S.R. 67 S.	1.1	6/25/10	Calc.	8	1	62.07	4	0	41.38	13.79	62.07	31.03	13.79	3.45	29.00	24	69.0	NA
			Score	3	1	5	3	1	3	1	1	3	1	1	1			
Muncie Creek McCulloch Park	0.1	7/2/10	Calc.	11	3	0.00	5	3	43.31	35.03	23.57	77.07	9.55	0.00	128.45	30	61.5	NA
			Score	3	3	1	3	3	3	3	1	1	1	5	3			
Muncie Creek Highland Ave.	0.3	6/18/10	Calc.	14	3	0.00	3	6	47.70	45.07	52.63	47.70	43.09	0.00	304.00	46	45.0	NA
			Score	5	3	1	3	5	3	3	5	3	5	5	5			
Jake's Creek C.R. 500 W.	3.0	6/30/10	Calc.	13	3	30.99	6	0	58.45	25.35	44.37	49.30	16.90	0.70	142.00	32	65.0	NA
			Score	3	3	3	5	1	1	3	3	3	1	3	3			
Killbuck Creek Wheeling Ave.	20.1	7/1/10	Calc.	22	6	6.40	9	4	52.83	47.90	33.46	71.85	3.66	0.00	547.00	38	58.5	NA
			Score	5	5	1	5	3	1	3	3	1	1	5	5			

Appendix B-3 cont'd:

Sample Site	River Mile	Date Sampled		# Total Species	# Darter/ Madtom/ Sculpin	% Head-water Sp.	# Min-now Spe-cies	# Sensi-tive Spe-cies	% Toler-ant	% Omni-vores	% Insecti-vores	% Pio-neering	% Simple Lithophils	% DELT	Relative Number	IBI Score	QHEI Score	MIwb Score
			IBI METRICS - HEADWATER SITES															
Prairie Creek Spillway	0.0	7/8/10	Calc.	23	1	0.00	4	5	10.32	5.41	90.91	4.42	37.84	0.00	162.80	46	65.0	NA
Prairie Creek Spillway			Score	5	1	1	3	5	5	5	5	5	3	5	3			
York Prairie Creek	1.7	6/18/10	Calc.	19	3	4.22	7	6	47.59	36.75	30.72	37.35	36.75	0.00	166.00	42	45.0	NA
C.R. 750 W.			Score	5	3	1	5	5	3	3	3	3	3	5	3			
York Prairie Creek	2.8	6/18/10	Calc.	19	5	3.75	9	5	17.19	11.25	22.19	73.75	15.31	0.00	320.00	44	40.0	NA
C.R. 50 S.			Score	5	5	1	5	5	5	5	1	1	1	5	5			
Jake's Creek	0.7	6/7/10	Calc.	14	5	74.73	6	3	16.13	5.91	82.80	16.67	12.90	0.00	186.00	48	68.5	NA
C.R. 700 W.			Score	3	5	5	3	3	5	5	5	5	1	5	3			

Appendix B-3 cont'd:

