Muncie
SANITARY DISTRICT

**Indiana Scientific Purpose License Number: 15-114**

# Bureau of Water Quality Annual Macroinvertebrate Community Report 2016

Bureau of Water Quality
5150 W. Kilgore Ave.
Muncie, IN 47304

Phone: 765-747-4896
Fax: 765-213-6444
www.munciesanitary.org/bwq

Prepared by:
Laura Bowley, Macroinvertebrate Biologist, BWQ
April 2017

TABLE OF CONTENTS

***Acknowledgements**: Thank you to Scott Zello and Kylie Schofield for their assistance in obtaining these samples, and for their countless hours of data entry, proofing, slide preparation, and general assistance in preparing this report.*

## PREFACE

This paper contains results of the Bureau of Water Quality's (BWQ's) macroinvertebrate and mussel biomonitoring for the year 2016. For the purpose of displaying trends, some graphs and tables will present data from past years. However, the analysis given here is only for 2016. If further investigation of past years is needed, please refer to prior reports from this organization.

From 2013-2016 an additional Buck Creek site was sampled. This site (BUC 0.0) was sampled to observe changes in the site before and after best management practices (implemented in late 2013) were put into place.

In 2016, to provide more accuracy and adherence with the Indiana Department of Environmental Management, we obtained and implemented the use of the identification keys they use for identification of macroinvertebrates.

In 2014, one zebra mussel *Dreissena polymorpha* was found on a sampler in Prairie Creek Reservoir, upstream of Muncie. The reservoir is very near White River, connected via Prairie Creek. In 2016, zebra mussels were found on a sampler in Prairie Creek. It is expected that *Dreissena spp*. will be found in White River in 2017.

Due to additional studies comparing multiple sampling methods, one mussel site was sampled in 2016. However, mussel populations at other sites are always qualitatively observed and monitored.

## INTRODUCTION

**West Fork White River and the Bureau of Water Quality.—**The headwaters of the West Fork White River (WFWR) can be found near Winchester, Indiana, moving westward through Muncie, draining approximately 384 square miles at the Madison County/Delaware County line (Hoggat 1975). The land along the river in Delaware County is primarily used for agriculture (corn, soybeans, and livestock), but also includes the urban area of Muncie. Muncie is a heavily industrialized community that has included electroplating firms, transmission assembly plants, a secondary lead smelter, foundries, heat treatment operations, galvanizing operations, and tool and die shops (ICLEI Case Study #19 1994).

In 1972, the Division of Water Quality (DWQ), now named the Bureau of Water Quality (BWQ), was established out of a need to regulate and control the sources responsible for polluting White River and its tributaries in and around Muncie, Indiana. The BWQ also wanted to attain those goals set forth by legislation of the 1970's and 1980's (The Water Pollution Act of 1972, the Clean Water Act of 1977 and the Water Quality Act of 1987). One of the ultimate goals is biological integrity, defined by Karr & Dudley (1981) as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region."

Since the establishment of the BWQ, industries have installed millions of dollars in industrial pretreatment equipment, and corrective action is constantly being taken to prevent spills from entering the sewers and waterways. In addition, an ongoing program has reduced, and in some cases eliminated, pollution entering White River from combined sewer overflows (CSOs). Improvements have been made to the Muncie Water Pollution Control Facility (MWPCF), local sewers have been built to correct septic tank problems, and wildlife habitat has been developed along the river (Craddock 1990).

To get the best representation of the quality of a water system, both chemical and biological monitoring should be implemented. The benefits of chemical testing are vast; however, chemical monitoring can miss or underestimate combined chemical effects, sporadic events, and other factors such as habitat degradation (Karr 1981).

A benefit to using biological communities as indicators of water quality is their longevity and sensitivity to disturbances in the habitat in which they live. The observed condition of the aquatic biota, at any given time, is the result of the chemical and physical dynamics that occur in a water body over time (OEPA DWQMA 1987). Alone, neither gives a complete picture of water quality, however, the combination of biological and chemical monitoring increases the chances that degradation to the water body will be detected (Karr 1991).

**Mussels as biomonitors**.—Freshwater mussels are considered the most imperiled group of organisms in North America (Lydeard et al. 2004; Strayer et al. 2004), if not the world (Strayer 2008), and are declining at alarming and unprecedented rates (Neves et al 1997; Ricciardi & Rasmussen 1999; Vaughn & Taylor 1999; Strayer & Smith 2003; Poole & Downing 2004; Regnier et al. 2009). In North America alone, 72% of the native mussel fauna is either federally listed as endangered or threatened or considered to be in need of some protection (Haag 2009). At one time, 90 species of Unionid (of the family Unionidae) mussels were known to have existed in the eight Great Lake and Upper Mississippi states. Now, 33% are listed as extinct, endangered, or are candidates for that listing (Ball & Schoenung 1995). In the United States, 71 taxa are currently listed as endangered or threatened by the Endangered Species Act (USFWS 2005) and are suffering an extinction rate higher than any other North American fauna (Ricciardi & Rasmussen 1999). Contributors to this decline include commercial harvest, degradation of habitat (including channelization and dredging), toxic chemicals, and siltation. Other significant contributors include: impoundments (Vaughn & Taylor 1999; Watters 2000; Dean et al. 2002), water pollution (organic, inorganic, and thermal) (Mummert et al. 2003; Keller & Augspurger 2005; Valenti et al. 2005; 2006; Gooding et al. 2006; Bringolf et al. 2007; March et al. 2007; Wang et al. 2007; Cope et al. 2008; Besser et al. 2009), habitat alterations, and land use practices (Clarke 1981; Ball & Schoenung 1995; Biggins et al. 1995; Couch 1997; Gatenby et al. 1998; Payne et al. 1999; Watters 1999; Poole & Downing 2004). In 1990, the US EPA listed sedimentation as the top pollutant of rivers in the United States (Box & Mossa 1999). Studies have shown that silt accumulation of 0.25 to 1 inch resulted in nearly 90% mortality of mussels tested (Ellis 1936). This affects mussels by reducing interstitial flow rates, clogging mussel gills, and reducing light for photosynthesis of algae (primary forage of the mussel). Suspended particles also cause difficulty with the necessary fish and mussel interactions needed for reproduction and survival (Box & Mossa 1999). These indicate the importance of water quality as a factor in mussel survival. It is for these reasons, as well as their long life span, feeding habits, persistent shells (Strayer 1999a) and sensitive growth and reproductive rates (Burky 1983) that mussels serve well as biological indicators.

**Macroinvertebrates as Biomonitors**.—There are numerous reasons for using macroinvertebrates as indicators of water quality. Their ubiquitous nature, large numbers (individuals and species), and relative ease of sampling with inexpensive equipment make them ideal for bioassessments (Lenat et al. 1980; Hellawell 1986; Lenat & Barbour 1993). Macroinvertebrates are relatively sessile, allowing spatial analysis of disturbances (Tesmer & Wefring 1979; Hellawell 1986; Abel 1989). The extended life cycles of most aquatic insects allows for temporal analysis as well (Lenat et al. 1980; Hellawell 1986). Finally, macroinvertebrate species are well documented; many identification keys and forms of analysis are available, and specific responses to pollutants and stressors are well known (Hellawell 1986; Abel 1989; Rosenberg & Resh 1993). They are especially useful in situations where intermittent or mild organic enrichment is present (Chutter 1972).

