Impacts of Urban River Restoration Structures on the Fish Community in the Ottawa River, NW Ohio

Final Report to the Ohio Lake Erie Commission (Grant SG-455-13)

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Abstract:

Urban rivers are often viewed as prime targets for rehabilitation projects. However, few analyses of the biological and habitat effects of such rehabilitations are published. A 1,500m section of the Ottawa River located on the University of Toledo Main Campus was rehabilitated. Using a before/after – control/impact design, we analyzed the effect of the rehabilitation on the fish community and stream habitat. Eight 20m sites were selected in the river; four control sites and four impact sites, where structures were installed. Each site was sampled twice during low discharge in the summers of 2013 (pre-installation) and 2014 (post-installation). Fish community metrics, collected in blocked-off segments of the river with seines and a backpack shocker, included species presence, diversity, richness, Index of Biotic Integrity (IBI) and spawning condition. Concurrently measured habitat variables included a Qualitative Habitat Evaluation Index (QHEI), surface sediment texture, water depth and flow, water quality and canopy cover. Fish community and habitat metrics were analyzed using poisson and linear models, respectively. Post installation, fish abundance increased in impacted relative to control sites ($p<0.10$). Other fish community variables were not impacted by the rehabilitation and little evidence was noted for the use of structures by specific fish species. QHEI scores in the rehabilitated sites averaged 45.6 and were higher ($p<0.05$) than the values in control sites. Impacted sites in 2014 had greater percentages of fine sands (63µm-0.5mm) than control sites ($p<0.05$). Coarse sand (0.5mm-2mm) decreased across all sites in 2014 ($p<0.001$) concurrent with an increase in fine sand ($p<0.001$), likely due to the much lower rainfall in the summer of 2014. Within each site, the variability of water depths and the heterogeneity of surface sediment texture were not impacted by the rehabilitation. Improvement in QHEI scores coupled with a minimal fish community response suggest a delay between habitat and biological improvements of the river. Moreover, bringing back native species that once inhabited these streams may rely on restoring original habitat features such as woody debris, once abundant in this reach of the Great Black Swamp.

Executive Summary

1. Urban rivers are often viewed as prime candidates for rehabilitation efforts. However, few analyses of the biological and habitat effects of rehabilitation structures have been published. A 1,500m section of the Ottawa River located on the University of Toledo (UT) main campus was the site of such an urban river rehabilitation. A before/after – control/impact (BACI) study design was implemented to analyze the effect of the rehabilitation on the fish community and stream habitat.

2. Eight 20m sites were selected in the river; four control and four impacted sites, where structures were installed. Each site was sampled twice during low discharge in the summers of 2013 (pre-installation) and 2014 (post-installation). Fish community metrics, collected in blocked-off segments of the river with bag seines and a backpack shocker, included species presence, diversity, richness, Index of Biotic Integrity (IBI) and spawning condition. Concurrently measured habitat variables included a Qualitative Habitat Evaluation Index (QHEI), surface sediment texture, water depth and flow, water quality and canopy cover. Fish community and habitat metrics were analyzed using poisson and linear models, respectively.

3. During both summers, a combined 1,737 fish belonging to 33 species (including an Ohio listed species of concern) were caught, identified, checked for gravidity and released. Most common species included bluntnose minnow (*Pimephales notatus*), emerald shiner (*Notropis atherinoides*), round goby (*Neogobius melanostomus*), white sucker (*Catostomus commersonil)* and various centrarchids. Nine species were gravid and juveniles were encountered for 15 species. Non-natives made up 12% of the total catch. For all sites combined, IBI scores averaged 21.3 in 2013 and were significantly higher (26.8) in 2014.

4. Fish abundance increased in impacted relative to control sites in 2014 ($p<0.10$), although all sites had lower fish abundance in the post-installation sampling ($p<0.001$). Other fish community metrics were not impacted by the rehabilitation and little evidence was noted for the use of structures by specific fish species. QHEI scores in the rehabilitated sites averaged 45.6 and were significantly higher ($p<0.05$) than the values in control sites. Impacted sites in 2014 had greater percentages of fine sands (63µm-0.5mm) than control sites ($p<0.05$). Percent coarse sands (0.5mm-2mm) decreased across all sites in 2014 ($p<0.001$) concurrent with an increase in fine sands ($p<0.001$) likely due to the much lower rainfall in the summer of 2014. Within each site, the variability of water depths and the heterogeneity of surface sediment texture were not impacted by the installation of the rehabilitation structures.

5. Observed improvement in QHEI scores coupled with a minimal fish community response suggest a delay between habitat and biological improvements of the river. Monitoring should continue, while efforts should include improvements at a larger spatial scale such as management of storm water, non-native species, and floodplains. Moreover, bringing back native species that historically inhabited these streams may rely on restoring original habitat features such as woody debris, once abundant in this reach of the Great Black Swamp. This study contributes important baseline information on impacts of urban river rehabilitation on a resident fish community. With continued monitoring, long-term fish community responses may emerge and a better understanding of the value of urban river rehabilitations will be achieved. This project provided field training for nearly 600 undergraduate students from freshmen to senior level UT courses. It also formed the basis of a graduate student's thesis and an undergraduate honors project.

Acknowledgements

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Introduction

River restoration projects, or rehabilitations, have become increasingly popular over the last several years with over $1 billion spent on more than 6,000 individual projects from 1999 to 2009 alone (Purcell *et al.* 2002, Miller *et al*. 2010). Despite this increasing investment in rehabilitation efforts, there is no uniform standard of project monitoring to determine functionality of these efforts (Palmer *et al.* 2005). Hassett *et al.* (2005) reported that from a total of 4,700 river and stream restoration projects completed in the Chesapeake Bay watershed, only 5.4% had project performance monitoring programs in place post construction. Similarly, in a survey of nationwide restoration projects, Bernhardt *et al.* (2005) found that only 10% of projects indicated that some form of post monitoring had occurred. With this rapid increase in habitat rehabilitation, there is a need to understand the effects and the success rate of installed structures on the biology of the river, including resident fish populations (Schwartz and Herricks 2007). Some previous studies have analyzed the overall effects of river rehabilitation as well as the impact of individual components of habitat rehabilitation such as the effectiveness of large woody debris (Larson 2001) and bank habitat use by specialized fish species in restored rivers (Pander and Geist 2009). Little analysis has been done, however, on the effects of multiple rehabilitation structures on the resident fish community in an urban river.

In a paper on the underlying principles of ecological restoration, Bradshaw (1996) defined the term restoration as well as five other terms that he believed may be better descriptions of some restoration projects. He defined a restoration as "the act of restoring to a former state or position (…) or an unimpaired or perfect condition." He then defined rehabilitation as "the action of restoring a thing to a previous condition or status." This is similar to a restoration with the crucial difference being that there is no implication for perfection (Bradshaw 1996). Palmer *et al.* (2005) expressed a similar view by stating that project goals should be to transition a river to the least degraded and most ecologically beneficial state possible rather than attempting to return it to some unknown historical condition. Moerke and Lamberti (2004) also noted the unrealistic definition of a restoration and instead use the definition "the process of returning a stream reach to a condition that promotes re-expression of natural ecosystem structure and function." The rehabilitation concept is a more accurate description of a recently completed project located in the Ottawa River at The University of Toledo, the location of this project, and thus, the term "rehabilitation" will be used henceforth.