Sample Site	River Mile	Date Sampled	IBI METRICS - WADING SITES	# Total Species	# Darter Species	# Sunfish Species	# Sucker Species	# Sensitive Species	% Tolerant	% Omnivores	% Insectivores	% Top Carnivores	% Simple Lithophils	% DELT	Relative Number	IBI Score	QHEI Score	MIwb Score
Buck Creek C.R. 700 S.	13.8	7/13/10	Calc.	12	2	2	2	1	74.85	22.70	28.83	1.23	33.74	0.00	326.00	36	60.0	5.7
			Score	3	3	3	3	1	1	5	3	1	3	5	5			
Buck Creek C.R. 800 S.	14.9	7/15/10	Calc.	15	3	2	2	3	71.98	14.01	34.24	0.39	17.90	0.00	257.00	36	60.0	5.8
			Score	3	5	3	3	3	1	5	3	1	1	5	3			
Buck Creek C.R. 578 S.	12.5	7/21/10	Calc.	8	3	1	1	1	66.67	24.07	41.20	0.00	37.50	0.00	216.00	30	52.0	NA
			Score	1	5	1	1	1	1	5	3	1	3	5	3			
Buck Creek C.R. 400 S.	10.5	6/8/10	Calc.	16	3	3	2	4	58.82	29.41	52.21	0.00	40.44	0.00	136.00	40	51.0	6.2
			Score	5	5	3	3	3	1	3	5	1	5	5	1			
Buck Creek S.R. 3	11.3	7/13/10	Calc.	14	1	3	2	3	82.54	12.13	74.56	0.30	15.68	0.59	338.00	34	55.0	5.5
			Score	3	1	3	3	3	1	5	5	1	1	3	5			
Buck Creek C.R. 325 W.	4.0	7/30/10	Calc.	20	4	2	4	6	59.14	10.96	82.39	1.33	21.93	0.33	275.92	42	63.5	7.8
			Score	5	5	3	5	3	1	5	5	1	3	3	3			
Buck Creek C.R. 325 W.	4.0	7/2/10	Calc.	20	5	2	3	8	57.31	11.46	80.24	1.19	27.27	0.00	253.00	44	69.0	7.5
			Score	5	5	3	3	5	1	5	5	1	3	5	3			
Buck Creek C.R. 200 S.	5.1	7/20/10	Calc.	11	2	4	3	4	75.33	2.00	93.33	3.33	6.67	0.67	150.00	34	57.0	5.9
			Score	3	3	5	3	3	1	5	5	1	1	3	1			
Buck Creek Tillotson Ave.	5.9	7/20/10	Calc.	17	5	2	2	4	64.10	13.46	73.72	1.28	20.19	0.32	312.00	40	62.5	6.8
			Score	5	5	3	3	3	1	5	5	1	3	3	3			
Buck Creek C.R. 100 W.	7.0	7/20/10	Calc.	12	3	2	2	5	61.96	25.77	65.64	1.84	46.01	0.00	163.00	40	59.5	6.6
			Score	3	5	3	3	3	1	3	5	1	5	5	3			
Buck Creek 23rd St.	8.0	6/8/10	Calc.	15	3	2	3	5	37.11	14.47	61.64	0.00	42.77	0.00	159.00	44	64.0	7.3
			Score	3	5	3	3	3	3	5	5	1	5	5	3			
Buck Creek C.R. 300 S.	9.5	7/22/10	Calc.	23	5	3	2	7	59.78	11.49	78.57	0.47	15.37	0.00	644.00	46	58.0	7.1
			Score	5	5	5	3	5	1	5	5	1	1	5	5			
Buck Creek Morrow's Meadow	0.2	7/8/10	Calc.	17	3	1	2	6	57.08	23.18	36.05	3.00	23.18	0.00	262.13	34	66.5	7.2
			Score	3	3	1	3	3	1	5	3	1	3	5	3			
Buck Creek S.R. 32	0.5	7/16/10	Calc.	20	5	2	3	9	13.17	5.85	86.83	4.39	43.90	0.00	328.00	50	71.0	8.5
			Score	5	5	3	3	5	5	5	5	1	5	5	3			
Buck Creek C.R. 428 W.	3.1	7/22/10	Calc.	20	6	2	3	9	21.20	1.90	90.82	1.90	22.78	0.00	379.20	50	69.0	7.7
			Score	5	5	3	3	5	5	5	5	1	3	5	5			
White River Inlow Springs Rd.	323.8	7/23/10	Calc.	25	5	3	4	11	40.53	30.23	57.14	11.96	15.28	0.00	699.83	50	60.5	8.6
			Score	5	5	3	5	5	3	3	5	5	1	5	5			
White River Camp Red Wing (B)	322.2	7/19/10	Calc.	36	6	5	5	12	32.62	30.24	62.86	6.19	23.33	0.00	1102.50	52	67.0	9.4
			Score	5	5	5	5	5	3	3	5	3	3	5	5			
White River Camp Red Wing (A)	322.4	8/13/10	Calc.	20	0	3	7	8	10.82	11.26	74.89	12.12	76.62	0.00	249.48	50	66.0	8.3
			Score	3	1	3	5	5	5	5	5	5	5	5	3			
White River Water Company (B)	318.8	7/19/10	Calc.	25	6	3	3	10	32.32	30.51	57.78	11.11	13.13	0.00	1002.38	48	66.5	8.9
			Score	5	5	3	3	5	3	3	5	5	1	5	5			

Appendix B-3 cont'd:

Sample Site	River Mile	Date Sampled	IBI METRICS - WADING SITES	# Total Species	# Darter Species	# Sunfish Species	# Sucker Species	# Sensitive Species	% Tolerant	% Omnivores	% Insectivores	% Top Carnivores	% Simple Lithophils	% DELT	Relative Number	IBI Score	QHEI Score	MIwb Score
White River Water Company (A)	319.0	8/10/10	Calc.	19	1	5	7	9	4.74	8.30	77.87	13.44	71.94	0.00	311.19	52	58.5	8.5
			Score	3	1	5	5	5	5	5	5	5	5	5	3			
White River McCulloch Park	316.3	8/14/10	Calc.	13	0	0	6	6	2.63	4.14	91.35	4.51	81.20	0.00	446.88	44	43.0	7.6
			Score	3	1	1	5	3	5	5	5	1	5	5	5			
White River McCulloch Park(BelowDam)	316.3	7/30/10	Calc.	32	5	4	5	17	9.19	11.35	71.62	15.14	34.32	0.00	1110.00	58	66.5	10.0
			Score	5	5	5	5	5	5	5	5	5	3	5	5			
White River Ball Rd. (QL's)	316.8	7/7/10	Calc.	26	5	3	5	12	16.25	16.53	70.31	4.20	40.34	0.00	1160.25	54	65.0	9.4
			Score	5	5	3	5	5	5	5	5	1	5	5	5			
White River E. Jackson St. (B)	317.4	7/9/10	Calc.	31	6	4	5	14	27.45	25.49	63.73	10.59	15.49	0.20	1300.50	50	61.5	9.0
			Score	5	5	5	5	5	3	3	5	5	1	3	5			
White River E. Jackson St. (A)	317.6	8/4/10	Calc.	19	0	4	6	7	4.61	4.61	81.91	13.49	76.32	0.00	319.20	50	60.0	8.5
			Score	3	1	5	5	3	5	5	5	5	5	5	3			
White River High St. (B)	314.8	7/15/10	Calc.	31	5	5	4	14	33.79	29.34	60.98	7.07	19.35	0.00	2025.33	50	67.0	10.3
			Score	5	5	5	5	5	3	3	5	3	1	5	5			
White River High St. (C)	315.0	8/10/10	Calc.	18	0	6	6	6	10.47	8.38	83.25	7.33	75.39	0.00	269.31	48	54.5	7.8
			Score	3	1	5	5	3	5	5	5	3	5	5	3			
White River High St. (A)	315.3	8/3/10	Calc.	18	1	5	7	8	7.59	5.91	81.43	12.24	77.64	0.00	319.95	50	59.5	8.5
			Score	3	1	5	5	3	5	5	5	5	5	5	3			
White River Walnut St.	315.6	7/7/10	Calc.	27	6	4	5	12	33.71	28.76	63.62	7.43	11.81	0.19	1120.00	48	68.5	8.9
			Score	5	5	5	5	5	3	3	5	3	1	3	5			
White River MWPCF (B)	311.6	8/2/10	Calc.	17	1	5	6	8	10.11	11.24	80.52	7.12	82.77	0.00	296.37	48	62.0	8.5
			Score	3	1	5	5	3	5	5	5	3	5	5	3			
White River MWPCF (A)	311.7	7/6/10	Calc.	22	5	3	2	9	6.92	3.08	80.00	15.77	27.31	0.77	607.08	52	67.5	8.8
			Score	5	5	3	3	5	5	5	5	5	3	3	5			
White River West Side Park	313.4	7/21/10	Calc.	24	6	4	4	12	24.44	13.20	73.03	13.48	21.63	0.00	774.30	58	67.5	9.1
			Score	5	5	5	5	5	5	5	5	5	3	5	5			
White River Godman Ave.	313.8	7/6/10	Calc.	28	5	5	4	12	24.33	14.44	72.99	9.36	11.50	0.27	729.30	52	70.0	9.3
			Score	5	5	5	5	5	5	5	5	3	1	3	5			
White River C.R. 400 W.	310.7	7/14/10	Calc.	21	3	5	4	9	51.90	23.33	67.62	7.14	8.10	0.00	424.57	48	64.0	7.4
			Score	5	3	5	5	5	1	5	5	3	1	5	5			
White River C.R. 575 W. (B)	308.5	7/16/10	Calc.	18	3	4	2	9	24.73	6.04	82.42	10.44	15.93	0.00	259.35	48	72.5	8.3
			Score	3	3	5	3	5	5	5	5	5	1	5	3			
White River C.R. 575 W. (C)	308.6	7/16/10	Calc.	23	4	4	4	12	26.05	7.91	76.74	15.35	22.79	0.47	483.75	54	71.0	8.3
			Score	5	5	5	5	5	3	5	5	5	3	3	5			
White River C.R. 575 W. (A)	308.7	8/2/10	Calc.	22	1	6	7	9	10.22	8.44	79.11	8.44	65.78	0.00	236.25	52	72.5	9.2
			Score	5	1	5	5	5	5	5	5	3	5	5	3			
White River C.R. 750 W.	306.5	7/14/10	Calc.	25	6	5	1	12	28.76	11.16	76.39	10.73	14.59	0.00	524.25	50	71.0	8.4
			Score	5	5	5	1	5	3	5	5	5	1	5	5			
White River C.R. 300 S.	304.4	7/14/10	Calc.	22	5	5	3	11	20.39	11.17	73.30	9.22	42.72	0.00	706.29	56	68.0	8.7
			Score	5	5	5	3	5	5	5	5	3	5	5	5			

Appendix C-1: Correlation Tables

The following tables list the Pearson's correlation coefficient (*r*) between IBI and QHEI scores. Significant relationships (P < 0.05) are highlighted in orange.