## MUSSEL METHODS

**Mussel Field Sampling.**—Sampling methods followed an adaptive cluster sampling with initial random samples without replacement, described by Strayer & Smith (2003), originated by Thompson (1992). Studies have shown a decrease in variance (Mwangi & Salim 2012) and an increase in sampling efficiency (Mwangi & Salim 2012; Smith et al. 2004) compared to conventional sampling methods. Additionally, the yield of individual mussels and rare species has been found to be increased (Smith et al.2003). Sample size was determined following Cochran (1977) and Hansen et al. (2007).

The equation is as follows:

$$n = \frac{s^2 t^2_{n-1}}{\delta^2}$$

Where:

$n$ = sample size needed

$s^2$ = variance estimated from a pilot study

$t$ = t-statistic defined for a given $\alpha$ level

$\delta$ = precision in absolute terms

Field sheets (Appendix A, Table 8) were completed at each site (Appendix A, Table 5). A site was 100 m in river length; widths were taken at each meter along the river banks. A sampling grid was then plotted, and quadrats were then randomly chosen. Finally, a condition variable was then chosen, based on pilot studies.

Quadrats constructed with 0.25 $m^2$ PVC tubing were then secured in the randomly selected quadrat positions. A glass-bottom bucket was used to examine the river bottom for protruding mussels, which were removed and placed in a bucket, which was submerged and secured in the stream. Then, wearing neoprene gloves and using a garden claw, biologists began digging within the quadrat, removing all mussels and clams to a uniform depth of 10-15 cm (Dunn 1999; Smith et al. 1999). All retained mussels were identified, measured, aged (counting external annuli), and sex was recorded if the species was sexually dimorphic. Mussels were then replaced in the substrate as close to the original position as possible.

If the condition variable was not met, sampling

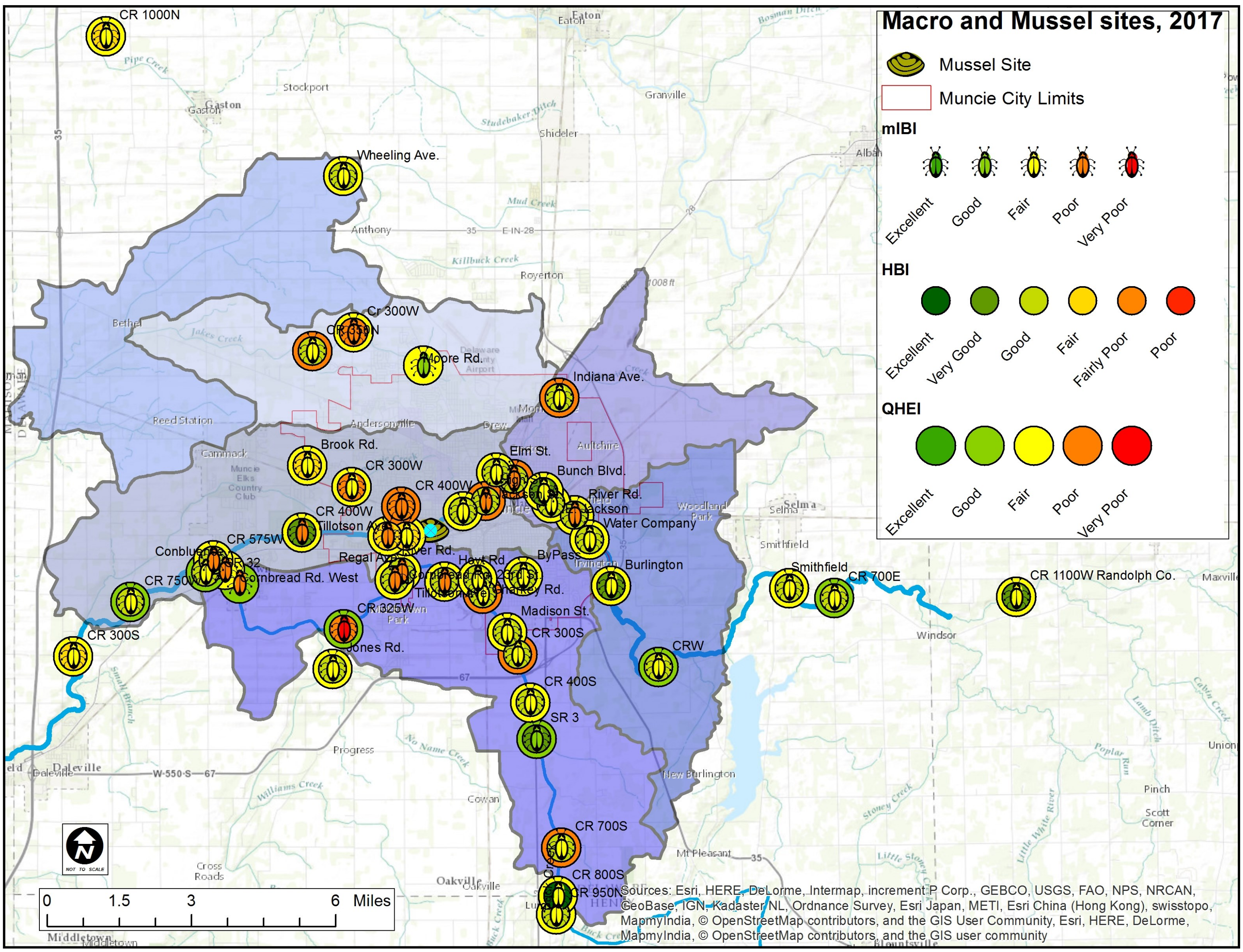



Figure 1.—Macroinvertebrate and mussel sites, 2016

proceeded at the next randomly chosen quadrat. If the condition was met, neighboring quadrats in a cross-shaped pattern (Smith et al. 2004) were sampled. This continued until all quadrats did not meet the condition variable. The site was considered complete when all randomly chosen quadrats and their corresponding neighborhoods were sampled.

Asian clam, *Corbicula fluminea,* were also recorded. The largely fluctuating populations of this invasive species can greatly affect native mussel populations. Occasional rapid die-offs of Asian clam can occur after reproduction and sudden drops in dissolved oxygen (D.O.) (usually during the warm summer months). This can result in high levels of ammonia, detrimental to the entire aquatic ecosystem (Schiller 1997; Cherry et al. 2005; Cooper et al. 2005). It was determined that calculations of Asian clam means cannot be accurately determined from this type of sampling; the condition variable is set with the focus on Unionid density determinations. Future considerations will include an accurate way to include calculations of Asian clam and fingernailclam, *Sphaerium* spp..

**Mussel Data Tabulation**.—The Horvitz-Thompson (Thompson 1990) population estimator has been determined to be the superior choice for determining total population (per $m^2$) when utilizing the adaptive cluster method (Salehi 1999, 2003; Salehi & Smith 2005; Su & Quinn 2003). This complex calculation was determined using Philippi's (2005) code in SAS (2008). Significance was determined by $P < 0.05$ unless otherwise noted.