Some previous projects have documented the effects of river rehabilitation on fish communities. A meta-analysis of multiple rehabilitation projects in the Danube River (Schmutz *et al.* 2013) found a 55% increase in the number of fish species at rehabilitated sites compared to unaltered locations across six rehabilitation projects spanning 19 sites along the river. They also found an increase in the number of species for each habitat guild at all rehabilitated sites. This demonstrated that, on a large scale, rehabilitation efforts may have positive impacts on the fish community. In much smaller rivers, Pretty *et al.* (2003) analyzed fish community responses to 13 in-stream rehabilitation projects. The rehabilitations were completed between 1992 and 1997 with post-installation monitoring carried out in 2000 and 2001. Rehabilitated sites did not contain greater species richness, diversity, or abundance compared to control sites after four to nine years. Bond and Lake (2003) emphasized the importance of creating habitat in improving biotic responses to rehabilitation

efforts. Such efforts should also address physical barriers that may prevent colonization of rehabilitated sites as well as the potential issue of temporal changes in habitat requirements. Some rehabilitation structures, when implemented, may only benefit certain life stages of a species. For example, adult white suckers (*Catostomus commersonii*) may utilize a structure installed in a deep pool, while juveniles located in the shallows would not. If a species is only able to effectively utilize a structure in one life stage, biotic responses to rehabilitations would be limited. Similarly, species not located within the regional species pool, or an immediate geographical range, would likely not establish in rehabilitated sites (Stoll *et al.* 2013).

More research has been published on outcomes of terrestrial than of aquatic rehabilitations. For example, Tews *et al.* (2004) reviewed 85 terrestrial rehabilitation projects and noted that 75 found a significant correlation between increased habitat heterogeneity and increased faunal diversity. They put forth the concept of a "keystone structure" or heterogeneity that, when added to the environment, promoted species diversity at a defined scale of space and time. These keystone structures may provide cover, resources or services crucial for a species' survival. Interestingly, no aquatic studies were included in this paper. The keystone structure, while originally conceptualized as forms of vegetation, may be adapted to the aquatic setting in the form of man-made structures, such as rocks and large woody debris.

By providing additional niches, habitat heterogeneity may be an important feature for fish communities in rivers. The availability of different substrate types, the presence of structures such as boulders or large woody debris, and the formation of riffles, runs, and pools all provide habitat that may promote a variety of ecological niches supporting greater species diversity (Karr *et al.* 1986). Habitat heterogeneity in a river system can also promote increased richness and abundance of macroinvertebrates that, in turn, provides an influx of available resources that can support a more diverse fish community (Power 1992).

Habitat diversity of a river may be quantified with a Qualitative Habitat Evaluation Index (QHEI), an assessment method utilized by the Ohio EPA to score the physical habitat quality of a river system (Rankin 1989). It includes of six assessments, each assigned a score based on the observed presence of listed criteria with a maximum possible score of 100 for the entire assessment. These six assessment areas are substrate, in-stream cover, channel morphology, bank erosion and riparian zone, pool/glide and riffle/run quality, and gradient. Scored river sections are assigned a narrative rating from Very Poor to Excellent based on defined score ranges (Ohio EPA 2006).

Previous research found a positive correlation between QHEI scores and fish community diversity. Thus, QHEI may be an important metric for evaluating the effects of a rehabilitation effort on the biological community (An *et al.* 2004). Sullivan *et al.* (2004) also found significant positive correlations between Index of Biotic Integrity scores and four QHEI assessment criteria (e.g., substrate, in-stream cover, channel morphology, and pool/glide and riffle/run quality). Thus, compiling QHEI scores for a river, such as the Ottawa River located in northwest Ohio and southeast

Michigan, gives researchers and managers the ability to quantify the physical quality of the river's habitat and correlate that habitat with aspects of the fish community.

The Ottawa River watershed is one of eight sub-watersheds located within the Maumee River watershed and is part of the Maumee Area of Concern. It contains a drainage area of 572 km^2 (U.S. ACE 2009). The river has an average gradient of less than $1m \cdot km^{-1}$. It originates in northeast Fulton County (Ohio) as agricultural ditches and flows east through Lucas County where it is joined by North Ten Mile Creek out of Lenawee and Monroe counties in Michigan (Maumee RAP 2006). The river continues to flow westward through Lucas County until it empties into Maumee Bay and western Lake Erie. This Ottawa watershed may be divided into three main reaches: the upstream, middle, and lower reaches. The upstream Ten Mile Creek reach runs from the originating agricultural ditches to river kilometer (RK) 32 in the City of Sylvania, contains indistinct valleys and flood plains and banks 1-3 meters high with land use being primarily agricultural (U.S. ACE 2009). The middle reach falls between RK 32 and RK 12 and has banks rising in excess of 3 meters. This middle reach flows through a Boy Scout Reservation, a Toledo metropark, the Village of Ottawa Hills, the University of Toledo main campus, and Ottawa Park. Land use in the middle reach is largely urban and suburban, with some park conservation. Between 2003 and 2008, two small dams were removed from this reach (Roberts *et al.* 2007, Gottgens *et al.* 2009). The lower reach runs from RK 12 to the mouth where it is under seiche influence from Maumee Bay, which can reverse water flow in that reach. Land use in the lower reach is urban/industrial (Maumee RAP 2006).

The focus of this project is a 1,500m section of the Ottawa River on the main campus of The University of Toledo between river kilometer 16.7 and 18.2. As of 2012, this section of river was found to be home to 46 species of fish throughout the year, including the least darter (*Etheostoma microperca*), a state listed species of concern. This historically meandering stretch of the river had levies built on both sides in the early 1960s producing a straight channel flow. Coupled with the river's high energy during high flow events, this channeling has caused downward scouring and a steepening of the bank slopes. In 1982, a collaborative study was completed on the university section of the Ottawa River to help beautify the river (Kunkle *et al.* 1982). This study analyzed hydrologic properties (e.g., flooding, bank erosion and fluctuating water levels) and included a hydrologic investigation of the river to better understand its functioning. The study was coupled with a concept plan for proposed future developments of the river channel which were never implemented.

In 2009, the UT reach of the Ottawa River was selected for rehabilitation by the University's Commission on the River. In August 2013, seven types of habitat rehabilitation structures were installed throughout the campus stretch of the river (Lawrence 2012). These structures included bendway weirs, hydraulic cover stones, lunkers, locked logs, structural transitioning stones, stone toe protections, and arcing stones formations referred to as "smiles" or "frowns" (Figure 1). They intended to affect water flow, sediment distribution, fish species diversity and community composition, as well as other habitat and biological factors.

In order to understand the effects of these rehabilitation structures on the resident fish community, we carried out a before/after – control/impact (BACI) study. According to Smith (2002), the preferred type of impact assessment to perform in order to best quantify the ecological response to habitat rehabilitation is such a BACI design. Eight sites throughout the five reaches of river on campus were selected for this study. Four of the sites were control sites, where no rehabilitation structures were placed, and four sites were impact sites or sites where structures were placed after our initial sampling in the summer of 2013.

Our overall goal was to quantify the effects of this rehabilitation effort on the Ottawa River habitat quality and its fish community. We hypothesized that the rehabilitation structures would have a positive impact on the fish community in terms of abundance, richness, diversity, and the index of biotic integrity (IBI). We predicted that rehabilitation structures would also lead to an increase in QHEI scores in impacted sites and that those sites would show increased variation in water depth after the rehabilitation compared to control site depth variation. Testing these hypotheses contributes to a better understanding of the short-term effects of river rehabilitation structures on fish communities and stream habitat.

Figure 1. Examples of the installation of river rehabilitation structures in the Ottawa River on the University of Toledo main campus. From upper left to lower right, hydraulic cover stones, lunker, longitudinal peaked toe stone protection, locked log and bendway weir (Photos courtesy of the UT River Commission).

Methods

Site Description

The region of the Ottawa River located on the University of Toledo campus is comprised of five river reaches spanning 1,500 meters (Figure 2). This stretch of the river consists of a straight channel with eroded banks and riparian zones 5-10m in width. Average water depth and discharge greatly fluctuate throughout the year. Periods of peak flow and depth occur during spring months while lowest depths and flow rates generally occur in the summer. The discharge of this section of the river can be characterized as "flashy". For example, water depth and discharge in these five reaches ranged from 0.6 – 3.0m and 0.1 – 33.4$m^3 \cdot sec^{-1}$, respectively between June and August 2014 (USGS 2014) during our second summer of sampling. After a rainfall, the hydrograph will rapidly increase and decrease due to the large acreage of impermeable surfaces and tiled farmland upstream.