All sites from 2004 - 2010

	IBI score	Metric #1	Metric #2	Metric #3	Metric #4	Metric #5	Metric #6	Metric #7	Metric #8	Metric #9	Metric #10	Metric #11	Metric #12
QHEI	0.72	0.38	0.34	0.55	0.26	0.68	0.59	0.61	0.60	0.35	0.41	0.34	0.49
Substrate Score	0.66	0.31	0.32	0.47	0.27	0.56	0.58	0.58	0.50	0.32	0.42	0.32	0.44
Cover Score	0.56	0.34	0.22	0.47	0.23	0.51	0.31	0.49	0.51	0.32	0.32	0.28	0.32
Channel Score	0.60	0.36	0.33	0.46	0.20	0.59	0.52	0.50	0.46	0.24	0.30	0.25	0.40
Riparian Score	0.39	0.18	0.17	0.45	-0.01	0.42	0.34	0.28	0.32	0.25	0.24	0.16	0.18
Pool/Current Score	0.65	0.37	0.33	0.45	0.21	0.64	0.46	0.53	0.58	0.27	0.32	0.31	0.50
Riffle/Run Score	0.62	0.37	0.34	0.39	0.21	0.62	0.56	0.47	0.52	0.27	0.30	0.26	0.47
Gradient Score	0.46	0.10	0.07	0.39	0.19	0.37	0.49	0.42	0.35	0.30	0.34	0.18	0.28

Appendix D-1: Change in Akaike's Information Criteria for models describing total IBI and IBI metrics based on QHEI values and reference to city limits or 14-digit HUC watershed.

	Covariance Structure				
Dependent Variable	Unstruc-tured	Autoregres-sive (1)	Compound Symmetry	Variance Compo-nents	Toeplitz
White River Only Model: IBI ~ City Limits + QHEI + Year					
Total IBI	0.0	4.9	2.8	3.0	5.5
One	21.8	16.2	0.0	17.9	6.3
Two	0.0	31.6	16.3	42.0	24.2
Three	0.0	34.4	19.5	61.0	24.8
Four	14.5	23.3	0.0	57.7	4.2
Five	16.6	26.4	0.0	36.3	6.2
Six	0.0	7.3	9.3	32.6	15.8
Seven	0.0	22.4	26.5	29.6	29.4
Eight	0.0	7.4	4.6	12.9	10.4
Nine	20.3	0.0	26.2	73.6	4.9
Ten	0.0	14.3	3.5	104.2	7.3
Eleven	*	*	32.0	0.0	*
Twelve	22.4	19.2	0.0	69.3	4.3
14-digit HUC Watershed Models: IBI ~ HUC + QHEI + Year					
Total IBI	0.0	47.3	3.5	92.2	7.6
One	9.3	51.1	0.0	111.2	0.8
Two (Headwater)	*	10.5	0.0	45.1	9.2
Two (Wading)	0.0	14.0	3.2	22.4	10.6
Three (Headwater)	*	24.9	18.1	24.2	0.0
Three(Wading)	0.0	27.2	9.1	58.2	14.0
Four (Headwater)	*	7.3	0.0	16.2	6.0
Four (Wading)	*	23.1	0.0	59.7	5.6
Five	0.0	71.3	8.7	205.8	15.1
Six	0.0	38.2	12.6	83.9	13.1
Seven	10.9	4.7	0.0	13.5	1.5
Eight	0.0	31.5	6.3	61.2	7.6
Nine (Headwater)	*	3.7	0.0	3.0	*
Nine (Wading)	6.4	4.4	10.9	107.6	0.0
Ten	17.4	27.0	0.0	127.6	4.4
Eleven	*	*	2.0	0.0	*
Twelve	3.1	18.4	2.9	116.2	0.0

Appendix E-1: Estimated covariance metrics of errors for the IBI and IBI metrics models describing White River sites.

Total IBI score, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.233	0.629	-0.672	0.261	-0.039	-0.103
2	0.233	1.000	0.412	0.171	0.470	0.306	0.242
3	0.629	0.412	1.000	-0.172	0.312	0.034	-0.293
4	-0.672	0.171	-0.172	1.000	-0.152	0.078	0.176
5	0.261	0.470	0.312	-0.152	1.000	-0.151	0.244
6	-0.039	0.306	0.034	0.078	-0.151	1.000	-0.231
7	-0.103	0.242	-0.293	0.176	0.244	-0.231	1.000

One, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.397	0.397	0.397	0.397	0.397	0.397
2	0.397	1.000	0.397	0.397	0.397	0.397	0.397
3	0.397	0.397	1.000	0.397	0.397	0.397	0.397
4	0.397	0.397	0.397	1.000	0.397	0.397	0.397
5	0.397	0.397	0.397	0.397	1.000	0.397	0.397
6	0.397	0.397	0.397	0.397	0.397	1.000	0.397
7	0.397	0.397	0.397	0.397	0.397	0.397	1.000

Two, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.565	0.498	0.449	0.229	0.586	0.678
2	0.565	1.000	0.240	0.625	0.396	0.490	0.572
3	0.498	0.240	1.000	0.464	-0.019	0.756	0.426
4	0.449	0.625	0.464	1.000	0.291	0.413	0.658
5	0.229	0.396	-0.019	0.291	1.000	0.083	0.478
6	0.586	0.490	0.756	0.413	0.083	1.000	0.650
7	0.678	0.572	0.426	0.658	0.478	0.650	1.000

Three, Unstructured

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.349	0.603	0.505	0.218	0.609	0.596
2	0.349	1.000	0.761	0.604	0.013	0.035	0.326
3	0.603	0.761	1.000	0.629	0.061	0.530	0.460
4	0.505	0.604	0.629	1.000	0.436	0.379	0.709
5	0.218	0.013	0.061	0.436	1.000	0.153	0.452
6	0.609	0.035	0.530	0.379	0.153	1.000	0.648
7	0.596	0.326	0.460	0.709	0.452	0.648	1.000

Four, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.552	0.552	0.552	0.552	0.552	0.552
2	0.552	1.000	0.552	0.552	0.552	0.552	0.552
3	0.552	0.552	1.000	0.552	0.552	0.552	0.552
4	0.552	0.552	0.552	1.000	0.552	0.552	0.552
5	0.552	0.552	0.552	0.552	1.000	0.552	0.552
6	0.552	0.552	0.552	0.552	0.552	1.000	0.552
7	0.552	0.552	0.552	0.552	0.552	0.552	1.000

Appendix E-1: Continued

Five, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.527	0.527	0.527	0.527	0.527	0.527
2	0.527	1.000	0.527	0.527	0.527	0.527	0.527
3	0.527	0.527	1.000	0.527	0.527	0.527	0.527
4	0.527	0.527	0.527	1.000	0.527	0.527	0.527
5	0.527	0.527	0.527	0.527	1.000	0.527	0.527
6	0.527	0.527	0.527	0.527	0.527	1.000	0.527
7	0.527	0.527	0.527	0.527	0.527	0.527	1.000

Six, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.001	0.001	-0.005	0.248	0.149	0.194
2	0.001	1.000	0.768	0.066	0.482	0.545	0.644
3	0.001	0.768	1.000	0.247	0.470	0.509	0.428
4	-0.005	0.066	0.247	1.000	0.321	0.457	0.363
5	0.248	0.482	0.470	0.321	1.000	0.479	0.753
6	0.149	0.545	0.509	0.457	0.479	1.000	0.776
7	0.194	0.644	0.428	0.363	0.753	0.776	1.000

Seven, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	-0.144	0.010	-0.185	-0.666	-0.108	-0.240
2	-0.144	1.000	0.789	0.269	0.530	0.372	0.735
3	0.010	0.789	1.000	0.457	0.609	0.304	0.548
4	-0.185	0.269	0.457	1.000	0.412	0.266	0.387
5	-0.666	0.530	0.609	0.412	1.000	0.352	0.687
6	-0.108	0.372	0.304	0.266	0.352	1.000	0.673
7	-0.240	0.735	0.548	0.387	0.687	0.673	1.000

Eight, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	-0.138	-0.011	0.377	0.296	0.382	0.419
2	-0.138	1.000	0.750	-0.057	0.231	0.237	0.538
3	-0.011	0.750	1.000	0.107	0.367	-0.003	0.352
4	0.377	-0.057	0.107	1.000	0.553	0.139	0.323
5	0.296	0.231	0.367	0.553	1.000	0.099	0.630
6	0.382	0.237	-0.003	0.139	0.099	1.000	0.428
7	0.419	0.538	0.352	0.323	0.630	0.428	1.000

Nine, Autoregressive(1) Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.688	0.474	0.326	0.225	0.155	0.106
2	0.688	1.000	0.688	0.474	0.326	0.225	0.155
3	0.474	0.688	1.000	0.688	0.474	0.326	0.225
4	0.326	0.474	0.688	1.000	0.688	0.474	0.326
5	0.225	0.326	0.474	0.688	1.000	0.688	0.474
6	0.155	0.225	0.326	0.474	0.688	1.000	0.688
7	0.106	0.155	0.225	0.326	0.474	0.688	1.000