Table 1.—mIBI submetrics and stand alone indices and their response to disturbance

| mIBI Sub-Metrics and Stand-Alone Indices | Response to Disturbance |
|---|---|
| Total Number of Taxa | Decrease |
| Total Abundance of Individuals | Decrease |
| Number of EPT taxa | Decrease |
| % Orthocladiinae & Tanytarsini | Increase |
| % Non-Insects (-Crayfish) | Increase |
| Number of Dipteran Taxa | Increase |
| % Intolerant Taxa (Score 0-3) | Decrease |
| % Tolerant Taxa | Decrease |
| % Predators | Decrease |
| % Shredders & Scrapers | Decrease |
| % Collectors/Filterers | Increase |
| % Sprawlers | Decrease |
| Hilsenhoff Biotic Index | Increase |
| Shannon-Wiener Diversity Index (H') | Decrease |
| Shannon Evenness Index (J') | Decrease |
| % Dominance of Top Three Taxa | Increase |
| % Chironomidae | Increase |

## MACROINVERTEBRATE METHODS

**Macroinvertebrate Field Sampling.**—Macroinvertebrate samples were taken at 14 sites on White River, and five sites along Buck Creek (Figure 1 and Appendix B, Table 9). Sampling followed the current IDEM Multi-habitat Macroinvertebrate Collection Procedure (MHAB) (IDEM 2010). This methodology includes a composite of a one minute riffle or mid-stream kick (if there is no riffle present) and an approximately 12-minute, 50-m riparian bank sample. The contents were elutriated six times and poured through a #30 USGS sieve. The remaining content in the sieve was then subsampled for 15 minutes. Organisms were placed in a vial with 99.5% isopropyl alcohol and returned to the lab for later identification.

Field sheets (Appendix B, Table 14) were completed, including the "Qualitative Habitat Evaluation Index" sheet (Appendix B, Table 18). Taxa sheets for each macroinvertebrate site can be found in Appendix B, Table 15. QHEI sheets and tabulations can be found in Appendix B, Table 18.

**Macroinvertebrate Laboratory Methods.**—All organisms were identified to the lowest practical level, usually genus. Non- Chironomid macroinvertebrates were identified using dichotomous keys by Peckarsky et al. (1990), Thorp & Covich (1991), Merritt & Cummins (1996), Wiggins (1996), and Smith (2001). Chironomids (with heads removed) were mounted on slides in a high viscosity mountant. Chironomids were then identified using Peckarsky et al. (1990), Mason (1998), and Epler (2001).

Table 2.—mIBI scores and corresponding ratings.

| Total Score | Narrative Rating |
|---|---|
| 54-60 | Excellent |
| 44-53 | Good |
| 35-43 | Fair |
| 23-34 | Poor |
| 0-22 | Very Poor |

**Macroinvertebrate Data Tabulation.**—Macroinvertebrate calculations were based on IDEM's Macroinvertebrate Index of Biotic Integrity (mIBI), the Hilsenhoff Biotic Index (HBI), Shannon-Wiener Diversity Index (H'), Shannon Evenness Index (J'), Percent Dominance of Top Three Taxa, and Percent Chironomidae.

*IDEM's Macroinvertebrate Index of Biotic Integrity (mIBI):* The mIBI is a multimetric index (Table 1) that has been calibrated using statewide data . After calculating each metric, the resulting score is assigned a specific "rank" (1, 3, or 5) based on the drainage area of the site. The sum of all metrics is then used to determine the final score. This final score is assigned a narrative rating (Table 2). IDEM ratings also include a designation of "Fully Supporting" of aquatic life (mIBI score $\geq$ 36), or "Not Supporting" of aquatic life (mIBI score <36).

Table 3.—HBI values and corresponding ratings.

| HBI Score | Water Quality | Degree of Organic Pollution |
|---|---|---|
| 0.00-3.50 | Excellent | No apparent organic pollution. |
| 3.51-4.50 | Very Good | Possible slight organic pollution. |
| 4.51-5.50 | Good | Some organic pollution. |
| 5.51-6.50 | Fair | Fairly significant organic pollution |
| 6.51-7.50 | Fairly Poor | Significant organic pollution. |
| 7.51-8.50 | Poor | Very significant organic pollution. |
| 8.51-10.00 | Very Poor | Severe organic pollution. |

*Hilsenhoff Biotic Index (HBI):* The HBI (Hilsenhoff 1987) is a biotic index that incorporates a weighted relative abundance of each taxon in order to determine a score for the community (Rosenberg & Resh 1993). Organisms are assigned a value between 0 and 10, according to their tolerance of organic and nutrient pollution

(Appendix B, Table 10). The number of each organism is multiplied by the tolerance value. The sum of these results is then averaged to get the resulting HBI value for the site. Modified descriptive ratings can be found below in Table 3.

The Hilsenhoff Biotic Index is calculated as follows:

$$HBI = \sum \frac{x_i t_i}{N}$$

Where:

$X_i$ = number of each species

$T_i$ = tolerance value for each species (Appendix. B, Table 10)

$N$ = total number of arthropods in the sample with tolerance ratings

*Shannon-Wiener Diversity Index (H'):* The Shannon-Wiener Diversity Index is based on the premise that species diversity decreases with decreasing water quality (Wilhm 1967; Rosenberg & Resh 1993) in an effectively infinite community (Kaesler et al. 1978). This index incorporates both species richness as well as evenness (Ludwig & Reynolds 1988). Higher H' scores indicate increased species diversity (Vandermeer 1981; Gerritsen et al. 1998). The Shannon Wiener Index is calculated as follows:

$$H' = \sum p_i \ln p_i$$

Where:

$p_i$ = relative abundance of each species calculated as a proportion of individuals of a given species to the total number of individuals in the community.

*Shannon Evenness Index (J'):* Shannon Evenness Index (Pielou 1966) is calculated from the Shannon-Wiener Diversity Index and is a ratio of observed diversity to maximum diversity in order to measure evenness of the community. Higher J' scores indicate increased community evenness.

The Shannon Evenness Index is calculated as follows:

$$J' = \frac{H'}{\ln s}$$

Where:

$s$ = number of species

*Percent Dominance of Top Three Taxa:* A well balanced community is indicative of a healthy community. Predominance of only a few macroinvertebrate species can be indicative of stressors in the system (Plafkin et al. 1989; Klemm et al. 1990).

*Percent Chironomidae:* Chironomidae are generally considered to be pollution tolerant. An overabundance of these organisms can be indicative of stressors in the system (Plafkin et al. 1989; Barbour et al. 1994).

*Qualitative Habitat Evaluation Index (QHEI):* The QHEI was assessed to better determine the effect of habitat quality on the resulting scores. The QHEI (Rankin 1989) is an index that evaluates macro-habitat quality that has been found to be essential for fish communities as well as other aquatic life. QHEI metrics include substrate, instream cover, channel morphology, riparian condition, pool and riffle quality, and gradient. Each metric in the habitat assessment was scored, with the final sum of these scores reflecting available habitat (higher scores reflect better habitat). Narrative ratings for QHEI scores can be found in Table 4.

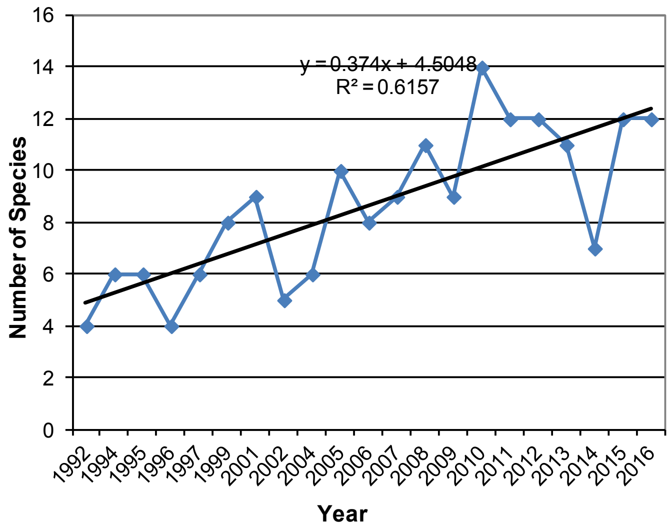
Table 4.—QHEI scores and corresponding ratings.