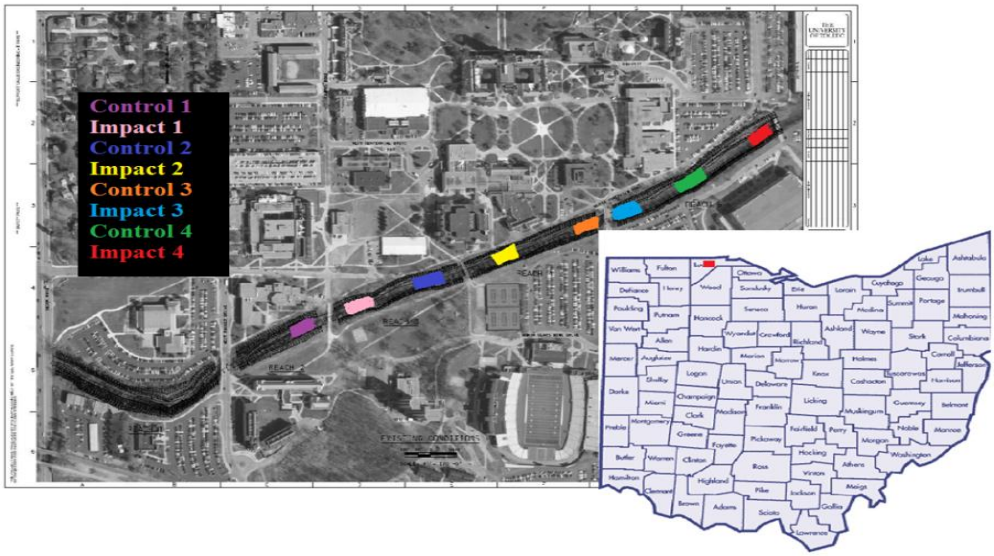

Figure 2. Sample sites in the Ottawa River on the University of Toledo campus. The four control sites did not get rehabilitation structures; the four impact sites had structures installed in August 2013.

Of the five campus reaches, sample sites were chosen from reaches two through five. Reach 1 was excluded from sampling because it did not receive rehabilitation structures. Instead, in 2012 a cutbank was formed on the North slope in that reach to help alleviate potential rises in the 100-year flood level of the river created by the downstream structures. Eight sites, each 20m in length, were selected from reaches 2 through 5. Four sites were control sites and four were impact sites. Impact sites were selected based on planned positions of the different structures placed throughout the river. Control sites were chosen in areas of the river containing no planned structures with as much distance from other structures and sites as possible. Due to the linearity of the river system and the limited length of the rehabilitation, we were not able to control for independence among sites.

Fish Sampling

During both years, sampling began at the most downstream site and moved upstream in order to minimize disturbance of subsequent sites. Before fish sampling, two 9.1m seines were carefully positioned to block each end of the 20m sites to prevent fish entering or leaving the site during data collection. At each site, both before and after the rehabilitation structures were installed, two sampling techniques were used. An electroshocking technique was employed first with a Smith & Root LR24 electrofisher backpack unit (Figure 3). Electroshock settings were maintained at 375 volts, 115 hertz, with an 8% duty cycle for each sampling event. Following electroshocking, a seining technique was utilized to gain a more complete measure of the composition of the fish community. Seining was performed with a 6m bag seine (6.4mm mesh size). Both electroshocking and seining were completed at each site before moving to the next site.

Electroshocking was completed by using a "W" formation that covered 150-200m^2 of each site (Figure 4). Collected fish were identified to species, evaluated for deformations, eroded fins, lesions, or tumors (DELTs), disease, gravidity and age class and released downstream of the sampling area. Seining was then completed using the 6m bag seine by making a "V" formation through the site, also covering 150- 200m^2 (Figure 4). Sampling duration was approximately 8-10 minutes for each method at each site.

A combination of the two sample methods used promoted a more complete fish assessment of each site (Pierce 2010). This is due to the advantages and shortcomings of the two sampling methods. For example, electroshocking, while better suited for areas with structure, is useful for capturing catostomids and centrarchids in more complex habitat, while seining will yield a more accurate representation of darters lacking swim bladders. In addition, electrofishing has proven more effective in areas with woody debris, undercut banks, and boulders where centrarchids and catostomids are more likely found and where seines have less access (Pierce 2010). The technique is less effective in large pools that may hold cyprinids where seining is more effective. All sampling was conducted during low flow (<1 $m^3 \cdot sec^{-1}$) periods for safety reasons and because the electroshocking backpack unit was designed only for use in wade-able conditions.

Several fish community response variables were calculated from the collections. These included species presence, Shannon diversity, species richness, and an index of biotic integrity (IBI). The latter was modified from the standard IBI, because we evaluated only 20m instead of the prescribed 200m sites. Twelve criteria were scored including total number of species, darter species, sunfish species, sucker species, intolerant species, % tolerant, % omnivores, % insectivores, % top carnivores, % simple lithophils, % DELT anomalies, and fish abundance (Ohio EPA 2014). Each metric was assigned a score based on the IBI reference charts for a sample area from a 50m^2 drainage area. Tolerance group, breeding group, and feeding group were also determined for each species caught according to the Ohio IBI protocol in order to correctly score each criterion. All sites were given a calculated IBI score for 2013 and 2014. Possible scores ranged from 12 to 60 and were assigned a rating from Very Poor to Exceptional (Table 1).

Figure 3: Electroshock sampling of a site in the Ottawa River on the University of Toledo campus in 2013 (Photo credit: Todd Crail).

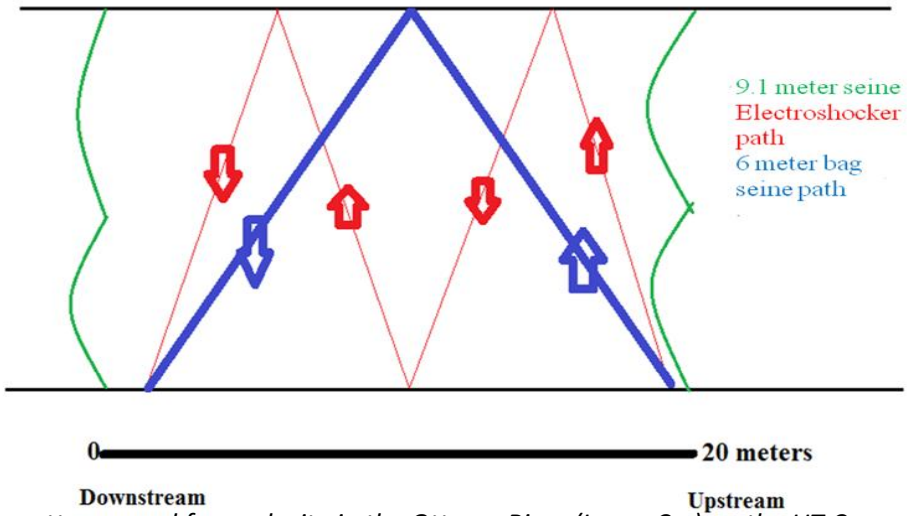

Figure 4: Sampling patterns used for each site in the Ottawa River (Lucas Co.) on the UT Campus during the summers 2013 of 2014. The red line indicates sampling pattern for electro shocking and the blue line indicates sampling pattern for seining. Colored arrows indicate direction of sampling.