Appendix E-1: Continued

Ten, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.677	0.659	0.570	0.579	0.640	0.724
2	0.677	1.000	0.823	0.609	0.827	0.748	0.828
3	0.659	0.823	1.000	0.825	0.876	0.799	0.845
4	0.570	0.609	0.825	1.000	0.647	0.785	0.818
5	0.579	0.827	0.876	0.647	1.000	0.792	0.877
6	0.640	0.748	0.799	0.785	0.792	1.000	0.918
7	0.724	0.828	0.845	0.818	0.877	0.918	1.000

Eleven, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	0.001	0.000	0.000	0.000	0.000	0.000	0.000
2	0.000	0.001	0.000	0.000	0.000	0.000	0.000
3	0.000	0.000	0.001	0.000	0.000	0.000	0.000
4	0.000	0.000	0.000	0.001	0.000	0.000	0.000
5	0.000	0.000	0.000	0.000	0.001	0.000	0.000
6	0.000	0.000	0.000	0.000	0.000	0.001	0.000
7	0.000	0.000	0.000	0.000	0.000	0.000	0.001

Twelve, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.604	0.604	0.604	0.604	0.604	0.604
2	0.604	1.000	0.604	0.604	0.604	0.604	0.604
3	0.604	0.604	1.000	0.604	0.604	0.604	0.604
4	0.604	0.604	0.604	1.000	0.604	0.604	0.604
5	0.604	0.604	0.604	0.604	1.000	0.604	0.604
6	0.604	0.604	0.604	0.604	0.604	1.000	0.604
7	0.604	0.604	0.604	0.604	0.604	0.604	1.000

Appendix F-1: Estimated covariance metrics of errors for the IBI and IBI metrics models describing HUC-14 watersheds.

Total IBI score, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.009	-0.052	-0.112	0.210	0.248	0.483
2	0.009	1.000	0.598	0.706	0.730	0.767	0.574
3	-0.052	0.598	1.000	0.794	0.770	0.719	0.671
4	-0.112	0.706	0.794	1.000	0.834	0.786	0.681
5	0.210	0.730	0.770	0.834	1.000	0.821	0.795
6	0.248	0.767	0.719	0.786	0.821	1.000	0.628
7	0.483	0.574	0.671	0.681	0.795	0.628	1.000

One, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.719	0.719	0.719	0.719	0.719	0.719
2	0.719	1.000	0.719	0.719	0.719	0.719	0.719
3	0.719	0.719	1.000	0.719	0.719	0.719	0.719
4	0.719	0.719	0.719	1.000	0.719	0.719	0.719
5	0.719	0.719	0.719	0.719	1.000	0.719	0.719
6	0.719	0.719	0.719	0.719	0.719	1.000	0.719
7	0.719	0.719	0.719	0.719	0.719	0.719	1.000

Two - Headwater, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.766	0.766	0.766	0.766	0.766	0.766
2	0.766	1.000	0.766	0.766	0.766	0.766	0.766
3	0.766	0.766	1.000	0.766	0.766	0.766	0.766
4	0.766	0.766	0.766	1.000	0.766	0.766	0.766
5	0.766	0.766	0.766	0.766	1.000	0.766	0.766
6	0.766	0.766	0.766	0.766	0.766	1.000	0.766
7	0.766	0.766	0.766	0.766	0.766	0.766	1.000

Two - Wading, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	-0.076	0.014	-0.009	0.400	0.282	0.225
2	-0.076	1.000	0.419	0.665	0.540	0.201	0.262
3	0.014	0.419	1.000	0.279	0.494	0.155	0.534
4	-0.009	0.665	0.279	1.000	0.544	0.390	0.176
5	0.400	0.540	0.494	0.544	1.000	0.472	0.380
6	0.282	0.201	0.155	0.390	0.472	1.000	0.470
7	0.225	0.262	0.534	0.176	0.380	0.470	1.000

Three - Headwater, Toeplitz Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.477	0.477	0.643	0.817	0.334	0.695
2	0.477	1.000	0.643	0.817	0.334	0.695	0.681
3	0.477	0.643	1.000	0.477	0.643	0.817	0.334
4	0.643	0.817	0.477	1.000	0.477	0.643	0.817
5	0.817	0.334	0.643	0.477	1.000	0.477	0.643
6	0.334	0.695	0.817	0.643	0.477	1.000	0.477
7	0.695	0.681	0.334	0.817	0.643	0.477	1.000

Appendix F-1: Continued

Three - Wading, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.098	0.253	0.335	0.309	0.671	0.023
2	0.098	1.000	0.759	0.687	0.666	0.143	0.086
3	0.253	0.759	1.000	0.791	0.741	0.341	0.446
4	0.335	0.687	0.791	1.000	0.663	0.641	0.612
5	0.309	0.666	0.741	0.663	1.000	0.534	0.320
6	0.671	0.143	0.341	0.641	0.534	1.000	0.481
7	0.023	0.086	0.446	0.612	0.320	0.481	1.000