| QHEI score | Narrative Rating |
|---|---|
| 90-100 | Excellent |
| 71-89.9 | Good |
| 52-70.9 | Fair |
| 27-51.9 | Poor |
| 0-26 | Very Poor |

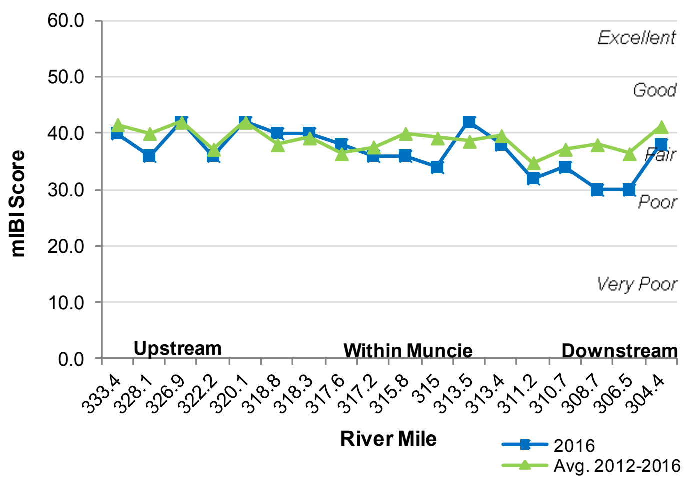
MUSSEL RESULTS

**WR 313.4.**—Mussels were collected at 30 initial quadrats at WR 313.4. The condition variable for adaptive sampling was set at $\geq$ 2 mussels per 0.25 $m^2$ quadrat, based on prior sampling efforts. Mussels collected at WR 313.4 in 2017 are reported in Appendix A, Table 6. Twelve Unionid species were sampled at this site. Species diversity has increased *($R^2$ = 0.64, $P$ < 0.001)* (Graph 1) since mussel sampling began in 1992. Unionid density (95% C.I.) at WR 313.4 was calculated to be 3.22/$m^2$ $\pm$ 1.80/$m^2$ (Appendix A, Table 7). Relative abundance (Appendix A, Graph 14) of all mussels indicated that Asian clam

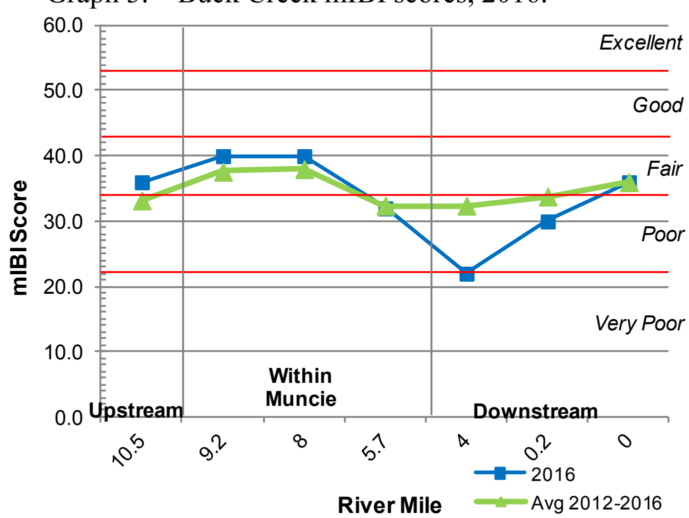
Graph 1.—Species diversity at WR 313.4, 1992-2016.




Graph 2.—White River mIBI scores, 2016




Graph 3.—Buck Creek mIBI scores, 2016.




comprised 85.0% of the sample, and Sphaeriidae comprised 0.09% of the sample. The three most abundant Unionid species at WR 313.4 were flutedshell *Lasmigona costata,* mucket *Actinonaias ligamentina,*, and elktoe *Alasmidonta marginata*.

## MACROINVERTEBRATE RESULTS

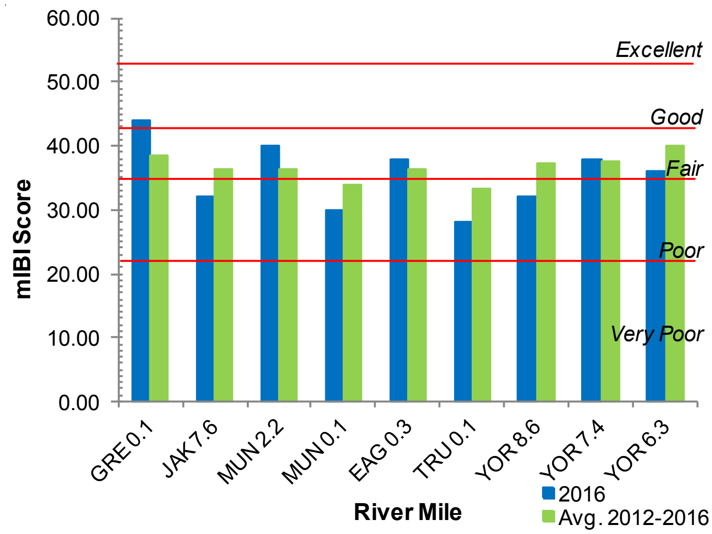
**mIBI.—White River:** White River mIBI scores (Graph 2 and Appendix B, Table 11) ranged from 30.0 (WHI 308.7 ) to 42 (WHI 326.9, WHI 320.1, and WHI 313.5), *Poor* to *Fair*. In 2016, WHI 315.0, WHI 311.2, WHI 310.7, and WHI 308.7 would be considered "Not Supporting" of aquatic life by IDEM. Mean mIBI scores (Appendix B, Table 12) upstream, within, and downstream of Muncie were all *Fair*. No spatial or temporal trends were detected.

Individual submetrics provide additional information and trends. "Number of Taxa" has significantly decreased since 2011 at WHI 311.2 (*$R^2$ = 0.94, $P < 0.05$).* "Number of Diptera Taxa" increased at WHI 313.5 (*$R^2$ = 0.78, $P < 0.05$)* since 2011. "Percent Tolerant Taxa" increased at WHI 311.2 (*$R^2$ = 0.93, $P < 0.05$*) since 2011.

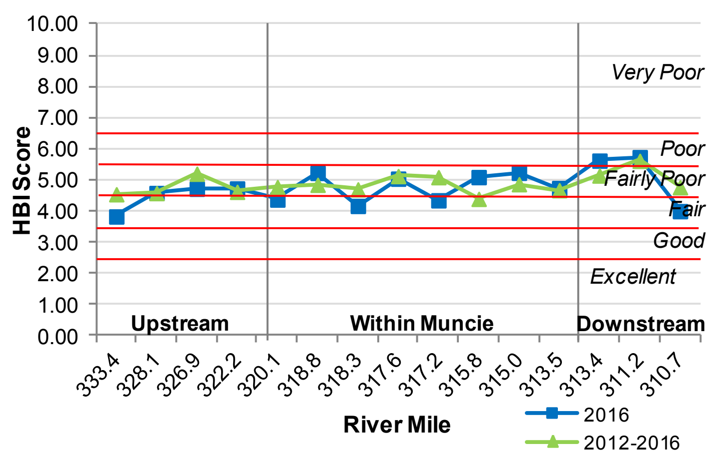
**Buck Creek:** Buck Creek mIBI scores (Graph 3 and Appendix B, Table 11) ranged from 22.0 (BUC 4.0) to 44.0 (BUC 11.3), *Poor* to *Good*. The mean mIBI score for Buck Creek was 35.1, *Fair*. In 2016, BUC 7.1, BUC 5.9, BUC 5.7, BUC 4.0, BUC 0.9, and BUC 0.2 would be considered "Not Supporting" of aquatic life by IDEM. The mean mIBI score (Appendix B, Table 12) on Buck Creek was 35 *Fair*. No spatial or temporal trends were detected.