Habitat Sampling

Habitat variables, such as QHEI, surficial sediment texture, water depth, pH, dissolved oxygen, conductivity, water temperature, canopy cover, and water flow were also monitored. Our QHEI protocol was modified from Ohio EPA (2006) in that we used 20m sites instead of 200m sites. QHEI has been adopted by many mid-western state agencies and shown to provide a reliable assessment of stream ecological condition (Moerke & Lamberti 2004). QHEI scores are assigned ratings by Ohio EPA ranging from Very Poor (<30) to Excellent (≥75) (Table 2).

Gage height and river discharge were recorded via the USGS gaging station 04177000 to check for consistency across sampling periods. Water depth and flow rate measurements (N=25 per site) were taken at each site (with a Marsh-McBirney Flo-mate 2000) during both years in June and July. Each site was split into five transects from which five depth and flow measurements, spaced evenly across each transect, were taken (Figure 5). Depth measurements were standardized in order to account for differences in the river gage height between sampling periods. Standardization was performed by calculating the average depth of each site from each sampling period and subtracting the difference in average between sampling periods from each recorded depth measurement from the sampling period with the greater average depth. A coefficient of variation was then calculated for each set of 25 water depth measurements from each site in June and July for both years in order to analyze within-site variation of depth between years. Within-site depth variation was monitored in order to quantify changes in a site's depth variation after implementation of the rehabilitation structures. Such changes to the river could impact the resident fish community.

Table 1: IBI score ranges and their assigned ratings for the Huron Erie Lake Plain (HELP) ecoregion (Ohio EPA, 1988, 2014).

Rating	IBI Range
Exceptional	50-60
Very Good	46-49
Good	40-45
Marginally Good	36-39
Fair	28-35
Poor	18-27
Very Poor	12-17

Table 2: QHEI score ranges and their assigned rating (Ohio EPA2006).

Rating	QHEI Range
Excellent	≥ 75
Good	60-74
Fair	45-59
Poor	30-44
Very Poor	< 30

To quantify surficial sediment composition, seventy-two surface sediment cores were taken each year from the eight sites (e.g., N=9 per site). Each site was split into three transects and three samples were taken from each transect using polycarbonate corers with a 7.5cm in diameter. Only the biologically-active zone of the sediment was collected, which we defined as the depth of the alluvium substrate. Core depths ranged from roughly 5 to 15cm, depending on the firmness of the substrate. Transects were located along the north and south banks and in the middle of the channel (Figure 5). All core samples were air-dried and pulverized using a mortar and pestle. Prior to pulverizing, coarse organic debris was removed from each sample by hand and weighed separately. Sediments were then sieved into four separate size classes based on the Wentworth Scale (Wentworth 1922); pebble (>2mm), very coarse and coarse sand (0.5mm - 2.0mm), fine and very fine sand (63um - 0.5mm), and silt or clay (<63um). The weight of each size class for every sediment core was recorded as percent of the total weight of the entire sample.

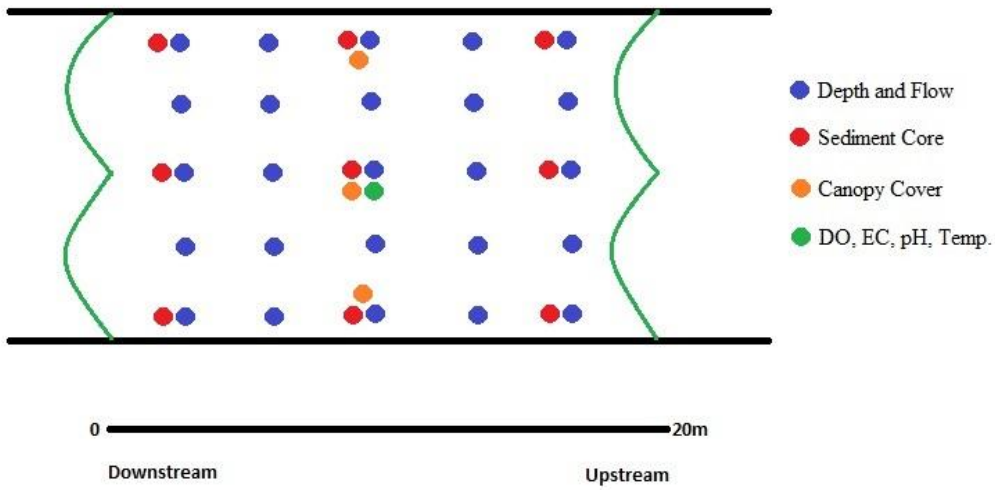

Figure 5: Locations of environmental data collected from each site in the Ottawa River (Lucas. Co.) on the UT Main Campus during 2013 and 2014. Data collected include: sediment particle size, canopy cover, dissolved oxygen (DO), electrical conductivity (EC), pH, water temperature (C°), water depth (cm), and flow (m/s).

Four maps were produced to visualize sediment data using ArcGIS software version 10.1(ESRI 2013). Data points for each site were entered as 25 coordinate points with a depth measurement assigned to each point. Nine points were also assigned sediment data consisting of the percent of each particle size class (on a weight basis) and displayed as pie charts. The depth data were applied to a kriging interpolation technique in ArcGIS to create depth distribution areas.

Data Analysis

The fish community and habitat data were analyzed with regression models. Due to the nature of the fish community data, a poisson regression was used to model species richness, abundance, diversity, and IBI. Poisson regressions are used when analyzing count data such as fish species abundance or richness (Qian, 2012). Count data can rarely be transformed to normality and are assumed to display a poisson distribution. Linear regressions models were performed to analyze the QHEI scores, sediment data, and site water depth variation to identify significant changes in these variables between control and impact sites in 2014.

Three predictor variables were included in the models; Site, Year, and the interaction term Site:Year. QHEI was removed as a predictor variable for the fish community data because the variance accounted for by QHEI was insignificant and indistinguishable from the variance due to time. In the model equation below, predictor variables are summed due to the additive nature of the coefficients. Each variable has an effect that is added to β_0, which represents the value of the response variable at control sites in 2013. RV is a response variable and β_0 is the model intercept.

$\beta_{1,s}$, $\beta_{2,y}$ and $\beta_{3,sy}$ are the predictor variables. $\beta_{1,s}$ is the Site variable and signifies the difference between control and impact sites in 2013. $\beta_{2,y}$ is the Year variable and represents the change in a response variable between 2013 and 2014 control sites. $\beta_{3,sy}$ is the Site:Year interaction term and represents the difference in a response variable between control and impact sites in 2014. The final term, ε, represents unaccounted model error.

$$RV = \beta_0 + \beta_{1,s} + \beta_{2,y} + \beta_{3,sy} + \varepsilon$$

Poisson regression in conjunction with the BACI assessment have been used in similar ecological studies. Lance *et al.* (2005) used a poisson regression to analyze data collected from a BACI design researching the effects of timber harvest and spruce beetle populations on a species of vole in Alaska. Moland *et al.* (2013) used a BACI design to evaluate the changes through time in lobster populations from both protected and unprotected regions. Their data analysis included the use of a poisson regression to test for differences in catch per unit effort between the regions. All our statistical analyses were performed using the program R, an open source statistical package (R Core Team, 2012).

Results & Discussion

Fish Community Responses

We predicted the rehabilitation structures would positively impact species richness, abundance, IBI, and SDI. The poisson model revealed that Site:Year, the interaction term, was a significant predictor of abundance ($p<0.10$) with a positive coefficient indicating an increase in abundance between control and impacted sites in 2014 (Table 3). A 0.10 significance level is often deemed acceptable in ecological studies with their inherent variability in field conditions and tolerance for an increased risk for Type I error (Toft and Shea 1983, Steidl *et al.* 1997). Inherent to the BACI approach, we accounted for observed differences in control site abundance between years. There were fewer fish sampled in control sites during 2014 than the previous year so this difference was included in 2014 impact site abundances. Site:Year was not found to be a significant predictor of richness, evenness, IBI, or SDI.