Four - Headwater, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.559	0.559	0.559	0.559	0.559	0.559
2	0.559	1.000	0.559	0.559	0.559	0.559	0.559
3	0.559	0.559	1.000	0.559	0.559	0.559	0.559
4	0.559	0.559	0.559	1.000	0.559	0.559	0.559
5	0.559	0.559	0.559	0.559	1.000	0.559	0.559
6	0.559	0.559	0.559	0.559	0.559	1.000	0.559
7	0.559	0.559	0.559	0.559	0.559	0.559	1.000

Four - Wading Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.579	0.579	0.579	0.579	0.579	0.579
2	0.579	1.000	0.579	0.579	0.579	0.579	0.579
3	0.579	0.579	1.000	0.579	0.579	0.579	0.579
4	0.579	0.579	0.579	1.000	0.579	0.579	0.579
5	0.579	0.579	0.579	0.579	1.000	0.579	0.579
6	0.579	0.579	0.579	0.579	0.579	1.000	0.579
7	0.579	0.579	0.579	0.579	0.579	0.579	1.000

Five, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.753	0.660	0.397	0.834	0.855	0.910
2	0.753	1.000	0.782	0.658	0.908	0.896	0.903
3	0.660	0.782	1.000	0.863	0.844	0.864	0.803
4	0.397	0.658	0.863	1.000	0.648	0.744	0.560
5	0.834	0.908	0.844	0.648	1.000	0.902	0.937
6	0.855	0.896	0.864	0.744	0.902	1.000	0.892
7	0.910	0.903	0.803	0.560	0.937	0.892	1.000

Six, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.564	0.365	0.270	0.429	0.366	0.168
2	0.564	1.000	0.724	0.672	0.607	0.522	0.388
3	0.365	0.724	1.000	0.717	0.609	0.510	0.270
4	0.270	0.672	0.717	1.000	0.818	0.547	0.654
5	0.429	0.607	0.609	0.818	1.000	0.861	0.517
6	0.366	0.522	0.510	0.547	0.861	1.000	0.342
7	0.168	0.388	0.270	0.654	0.517	0.342	1.000

Appendix F-1: Continued

Seven, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.201	0.201	0.201	0.201	0.201	0.201
2	0.201	1.000	0.201	0.201	0.201	0.201	0.201
3	0.201	0.201	1.000	0.201	0.201	0.201	0.201
4	0.201	0.201	0.201	1.000	0.201	0.201	0.201
5	0.201	0.201	0.201	0.201	1.000	0.201	0.201
6	0.201	0.201	0.201	0.201	0.201	1.000	0.201
7	0.201	0.201	0.201	0.201	0.201	0.201	1.000

Eight, Unstructured Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.097	-0.126	-0.225	0.252	-0.036	0.472
2	0.097	1.000	0.493	0.380	0.205	0.190	0.361
3	-0.126	0.493	1.000	0.748	0.543	0.601	0.534
4	-0.225	0.380	0.748	1.000	0.472	0.460	0.134
5	0.252	0.205	0.543	0.472	1.000	0.582	0.691
6	-0.036	0.190	0.601	0.460	0.582	1.000	0.150
7	0.472	0.361	0.534	0.134	0.691	0.150	1.000

Nine - Headwater, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.215	0.215	0.215	0.215	0.215	0.215
2	0.215	1.000	0.215	0.215	0.215	0.215	0.215
3	0.215	0.215	1.000	0.215	0.215	0.215	0.215
4	0.215	0.215	0.215	1.000	0.215	0.215	0.215
5	0.215	0.215	0.215	0.215	1.000	0.215	0.215
6	0.215	0.215	0.215	0.215	0.215	1.000	0.215
7	0.215	0.215	0.215	0.215	0.215	0.215	1.000

Nine - Wading Toeplitz Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.744	0.614	0.649	0.649	0.744	0.608
2	0.744	1.000	0.457	0.608	0.744	0.649	0.614
3	0.614	0.457	1.000	0.649	0.374	0.608	0.744
4	0.649	0.608	0.649	1.000	0.614	0.744	0.744
5	0.649	0.744	0.374	0.614	1.000	0.608	0.457
6	0.744	0.649	0.608	0.744	0.608	1.000	0.649
7	0.608	0.614	0.744	0.744	0.457	0.649	1.000