Individual submetrics provide additional information and trends. "Number of EPT Taxa" has increased significantly at BUC 5.7 (*$R^2$ = 0.93, $P < 0.05$)* since 2011. In 2016, "Percent Non-Insects" increased on Buck Creek as it moved downstream (*$R^2$ = 0.88, $P < 0.05$). "*Number of Diptera Taxa" increased at BUC 10.5 (*$R^2$ = 0.88, $P < 0.05$)* since 2011. "Percent Tolerant Taxa" increased at BUC 8.0 ($R^2$ = *0.82, $P < 0.05$)*

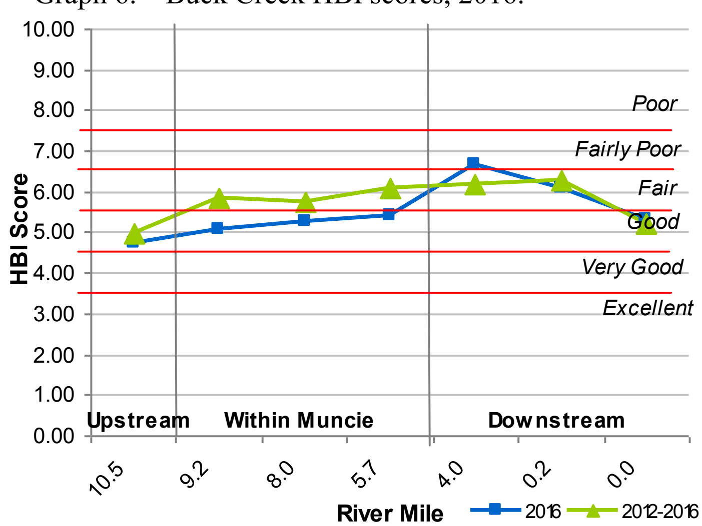
Graph 4.—Tributary mIBI scores, 2016.




Graph 5.—White River HBI scores, 2016.




Graph 6.—Buck Creek HBI scores, 2016.




since 2009. "Percent Sprawlers" has increased at BUC 10.5 ($R^2$ = *0.82, P < 0.05)* and decreased at BUC 8.0 ($R^2$ = *0.81, P < 0.05)* since 2009.

While no significant temporal mIBI trends were detected from 2009-2016 (Appendix B, Table 13), a few observations should be noted. On White River, there has only been one Poor mIBI score upstream of Muncie since 2009. Scores appear to fluctuate on White River from year to year, especially dramatic in 2012 and 2016. Negative mIBI scores appear to be fairly common among tributary sites.

**Smaller Tributary Sites:** York Prairie Creek mIBI scores (Graph 4 and Appendix B, Table 11) ranged from 32 (YOR 8.6) to 38 (YOR 6.3) *Poor* to *Fair*. One of the three York Prairie Creek sites sampled in 2016 (YOR 8.6 ) would be considered "Not Supporting" of aquatic life by IDEM. *Poor* mIBI scores were also found at JAK 7.6, MUN 0.1, and TRU 0.1. These sites would be considered "Not Supporting" of aquatic life by IDEM.

An average of mIBI scores from 2009-2016 (Graph 4) indicate that of all small tributaries, the sites at TRU 0.1, MUN 2.2, and MUN 0.1 were the most impacted sites. No spatial or temporal trends were detected.

Individual submetrics provide additional information and trends. "Percent Orthocladiinae and Tanytarsini" has decreased significantly at YOR 6.3 ($R^2$ = *0.84, P < 0.05)* since 2009. "Number of Diptera Taxa" increased at EAG 0.3 ($R^2$ = *0.92, P < 0.05)*, and YOR 7.4 ($R^2$ = *0.78, P < 0.05)* since 2011. "Percent Tolerant Taxa" has increased at YOR 6.3 ($R^2$ = *0.96, P < 0.05)* since 2011. "Percent Sprawlers" has decreased at LUI 0.1 ($R^2$ = *0.82, P < 0.05)* since 2011.

**Stand Alone Indices.—**

*HBI:* **White River:** White River HBI scores (Graph 5 and Appendix B, Table 11) ranged from 6.03 (WHI 304.4) to 4.01 (WHI 10.7), *Fair* to *Very Good*. Mean HBI scores (Appendix B, Table 12) dropped slightly from Very Good to Good within Muncie, and even improved slightly below Muncie city limits. No spatial or temporal trends were detected.

**Buck Creek:** Buck Creek HBI scores (Graph 6, Appendix B, Table 11) ranged from 6.67 (BUC 4.0) to 2.98 (BUC 14.9), *Fairly Poor* to *Excellent*. The mean HBI score (Appendix B, Table 12) was 5.1, *Good*. No spatial or temporal trends were detected.

**Smaller Tributary Sites:** York Prairie Creek HBI scores (Graph 7 and Appendix B, Table 11) ranged from 6.91 (YOR 7.4) to 5.55 (YOR 6.3), *Fairly Poor* to *Fair*. A *Fairly Poor* score was also found at JAK 7.6. A negative trend in HBI scores (2011-2016) was found at EAG 0.3 *($R^2 = 0.79, P < 0.05$)*. The most organically impacted sites from 2012-2016 appear to be JAK 7.6, YOR 8.6, and GRE 0.1.

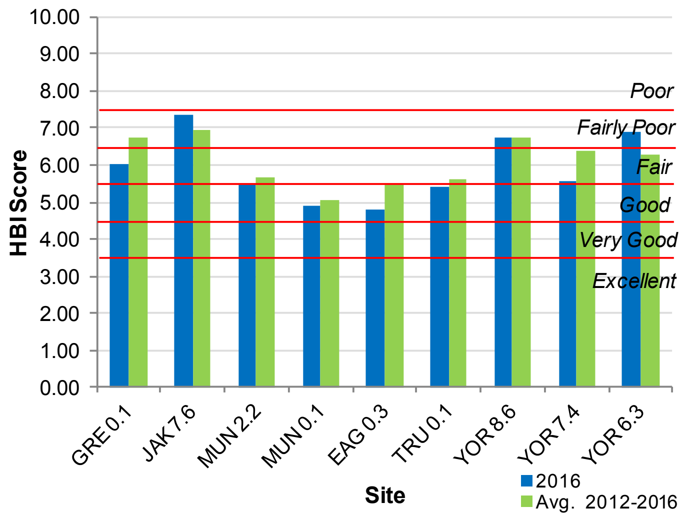
*H’*: **White River:** White River H’ scores (Graph 8 and Appendix B, Table 11) ranged from 2.71 (WHI 315.0) to 3.60 (WHI 326.9). Mean H’ scores (Appendix B, Table 12) dropped as White River progressed downstream. No significant spatial trends were detected in either 2009-2016 average data (Appendix B, Table 12) or the 2016 data.

**Buck Creek:** Buck Creek H’ scores (Graph 9 and Appendix B, Table 11) ranged from 2.00 (BUC 14.9) to 3.42 (BUC 15.2). The mean H’ score (Appendix B, Table 12) was 2.91. There were no significant spatial trends seen in either the 2009-2016 average scores or the 2016 data. However, BUC 8.0 exhibited a significant decrease in H’ scores from 2009-2016 ($R^2 = 0.79, P < 0.05$).