Year was a significant predictor of abundance ($p<0.01$) and IBI scores ($p<0.10$) (Table 3). Year had a negative coefficient in the abundance model and a positive coefficient value in the IBI model. In other words, abundance decreased and IBI increased from 2013 to 2014 in the control sites. No significant predictors were identified in the species richness, SDI, and species evenness models. Box and whisker plots demonstrate the significant difference between 2014 control and impact site abundances, and between 2013 and 2014 IBI scores (Figure 6). Quartiles and medians for the plots were calculated by compiling June and July samples from the same year, resulting in four calculations for control and impact sites each year for each response variable.

Table 3: P-values and coefficients of poisson regression models for abundance, species richness, Index of Biotic Integrity (IBI), Shannon Diversity Index (SDI), and species evenness. Italicized numbers represent statistically significant p-values.

	p-Value		
	Site	Year	Site:Year
Abundance	0.386	*0.005*	*0.069*
Richness	0.557	0.270	0.690
IBI	0.443	*0.094*	0.949
SDI	0.733	0.787	0.806
Evenness	0.957	0.693	0.984
	Coefficient		
	Site	Year	Site:Year
Abundance	-0.049	-0.271	0.151
Richness	-0.116	-0.223	0.116
IBI	-0.118	0.236	-0.013
SDI	-0.195	-0.153	0.201
Evenness	-0.053	0.352	0.026

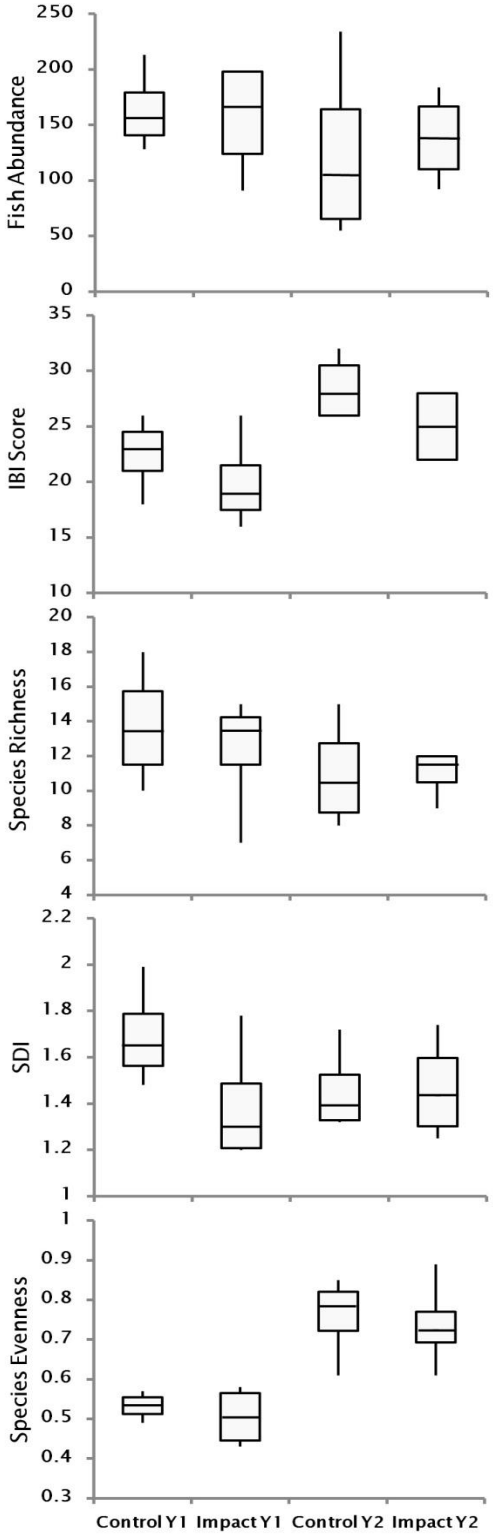

Figure 6: Box and whisker plots of abundance, IBI, richness, SDI, and evenness for control and impact sites in 2013 and 2014. Boxes represent the first and third quartiles. Horizontal lines within the boxes are median values while whiskers represent minimum and maximum values.

The time factor (Year) represented observed variation not attributable to the rehabilitation structures. It accounted for the majority of variance observed in fish community response variables. The relatively low replication, four sampling events for each site over two summers, may increase the effect of time-related variance (Booth 2005, Jahnig *et al.* 2010, Schiff *et al.* 2011, Bice *et al.* 2013). Abundance, richness and diversity of a system such as the Ottawa River vary considerably throughout a single year and interannually.

Spatial scale of the rehabilitation may also have limited the observed fish community response to the structures. In a meta-analysis consisting of 19 rehabilitated sites, Schmutz *et al.* (2013) found that biological responses to rehabilitated reaches were largely dependent upon spatial extent of the project. In fact, greatest increases to species richness were seen in river projects which were longer than 3.9km. Thus, the small size of this Ottawa River rehabilitation project may have affected the fish community response. Furthermore, regional species pools are important to consider in terms of biotic responses to a rehabilitation. Stoll *et al.* (2013) recently found that 91% of fish species present in a rehabilitated site were located in regional species pools within 5km of that site. Essentially, a rehabilitated site is not likely to bring back fish species that do not already occur within the hydrologic network and certain geographic range of the site. To underscore this, we compiled a regional species list (Appendix B) that includes additional connected aquatic systems (e.g., Maumee River, Swan Creek, the Ottawa River and the western basin of Lake Erie). The listing contains available data from the Ohio EPA and the University of Toledo (Ohio EPA 2008a, Grabarkiewicz and Crail 2009). All 33 species (plus 3 hybrid species) collected during this project had been previously documented within this regional species list. A total of 86 fish species were identified in the regional list and of those, 53 were not found in the Ottawa River during our research. These species represent a reservoir of potential inhabitants for the UT campus reach of the Ottawa River. Researchers who continue to monitor the Ottawa River should consider this regional species list when defining project goals and evaluating future fish community responses to the rehabilitation. Moreover, future attempts to reintroduce species from this regional pool into the Ottawa should carefully consider the historical conditions of the river. Prior to the mid-1800s, the Ottawa River was part of the Great Black Swamp and likely contained large amounts of woody debris (Sampson 1930). Rehabilitation focused on reintroducing woody debris may more readily elicit a positive response from regional fish species.

Structure specific field observations

Specific rehabilitation structures may promote the richness and abundances of certain species within impact sites relative to control sites. For example, the “smile” structure located in the Impact 4 site, the locked logs in Impact sites 1 and 3, and the bendway weirs in Impact sites 2 and 4, may all promote more darter species. We found no evidence, however, of an increase in darters at these impacted sites with only 2-4 species present per site in variable abundances (from 2-20 individuals per site). Similarly, increased sunfish richness and abundance was expected in impacted sites. Again, the data do not show such trend with little difference in richness (2-4 species) and abundances (10-12 sunfish) between control and impacted sites. Of note here is that darter abundance for all control and impacted sites dropped from 5.3 to 3.4 darters per site between 2013

and 2014, perhaps masking an effect. The same applied to sunfish abundance dropping from 8.1 to 5.1 fish per site between 2013 and 2014. Noteworthy as well is that impacted sites combined tripled their number of piscivorous fish from 2013 to 2014 while these abundances changed little in the control sites. Impact site 1, where a lunker was installed, contained both a juvenile northern pike and a sub-adult largemouth bass in 2014. That site had no piscivores in 2013.

Some additional evidence of structure use by fish was observed during 2014 sampling. Specifically, at the Impact 1 site, where a lunker structure is located, multiple fish were collected from inside and nearby the lunker using the backpack electroshocking unit. This included a northern pike, green sunfish and an adult spotted sucker. The latter is listed as a sensitive species in Ohio. Another northern pike was sampled in the Impact 2 site, while no northern pike were collected from any of the control sites. These field observations suggest that the Ottawa River fish community may be starting to respond to the presence of the rehabilitation structures and may begin to utilize them as part of their habitat.