Ten, Compound Symmetry Covariance

Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.620	0.620	0.620	0.620	0.620	0.620
2	0.620	1.000	0.620	0.620	0.620	0.620	0.620
3	0.620	0.620	1.000	0.620	0.620	0.620	0.620
4	0.620	0.620	0.620	1.000	0.620	0.620	0.620
5	0.620	0.620	0.620	0.620	1.000	0.620	0.620
6	0.620	0.620	0.620	0.620	0.620	1.000	0.620
7	0.620	0.620	0.620	0.620	0.620	0.620	1.000

Appendix F-1: Continued

Eleven, Variance Components							
Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	0.001	0.000	0.000	0.000	0.000	0.000	0.000
2	0.000	0.001	0.000	0.000	0.000	0.000	0.000
3	0.000	0.000	0.001	0.000	0.000	0.000	0.000
4	0.000	0.000	0.000	0.001	0.000	0.000	0.000
5	0.000	0.000	0.000	0.000	0.001	0.000	0.000
6	0.000	0.000	0.000	0.000	0.000	0.001	0.000
7	0.000	0.000	0.000	0.000	0.000	0.000	0.001
Twelve, Toeplitz Covariance							
Row	Col1	Col2	Col3	Col4	Col5	Col6	Col7
1	1.000	0.702	0.641	0.706	0.706	0.702	0.641
2	0.702	1.000	0.706	0.706	0.641	0.585	0.526
3	0.641	0.706	1.000	0.702	0.585	0.526	0.683
4	0.706	0.706	0.702	1.000	0.702	0.641	0.585
5	0.706	0.641	0.585	0.702	1.000	0.706	0.702
6	0.702	0.585	0.526	0.641	0.706	1.000	0.706
7	0.641	0.526	0.683	0.585	0.702	0.706	1.000

Appendix G-1: GIS Derived Statistics

The following tables contain information derived from GIS maps created in ArcView. Summary statistics are listed by stormwater watersheds and include land use information from Indiana's GAP data program.

Watershed Name	White River - Truitt Ditch		White River - Muncie Creek		Buck Creek - Macedonia Creek		White River - Buck Creek		White River - York Prairie Creek		Jake's Creek - Eagle Branch	
14-Digit HUC	05120201010120		05120201010130		05120201020020		05120201020060		05120201030010		05120201040030	
GAP Land Use	Area (Ha.)	30 meter Buffer	Area (Ha.)	30 meter Buffer	Area (Ha.)	30 meter Buffer	Area (Ha.)	30 meter Buffer	Area (Ha.)	30 meter Buffer	Area (Ha.)	30 meter Buffer
Ag: Pasture	6	7%	12	17%	12	11%	16	15%	17	15%	12	17%
Ag: Row Crop	25	29%	30	43%	63	58%	14	12%	38	34%	33	47%
Ag: Wet Areas	0	0%	0	0%	0	0%	0	0%	0	0%	0	0%
Deciduous Forest	17	20%	2	3%	11	10%	13	12%	18	17%	11	16%
Evergreen Forest	13	14%	0	0%	2	2%	4	4%	0	0%	0	0%
Open Water	0	0%	0	0%	0	0%	0	0%	0	0%	0	0%
Palustrine Forest	11	13%	3	5%	7	6%	25	23%	18	17%	3	4%
Palustrine Herbaceous	9	11%	6	9%	0	0%	23	21%	13	12%	3	4%
Palustrine Sparsely Veg.	1	1%	0	0%	0	0%	0	0%	0	0%	0	0%
Palustrine Deciduous	0	0%	0	0%	0	0%	1	1%	0	0%	0	0%
Shrubland	1	1%	0	0%	0	0%	1	1%	1	1%	0	0%
Urban: High Density	0	0%	4	6%	2	1%	1	1%	0	0%	0	0%
Urban: Low Density	0	0%	11	16%	10	9%	11	10%	3	2%	1	2%
Woodland	3	4%	0	1%	2	2%	1	1%	2	2%	6	9%
Total Riparian Area	88		70		109		108		111		69	
Total Watershed Area	4,769		3,480		6,339		3,553		4,852		4,698	

Watershed Name	White River - Truitt Ditch		White River - Muncie Creek		Buck Creek - Macedonia Creek		White River - Buck Creek		White River - York Prairie Creek		Jake's Creek - Eagle Branch	
14-Digit HUC	05120201010120		05120201010130		05120201020020		05120201020060		05120201030010		05120201040030	
Land Cover	Area (Ha.)	Percent Total	Area (Ha.)	Percent Total	Area (Ha.)	Percent Total	Area (Ha.)	Percent Total	Area (Ha.)	Percent Total	Area (Ha.)	Percent Total
Total Impervious Cover	166	3.5%	247	7.1%	447	7.0%	594	16.7%	437	9.0%	431	9.2%
Total Wetland	96	2.02%	98	2.81%	85	1.34%	117	3.31%	157	3.24%	97	2.07%