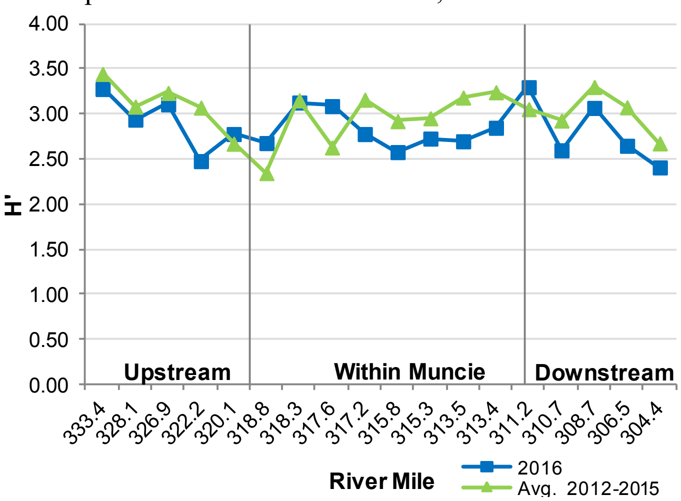
**Smaller Tributary Sites**: York Prairie Creek H’ scores (Graph 10 and Appendix B, Table 11 ranged from 2.17 (YOR 8.6) to 3.25 at YOR 7.4. The remaining smaller tributary H’ scores (Graph 10 and Appendix B, Table 11) ranged from 2.29 (EAG 0.3) to 3.55 (YFM 1.0).

*Remaining Stand Alone Indices:* **White River:** White River J’ scores (Appendix B, Table 11) ranged from 0.72 (WHI 304.4) to

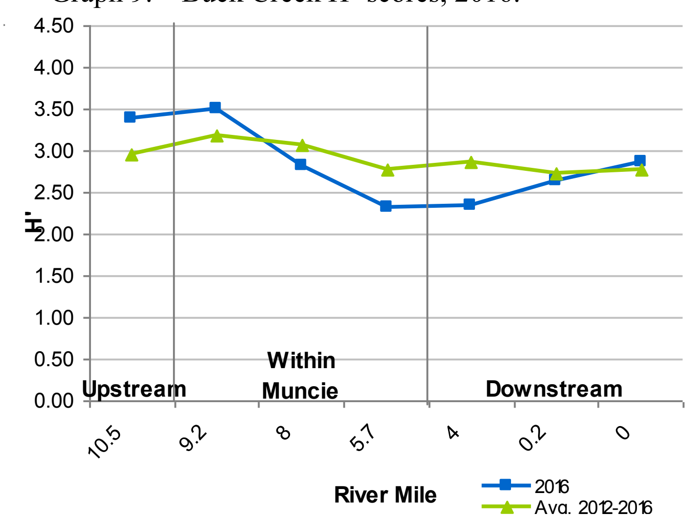
Graph 7.—Tributary HBI scores, 2016.



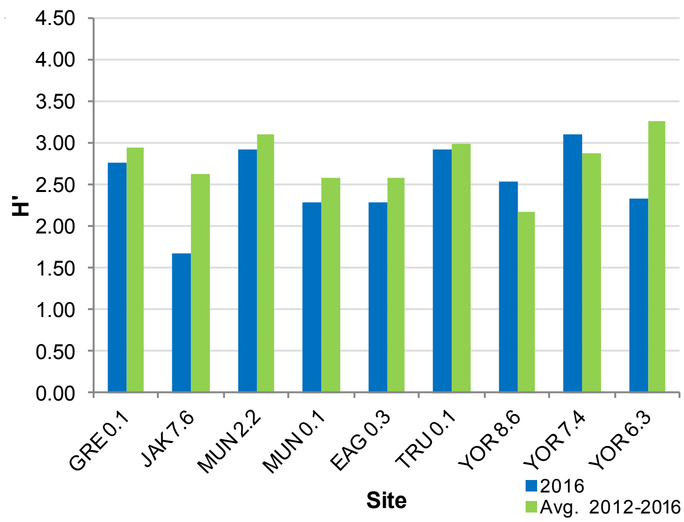

Graph 8.—White River H’ scores, 2016.



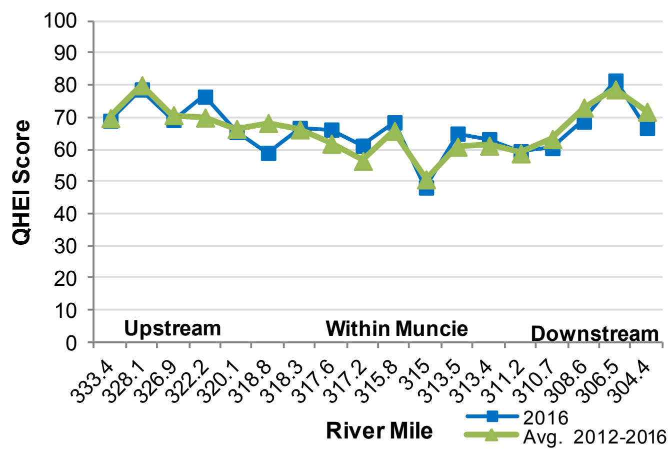

Graph 9.—Buck Creek H’ scores, 2016.



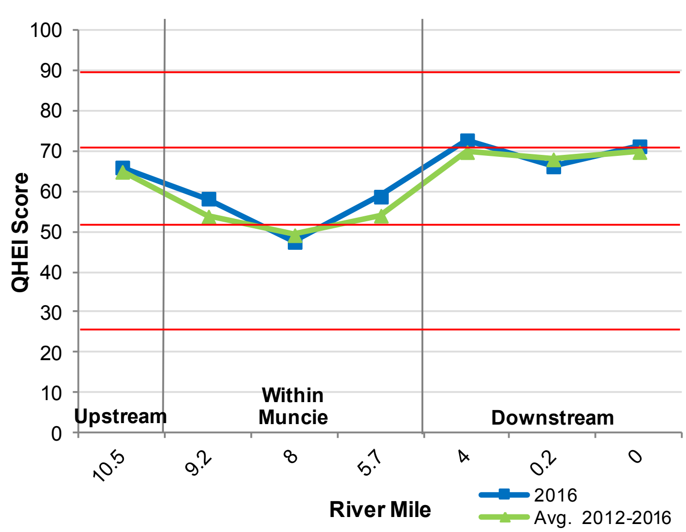
Graph 10.—Tributary H' scores, 2016




Graph 11.—White River QHEI scores, 2016




Graph 12.—Buck Creek QHEI scores, 2016.



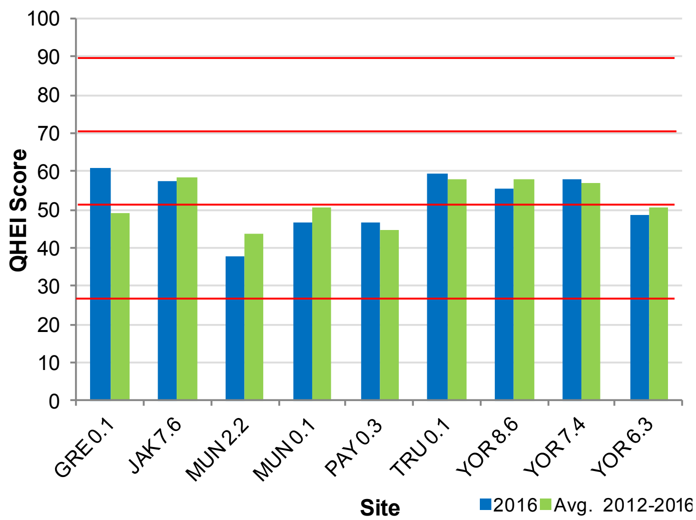

0.91 (WHI 326.9). Mean J' scores (Appendix B, Table 12) worsened downstream of the city limits. White River "Percent Dominance of Top Three Taxa" (Appendix B, Table 11) ranged from 0.53 (WHI 304.4) to 0.21 (WHI 326.9). Mean scores (Appendix B, Table 12) worsened as White River progressed downstream of Muncie. White River "Percent Chironomidae" (Appendix B, Table 11) ranged from 0.49 (WHI 317.2) to 0.05 (WHI 320.1 and WHI 304.4). Mean scores (Appendix B, Table 12) worsened as White River progressed downstream.