Overview of the Fish Community Composition in the summers of 2013 and 2014

A listing of all fish species recorded for both control and impact sites in 2013 and 2014, their scientific name and authority is provided in Appendix A. Excluding juveniles, a total of 1,004 fish were caught during the first year and 733 fish were caught in the second year. Average abundances, excluding juveniles, per site per sampling in 2013 and 2014 were 63 (SD=35) and 46 (SD=36), respectively. Juveniles were not detected with electroshocking due to their small body size. During seining, they were either collected by chance in great abundance or not collected at all. This led to a disproportionate impact of the juveniles on abundance and diversity measurements prompting us to remove them from abundance and diversity calculations. Species richness for the first year of sampling ranged from five to 15 species caught at each site (Table 4). Average richness of all sites in 2013 and 2014 was 13 and 11 species, respectively. The average IBI score for all eight sample sites in 2013 was 21.3 (SD=3.8) while site scores ranged from 16 (Very Poor) to 26 (Poor). In 2014, the average IBI score for all sites increased to 26.8 (SE=3.5) with individual site scores ranging from 22 (Poor) to 32 (Fair). IBI scores were significantly higher in 2014 (paired t-test, $p<0.05$) across all sites.

Twenty-six species were collected in 2013, while 2014 had a total richness of 29 species. Five gravid species were sampled in 2013, including bluntnose minnow (*Pimephales notatus*), creek chub (*Semotilus atromaculatus*), spotfin shiner (*Cyprinella spiloptera*), tadpole madtom (*Noturus gyrinus*), and yellow perch (*Perca flavescens*) (Table 5). In 2014, six species were identified as gravid; bluntnose minnow, common carp (*Cyprinus carpio*), fathead minnow (*Pimephales promelas*), redfin shiner (*Lythrurus umbratilis*), round goby (*Neogobius melanostomus*), and spotfin shiner. Juveniles of 12 and 11 species were sampled in 2013 and 2014, respectively. We noted the presence of juveniles per site as an indication of the use of that site as fish nursery habitat (Table 5).

Table 4: Species richness, abundance, Index of Biotic Integrity (IBI), species evenness, and Shannon Diversity Index (SDI) scores for eight sites in the Ottawa River (Lucas Co.) on UT Main Campus during 2013 and 2014 sampling.

	Control 1		Control 2		Control 3		Control 4		Impact 1		Impact 2		Impact 3		Impact4	
2013	June	July	June	July	June	July	June	July	June	July	June	July	June	July	June	July
Richness	9	6	6	11	11	14	15	14	5	7	8	10	10	13	10	9
Abundance	90	16	56	53	62	42	147	55	31	32	42	54	85	75	127	37
IBI	22		18		24		26		16		20		26		18	
Evenness	0.65	0.85	0.73	0.78	0.52	0.83	0.66	0.83	0.58	0.75	0.6	0.5	0.79	0.68	0.62	0.61
SDI	1.43	1.52	1.31	1.87	1.25	2.19	1.79	2.19	0.93	1.47	1.25	1.16	1.81	1.75	1.44	1.34
	Control 1		Control 2		Control 3		Control 4		Impact 1		Impact 2		Impact 3		Impact4	
2014	June	July	June	July	June	July	June	July	June	July	June	July	June	July	June	July
Richness	4	8	4	8	7	11	10	11	5	10	6	10	7	7	7	10
Abundance	8	13	15	15	75	143	63	70	22	19	58	57	39	13	38	85
IBI	26		26		30		32		28		28		22		22	
Evenness	0.58	0.41	0.3	0.5	0.25	0.31	0.41	0.4	0.27	0.41	0.23	0.41	0.47	0.65	0.39	0.36
SDI	1.21	1.7	0.82	1.82	1.08	1.57	1.69	1.74	0.84	1.65	0.93	1.7	1.71	1.77	1.41	1.69

Table 5: Number of fish sampled in each age class, species gravidity (yes or no), and number of fish sampled displaying DELTs (diseased, eroded fins, lesions, or tumors) collected from the Ottawa River (Lucas. Co.) on the UT Main Campus during 2013 and 2014.

	2013					2014				
	Adult	Sub-adult	Juvenile	Gravid	DELTs	Adult	Sub-adult	Juvenile	Gravidity	DELTs
Black Crappie	0	0			0	0	1			0
Blackside Darter	5	0	Yes		0	1	0	Yes		0
Bluegill Sunfish	20	11	Yes		2	0	1			0
Bluegill/Green Sunfish Hybrid	0	0			0	1	0			0
Bluntnose Minnow	497	7	Yes	Yes	180	124	7	Yes	Yes	15
Brown Bullhead	0	0			0	2	0			0
Central Stoneroller	0	0			0	4	0	Yes		0
Common Carp	2	1			1	4	0		Yes	0
Common Carp/Goldfish Hybrid	0	1			0	0	0			0
Common Shiner	1	0			0	0	0			0
Creek Chub	3	4		Yes	3	0	3	Yes		0
Emerald Shiner	114	0			1	269	9			0
Fathead Minnow	5	5			0	1	0		Yes	0
Freshwater Drum	1	1			1	0	0			0
Gizzard Shad	0	0			0	3	96			0
Golden Redhorse	3	2			0	0	0			0
Green Sunfish	10	1	Yes		0	0	5			0
Johnny Darter	2	1	Yes		0	0	1	Yes		0
Largemouth Bass	3	5	Yes		0	0	21	Yes		0
Least Darter	1	0			0	1	0			0
Logperch Darter	3	0	Yes		2	2	0			0
Northern Pike	0	0			0	2	0			0
Orangespotted Sunfish	0	0			0	1	0			0
Orangespotted/Pumpkinseed Hybrid	0	0			0	0	1			0
Pumpkinseed Sunfish	17	10	Yes		2	4	28			0
Quillback Carpsucker	0	0	Yes		0	0	0			0
Redfin Shiner	27	1	Yes		0	1	0	Yes	Yes	0
Round Goby	123	2	Yes		0	61	12	Yes	Yes	0
Sand Shiner	0	1			0	0	0			0
Spotfin Shiner	10	0		Yes	1	13	2		Yes	0
Spotted Sucker	1	1			0	2	0			0
Suckermouth Minnow	0	0			0	0	0	Yes		0
Tadpole Madtom	0	2		Yes	0	0	0			0
White Crappie	0	0			0	0	3			0
White Sucker	68	6	Yes		28	3	45	Yes		6
Yellow Perch	7	5		Yes	2	0	1			0

The 2013 community was dominated by bluntnose minnows and round gobies, which combined made up 50% of the total catch (Table 5). The fish community in 2014 largely consisted of bluntnose minnows and emerald shiners (*Notropis atherinoides*) which accounted for 54% of the total catch, not including juveniles. Two non-native species were sampled in 2013, common carp and round goby, as well as one hybrid common carp x goldfish (*C. carpio x C. auratus*). In 2014, three non-native species were collected; common carp, orangespotted sunfish (*Lepomis humilis*), and round goby. Two hybrids were also collected in 2014; bluegill x green sunfish (*L. macrochirus x L.*

cyanellus) and orangespotted x pumpkinseed sunfish (*L. humilis x L. gibbosus*). Round gobies made up 98% and 96% of the non-natives in 2013 and 2014, respectively. All non-native species combined accounted for 13% of the total catch in 2013 and 9% in 2014. Eleven species showed DELTs in 2013 while only two species were observed to have DELTs in 2014. In 2014, three species were sampled that were not previously included on the Ottawa River campus species list (Appendix B). These included black crappie (*Pomoxis nigromaculatus*), brown bullhead (*Ameiurus nebulosus*), and white crappie (*Pomoxis annularis*). Eighteen species were not sampled in either year, but were previously collected in the Ottawa (Appendix B).

Overall, the fish community we encountered during the summers of 2013 and 2014 was similar to fish communities of other urban rivers (Karr *et al.* 1985, Wang *et al.* 2000, Miltner *et al.* 2004, Schwartz and Herricks 2007) with an overall richness of 33 species dominated by tolerant species, such as bluntnose minnow and emerald shiner. Urban rivers often experience frequent high flow or low flow events indicative of a flashy hydrograph. These altered periods of flow, along with stream channelization and the presence of sewage outflow and stormwater runoff, contribute to low diversity and increased abundances of tolerant species (Moore *et al.* 1995, Konrad and Booth 2005). The community also contained species that would not typically be associated with an urban river. The least darter (*Etheostoma microperca*), a state listed species of concern in Ohio, and the spotted sucker (*Minytrema melanops*) were both present in our collections. The presence of such sensitive species may be demonstrative of an urban river fish community that is showing signs of recovery.

Habitat Variables

Rehabilitation structures produced improved QHEI scores of the impact sites relative to control sites. A linear model of QHEI scores revealed that the Site:Year interaction term had a positive coefficient and was statistically significant ($p<0.05$) (Table 6). Box and whisker plots (Figure 7) illustrate the significantly higher QHEI scores in 2014 impact sites and demonstrate the ability of such a rehabilitation effort to make a short-term and quantifiable improvement to river habitat. Moerke *et al.* (2004) also reported that rehabilitated sections of river consistently received higher QHEI scores than control sections over a five year period. The ability of rehabilitation structures to significantly improve the habitat of a river was also documented by Rosi-Marshall *et al*. (2006).

We also noted an unintended negative effect of the rehabilitation project on impact site QHEI scores. Bank vegetation was removed from impacted sites to allow access for construction equipment. At the reach three impact site, for example, vegetation was removed from the north bank in order to install the lunker structure which spans the entire 20m of the site. Plant removal was only performed on sites where rehabilitation structures were installed. This led to increased erosion rates of the banks somewhat depressing QHEI scores for those sites. Specifically, the section of the QHEI assessment that evaluates bank erosion and riparian zone was negatively affected. Yet, the average impact site QHEI score increased from 40.6 in 2013 to 45.6 in 2014. The negative effects of the removed bank vegetation were underscored by the canopy cover data we collected (Table 7). Impact sites had a significant reduction in percent canopy cover in 2014 (paired t-test, $p=0.07$) while control sites did not (paired t-test, $p=0.47$). Clearly, such canopy cover loss is

temporary as bank vegetation will return to the impact sites over time. Moerke *et al.* (2004) also reported a temporary decrease in canopy cover of impact sites immediately following rehabilitation. Bank erosion rates will also likely return to previous levels and may even improve due to the bank stabilization effects of the rehabilitation structures.

Table 6: P-values and coefficients of linear models for QHEI, coarse and fine sands, and coefficient of variation (CV) scores for coarse sands, fine sands and depth. Italicized numbers represent statistically significant p-values.

	p-Value		
	Site	Year	Site:Year
QHEI	0.581	0.127	*0.012*
Coarse Sands	0.983	*0.004*	0.146
Fine Sands	*0.054*	*0.006*	*0.013*
Coarse Sands CV	0.432	0.13	0.223
Fine Sands CV	0.234	0.655	0.304
Depth CV	0.24	0.981	0.703
	Coefficient		
	Site	Year	Site:Year
QHEI	-1.125	-3.25	8.25
Coarse Sands	0.001	-0.186	-0.081
Fine Sands	-0.069	0.166	0.126
Coarse Sands CV	0.053	0.105	-0.118
Fine Sands CV	0.13	-0.048	-0.158
Depth CV	-0.091	0.002	0.041

We predicted that the presence of the structures would have an effect on impact site habitat. One such effect would be a change in friable sediments. In this regard, impact sites would have greater heterogeneity in surficial sediment texture compared to control sites. Within-site and among-site sediment variation can be used to measure this heterogeneity. As such, coefficients of variation (CV) were calculated for coarse and fine sands per site in order to identify changes in within-site sediment heterogeneity (Table 8). However, no significant differences in coarse or fine sand CV scores were revealed. Instead, a shift in the dominant grain size across all sites occurred supported by linear models of the coarse and fine sand sediment data showing statistically significant differences between control and impact sites and between years (Table 6).

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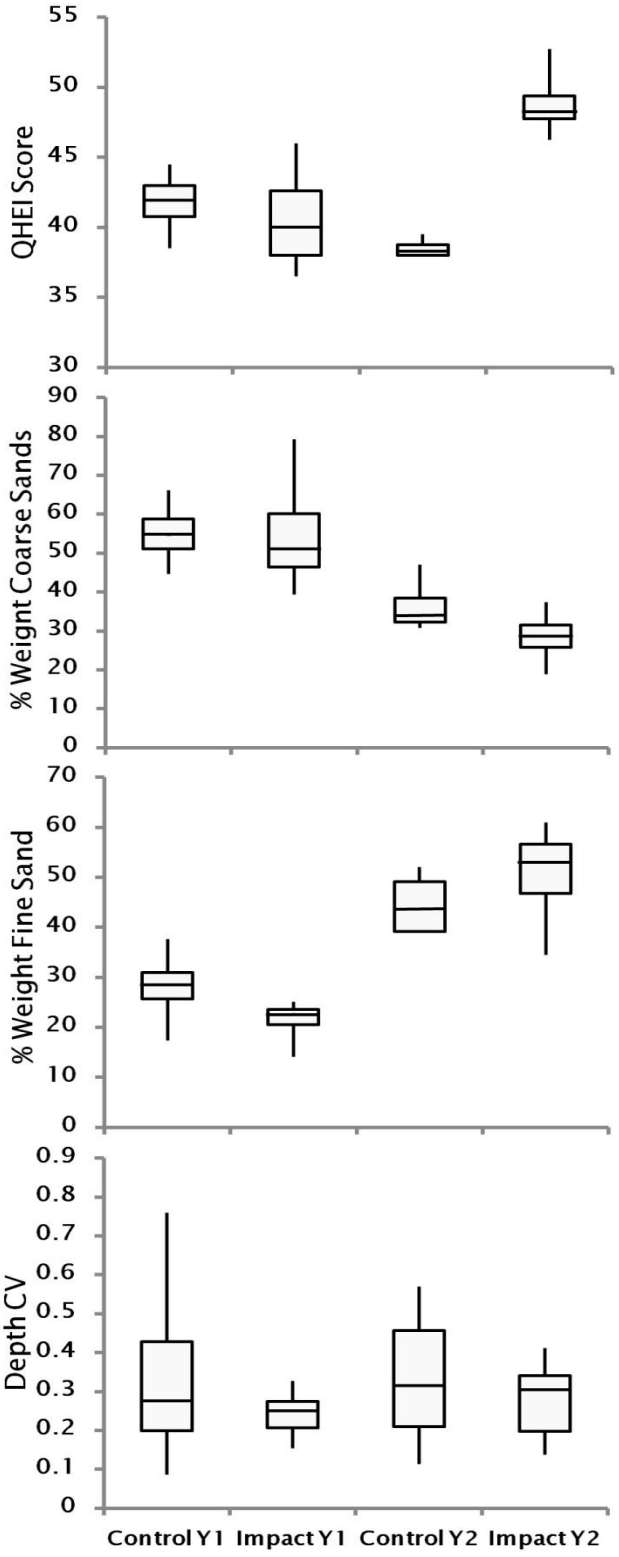

Figure 7: Box and whisker plots of QHEI scores, percent weight coarse and fine sands and depth coefficient of variation (CV) scores for control and impact sites in 2013 and 2014. Boxes represent the first and third quartiles. Horizontal lines within the boxes are median values while whiskers represent minimum and maximum values.