**Buck Creek:** Buck Creek J' scores (Appendix B, Table 11) ranged from 0.66 (BUC 14.9) to 0.95 (BUC 5.9). The mean Buck Creek J' score (Appendix B, Table 12) was 0.80. Buck Creek "Percent Dominance of Top Three Taxa" (Appendix B, Table 11) ranged from 0.68 (BUC 14.9) to 0.26 (BUC 15.2), with a mean of 0.40 (Appendix B, Table 12). Buck Creek "Percent Chironomidae" scores (Appendix B, Table 11) ranged from 0.26 (BUC 9.2) to 0.01 (BUC 14.9), with a mean of 0.10 (Appendix B, Table 12).

**Smaller Tributary Sites:** York Prairie Creek J' scores (Appendix B, Table 11) ranged from 0.65 (YOR 8.6) to 0.86 (YOR 7.4). The remaining smaller tributary sites ranged from 0.69 (EAG 0.3) to 0.88 (BEL 1.0). "Percent Dominance of Top Three Taxa" at York Prairie Creek (Appendix B, Table 11) ranged from 0.67 (YOR 8.6) to 0.36 (YOR 7.4). The remaining smaller tributary sites ranged from 0.66 (EAG 0.3) to 0.25 (YFM 1.0). "Percent Chironomidae" (Appendix B, Table 11) ranged from 0.10 (YOR 7.4) to 0.00 (YOR 6.3) at sites on York Prairie Creek. Remaining smaller tributary sites ranged from 0.42 (KIL 20.1) to 0.00 (BEL 1.0).

*QHEI:* **White River:** White River QHEI scores ranged from 48.3 (WHI 315.0) to 81.5 (WHI 306.5), *Poor* to *Good* (Graph 11 and Appendix B, Table 11). Mean scores worsened within Muncie city limits, but recovered downstream (Appendix B, Table 12). A significant increase in scores was seen

Graph 13.—Smaller tributary QHEI scores, 2016.




from 2011-2016 at WHI 313.5 ($R^2 = 0.79, P < 0.05$), and from 2009-2016 at WHI 313.4 ($R^2 = 0.95, P < 0.01$).

**Buck Creek:** Buck Creek QHEI scores (Graph 12 and Appendix B, Table 11) ranged from 47.25 (BUC 13.8) to 72.75 (BUC 4.0), *Poor* to *Good*, with a mean score of 62.02, *Fair* (Appendix B, Table 12). A significant increase in scores was seen from 2013-2016 at BUC 0.0 ($R^2 = 0.90, P < 0.05$).

**Smaller Tributary Sites:** York Prairie Creek QHEI scores (Graph 13 and Appendix B, Table 11) ranged from 48.75 (YOR 8.6) to 58.0 (YOR 6.3), *Poor* to *Fair*. QHEI scores (Graph 13 and Appendix B, Table 12) from the remaining smaller tributary sites ranged from 37.75 (MUN 2.2) to 68.25 (BEL 1.0), *Poor* to *Good*.

## DISCUSSION

**Mussels.**—Sampling results at WR 313.4 continue to indicate good water quality in this stretch of White River, impressive considering the urban location of this site. The significant increase in Unionid diversity suggests that populations at this site are thriving. The apparent fluctuation in diversity and density through the years is likely a product of random sampling, Therefore, further sampling and examination of sampling design will be necessary to determine if there is a decline in native populations, and if this sampling method remains to be the most accurate and efficient method.

One of the three most abundant mussels found at this site, the elktoe, is considered to be characteristic of streams with good water quality, and intolerant of impoundment (Watters 1995; Parmalee & Bogan 1998). In apparent contrast, this mussel species has been found throughout White River within the City of Muncie, which has many impoundments. However, it is usually found in firm substrate, not the softer substrates directly upstream and downstream of the impoundments.

*Corbicula* spp. density has also fluctuated at this site, appearing to increase in 2016. This is characteristic of invasive species. *Corbicula* spp. populations grow rapidly and are then susceptible to sudden die-offs, generally after reproduction, sudden changes in water temperature, and low dissolved oxygen (Strayer 1999b). *Corbicula* spp. will continue to be monitored in order to establish trends in population numbers and correlations with Unionid populations.

It has been noted that one mussel species, the white heelsplitter *Lasmigona complanata*, has not been found in White River upstream of Muncie. This species' opportunistic nature, and its ability to tolerate silt, habitat disturbance, and impoundments (Grabarkiewicz & Davis 2008), appear to make it an ideal species to inhabit White River within city limits. However, it is possible that this species is unable to expand its range upstream due to the inability of its host species to navigate the five impoundments within Muncie city limits. Dams are well documented as obstacles to mussel population abundance and expansion (Vaughn & Taylor 1999; Watters 2000; Dean et al. 2002). Habitats are altered upstream and downstream of the impoundment, resulting in an increase of pollutants, siltation, stagnation, thermal changes, and anoxic conditions (Watters 1999), causing additional complications for mussel populations (Watters 1996; Dean et al. 2002; Lessard & Hayes 2003; Tienmann et al. 2004; Poff et al. 2007; Maloney et al. 2008). Dams have been implicated as one of the leading causes of current-day decline in freshwater mussel populations in North America (Parmalee & Bogan 1998; Haag 2009). They have been cited as being responsible for the "local extirpation of 30-60% of the native freshwater mussel species in many United States rivers" (NRCS 2009). Studies have shown that the impacts of impoundments have resulted in reduced abundance, diversity, and species richness of mussel fauna (Dean et al. 2002; Baldigo et al. 2004; Tiemann et al. 2004; Santucci et al. 2005; Galbraith & Vaughn 2011: Tiemann et al. 2016).

Additional future considerations for mussel sampling at the BWQ include initial sample size, condition variable, and final sample size determination at BWQ mussel sites. Condition variables used in adaptive cluster sampling fluctuate among studies from 5-30% of the highest typical number found during a preliminary survey (Strayer and Smith 2003). Trial and error will likely be the best way to determine the optimum condition variable for each site. Through research of the newest methods and possibly trial and error, the best approximation of the condition variable will be attained. Research will also be focused on the introduction of a stopping rule, to prevent the nearly infinite sampling of a site. Investigation into statistical methods that will accurately determine population numbers for individual species when using adaptive cluster sampling will also be re-examined. This will enable us to further investigate the possible effects of water or habitat quality on a species level.

There is also continued concern about wide confidence intervals at mussel sites. It has been found that estimates of mussel population density tend to be skewed (Philippi 2005), making the usual approach to confidence intervals inaccurate. It appears that generally, these are found when populations are highly variable, common in *Corbicula* spp. populations. These limitations will be considered when contemplating further sampling and analytical strategies.

**Macroinvertebrates**—Many sites had lower mIBI scores in 2016. Most of these sites also had unusually low abundance and/or diversity.

*Poor* mIBI scores at some sites may be attributed to a lack of suitable habitat for macroinvertebrates, quantified by *Poor* QHEI scores. Sites at MUN 0.1, WHI 315.0, and YOR 8.6 all had *Poor* QHEI scores, indicating that a lack of habitat may limit the macroinvertebrates that can inhabit these sites.

A few sites stand out as having possible impairment due to organic issues. Sites that are of concern include BUC 4.0, JAK 7.6, and YOR 8.6. These sites not only had *Poor* mIBI scores, but also had *Fairly Poor* HBIs, which are more sensitive to organic pollution.

Further evidence supports the effects of an organic stressor on BUC 4.0. Not only does this site have the worst mIBI score in 2016, but also a *Fairly Poor* HBI score and low diversity. The *Good* QHEI score indicates that habitat is not likely a limiting factor at this site.

Organic impairment is also suspected at JAK 7.6 based on multiple indicators. This site had the worst HBI score, a *Poor* mIBI score, and low abundance. The sample was dominated (20.2%) by the tolerant *Ischnura* spp., with most other taxa having very few individuals represented. Tolerant organisms comprised 26% of the sample. The Fair QHEI indicates that habitat is not limited, and HBI results suggest that the stressor is likely organic.

YOR 8.6 results suggest habitat limitations as well as organic impairment. mIBI, HBI, and QHEI scores were all *Fairly Poor* or *Poor* at this site in 2016. This site also had low diversity, and was highly dominated by three species. Temporal trends also suggest a likely combination of organic impairment and habitat limitations.

Organic impairment appears to be a likely stressor at YOR 7.4. This site is highly abundant with decent diversity and *Fair* mIBI and QHEI scores. However, a *Fairly Poor* HBI score due to a dominance (36.0%) of three tolerant organisms, and a near absence (7.08%) of intolerant organisms, indicates organic stressors may be present. This is also supported by the significant increase in "Number of Dipteran taxa" since 2011. This suggests that organic enrichment is not only limiting the habitation by intolerant organisms, but allowing tolerant organisms to thrive and dominate in this system.

Many remaining sites with Poor mIBI scores do not suggest organic impairment or habitat limitations. These sites include BUC 7.1, BUC 5.9, BUC 5.7, BUC 0.9, BUC 0.2, TRU 0.1, WHI 311.2, WHI 310.7, and WHI 308.7. Most of these sites have very low abundance and/or diversity, exaggerating any effects on this sample (BUC 7.1, BUC 5.9, TRU 0.1, and WHI 311.2). BUC 5.7 is dominated by non-insects (39.72%) and tolerant taxa (34.75%), with the tolerant *Physa spp.* representing 31.9% of the sample. However, temporal trends show a significant increase in "Number of EPT Taxa" which is promising. BUC 0.9 is abundant and highly diverse, but is dominated by three organisms, with Physa comprising 24.8% of the sample, and tolerant organisms comprising 25.5% of the sample. BUC 0.2 is highly abundant and diverse, but 59.4% of the sample is dominated by four taxa, and 37.8% of the sample is tolerant. WHI 311.2 has low abundance, but is fairly diverse: however, representation in the EPT taxas are very low. Non-insects comprise 51.8% of the sample, with gastropods representing 36.5% of the sample. This site also has significant negative temporal trends in the "Number of taxa" and "Percent Tolerant Taxa" submetrics. WHI 310.7 is fairly abundant and diverse, but is dominated by three taxa (47.2%), with few organisms comprising each of the remaining taxa. However, two of the three dominant taxa at this site are intolerant taxa, resulting in the seemingly contrasting *Very Good* HBI score. WHI 308.7 is highly abundant and diverse, however, three taxa comprise 44% of the sample, and 23% of the sample is from the tolerant family Chironomidae. This site is also dominated by non-insects (47.4%,), with gastropods comprising 33.3% of the sample.

When looking at mIBI scores since implementation of IDEM's methods, we can see that in each year, multiple sites are considered "Poor". Some tributary sites are fairly consistent in this ranking. While this is in contrast to the more positive HBI scores we were used to, the multi-metric index may be picking up disturbance to submetrics, thus giving us a better picture of water quality at these sites.

Observed trends give us some indication of negative impacts on sample sites. Negative mIBI scores generally are not seen on White River upstream of Muncie city limits, likely indicating a negative impact from the anthropogenic sources of an urbanized area (ie– storm water, impervious surface, CSOs, impoundments, etc.). Multiple negative mIBI scores at tributary sites likely reflect impacts that are more apparent due to their smaller size.

Climatological fluctuations and extremes (such as the drought in 2012 and flooding in 2015) have been considered as factors in years with unusually low mIBI scores (Bowley 2012; Bowley 2015). We also need to consider that other stressors may need to be considered including the effects of multiple stressors. These may include ecological, morphological, hydrological, biological, chemical or climatological effects. To complicate an already challenging situation, most aquatic macroinvertebrates have complex life cycles that include multiple stages, some being terrestrial. Research and analysis, as well as continued monitoring, will be conducted in an attempt to determine all that is affecting macroinvertebrate communities.

Dramatic improvements have been seen since the inception of our macroinvertebrate and mussel sampling programs. Point source pollutants have been controlled through the utilization of local permits regulated by the Bureau of Water Quality. Improvements have been and continue to be made to our Water Pollution Control Facility. Whereas most analyses historically have focused on White River, studying the tributaries and nonpoint source pollution impacting them has become critical. These impacts on water quality include hydromodifications (channelization, impoundments, dredging, and removal of riparian zones), urban storm water (sources include CSOs, SSOs, and impervious surfaces), and sedimentation. In 1990, the US EPA listed sedimentation as the top pollutant of rivers in the United States (Box & Mossa 1999), and it has been determined that reductions in water quality are detectable at > 15% impervious surface (Roy et al. 2003).

This shift in focus would benefit from public outreach, education, and cooperation to instill better management practices throughout Delaware County. These include buffer strips, rain barrels, rain gardens, better construction site practices, and the further separation of CSOs. As improved management practices are implemented, it is expected that water quality will continue to improve.

Overall, the water systems in this area appear to be in good condition, especially considering the industrial, urban, and agricultural areas through which they flow. Efforts by the citizens of

Delaware County, the City of Muncie, the Muncie Sanitary District, the Bureau of Water Quality, and the industrial community are responsible for the improvements in water quality since the BWQ was established in 1972.

## ERRATUM

An error in reporting York Prairie Creek site numbers was discovered in 2016. During these years, YOR 8.6 was reported as York Prairie Creek CR 300W, YOR 7.4 was reported as York Prairie Creek 400W, and YOR 6.3 was reported as YOR Storer. Sites should have reported YOR 8.6 as Storer, YOR 7.4 as 300W, and YOR 6.3 as 400W. Site assessments were correct for each site, however, the site name used was incorrect and has now been rectified.

With the use of more detailed dichotomous keys, it became apparent that there were identification errors in a few previously identified Coleoptera species. *Optioservus* spp. and *Dubiraphia* spp. prior to 2017 were switched, and *Derallus* spp. were likely *Laccobius* spp.

Due to an error in calculation of the submetrics “Percent Sprawlers” and “Percent Orthocladiinae and Tanytarsini” that was corrected in 2014, some resulting mIBI scores from 2009-2013 were altered. Trends included in this report were calculated with corrected scores, and all corrected scores are listed in Appendix B, Table 13.

**Appendix A.—**Mussel sites, taxa identified, graphs, density, Horvitz-Thompson results, and field sheet.