Table 7: Environmental data (expressed as an average with ranges in parentheses) taken from the Ottawa River (Lucas Co.) on the UT Main Campus during 2013 and 2014 sampling.

Site #	2013	Width (m; N=5)		Depth (cm; N=25)		Flow (m/s; N=25)		Canopy Cover (%; N=3)		QHEI	DO (mg/L)	pH	EC (mS)	Temp (C°)
Control 1	June	17.6	(17.2-18.1)	41.8	(15-67)	0.18	(0.02-0.27)				8.4	8.0	0.98	23.0
	July	18.0	(16.0-19.7)	51.6	(23-76)	0.17	(0.01-0.32)	97	(95-99)	42.5	7.7	8.2	0.93	24.1
Control 2	June	16.1	(15.3-17.0)	37.4	(18-52)	0.20	(0.11-0.29)				8.5	8.1	0.93	22.3
	July	17.4	(16.9-18.0)	44.8	(29-67)	0.22	(0.07-0.40)	89	(82-99)	44.5	7.6	8.1	0.90	24.7
Control 3	June	15.8	(12.9-17.6)	60.6	(36-80)	0.11	(-0.02-0.20)				8.7	8.1	0.94	21.4
	July	17.4	(16.1-19.0)	69.4	(46-86)	0.13	(0.03-0.20)	18	(7-40)	38.5	8	8.1	0.91	25.2
Control 4	June	17.1	(16.4-17.7)	77.2	(52-89)	0.07	(-0.03-0.13)				9.4	8.1	0.48	16.6
	July	18.0	(17.0-18.3)	91.8	(78-102)	0.10	(-0.05-0.17)	57	(49-70)	41.5	7.7	8.1	0.94	24.3
Impact 1	June	17.9	(17.0-18.5)	34.4	(20-55)	0.19	(0.02-0.29)				8.4	8.1	0.97	21.7
	July	17.2	(16.3-17.7)	47.8	(36-62)	0.17	(0.04-0.31)	94	(88-98)	46.0	7.6	8.1	0.94	24.4
Impact 2	June	18.0	(16.9-19.0)	35.0	(15-44)	0.18	(0.01-0.29)				8.5	8.1	0.93	22.4
	July	18.7	(18.1-19.4)	49.5	(39-59)	0.19	(0.03-0.32)	43	(34-55)	36.5	7.7	8.1	0.91	25.2
Impact 3	June	17.7	(13.5-19.3)	61.3	(33-81)	0.10	(0.01-0.25)				8.6	8.2	0.92	21.4
	July	17.8	(17.1-18.2)	76.9	(58-89)	0.12	(0.01-0.25)	62	(56-65)	41.5	7.5	8.1	0.97	24.3
Impact 4	June	17.4	(16.4-18.2)	54.0	(27-76)	0.08	(-0.02-0.15)				9.5	8.2	0.96	17.1
	July	17.8	(17.3-18.4)	70.2	(40-84)	0.11	(0.02-0.22)	24	(13-40)	38.5	7.8	8.0	0.92	24.8
Site #	2014	Width (m; N=5)		Depth (cm; N=25)		Flow (m/s; N=25)		Canopy Cover (%; N=3)		QHEI	DO (mg/L)	pH	EC (mS)	Temp (C°)
Control 1	June	16.3	(15.1-17.3)	36.9	(08-61)	0.07	(-0.03-0.15)				9.5	7.4	1.20	24.5
	July	16.1	(15.6-16.8)	28.4	(05-50)	0.05	(-0.01-0.15)	85	(82-87)	38.5	7.6	7.6	1.50	21.5
Control 2	June	15.4	(14.8-16.2)	20.3	(07-44)	0.20	(0.03-0.39)				8.9	7.4	1.20	23.6
	July	14.7	(13.5-15.4)	15.7	(02-39)	0.12	(-0.03-0.31)	99	(96-100)	38.0	7.8	7.6	1.50	21.6
Control 3	June	16.1	(14.6-17.7)	50.8	(24-70)	0.06	(0.02-0.10)				9.3	7.5	1.20	22.8
	July	15.0	(12.1-16.9)	40.6	(19-63)	0.04	(-0.04-0.08)	27	(20-31)	38.0	9.6	7.7	1.40	22.3
Control 4	June	17.6	(17.1-18.4)	68.4	(46-78)	0.05	(0.00-0.12)				10.6	7.5	1.15	24.1
	July	17.1	(16.5-17.6)	60.4	(37-69)	0.01	(-0.02-0.04)	48	(39-58)	39.5	8.2	7.7	1.40	19.8
Impact 1	June	15.3	(14.2-16.3)	41.3	(16-81)	0.07	(0.01-0.18)				9.1	7.5	1.20	24.0
	July	15.8	(15.4-16.5)	40.3	(12-70)	0.04	(-0.01-0.11)	61	(48-73)	49.5	7.2	7.7	1.40	21.5
Impact 2	June	16.2	(15.1-17.5)	13.4	(09-20)	0.09	(-0.01-0.19)				8.8	7.4	1.25	23.4
	July	13.9	(12.4-15.4)	30.8	(06-46)	0.05	(-0.02-0.08)	39	(32-49)	45.0	8.9	7.5	1.30	22.1
Impact 3	June	17.4	(15.8-18.4)	55.2	(27-70)	0.07	(-0.01-0.15)				10.7	7.6	1.18	24.5
	July	16.6	(15.9-17.6)	53.5	(36-64)	0.02	(-0.02-0.05)	47	(37-53)	43.0	8.6	7.6	1.50	19.5
Impact 4	June	16.9	(15.8-17.9)	45.5	(12-62)	0.20	(0.12-0.30)				10.0	7.5	1.12	23.0
	July	16.2	(15.0-17.2)	36.2	(09-50)	0.03	(-0.03-0.07)	22	(09-33)	45.0	8.4	7.7	1.40	20.0

Table 8: Coefficient of variation (CV) measurements for coarse and fine sands from control and impact sites in the Ottawa River (Lucas Co.) in 2013 and 2014 (N=9 per site).

Coarse Sands	2013	2014
Control 1	0.32	0.38
Control 2	0.19	0.40
Control 3	0.39	0.36
Control 4	0.20	0.38
Impacted 1	0.41	0.31
Impacted 2	0.18	0.22
Impacted 3	0.35	0.48
Impacted 4	0.37	0.25
Fine Sands		
Control 1	0.32	0.40
Control 2	0.35	0.28
Control 3	0.64	0.24
Control 4	0.31	0.51
Impacted 1	0.46	0.50
Impacted 2	0.78	0.22
Impacted 3	0.36	0.30
Impacted 4	0.54	0.30

Percent coarse and fine sands were chosen to analyze sediment heterogeneity as they represented the majority of the grain sizes collected during sampling. This is evident in Figure 8, showing a decrease from 2013 to 2014 in both control and impact sites percent weight of coarse sands with Year being a highly significant predictor of percent weight of coarse sands ($p<0.001$). This decrease is not a measurement of heterogeneity, but does provide a better understanding of changes that occurred in surficial sediments texture between 2013 and 2014. Modeling also showed that Year ($p<0.001$) and Site:Year ($p<0.05$) were significant predictors of percent weight of fine sands (Table 6). Year had a positive coefficient, indicating an increase of fine sands in 2014 control sites (see also Figure 7). Very coarse and coarse sands (0.5mm-2mm) was the dominant substrate in the Ottawa River prior to the rehabilitation project (Table 9). In 2014, fine and very fine sands (0.5mm-63um) was the dominant substrate in the Ottawa River on campus. Figure 8 displays this shift in dominant grain size. This shift from coarse to finer sands was likely due to different weather patterns in 2013 and 2014. Larger rain events during the 2013 sampling season and the resulting periods of

increased flow in may have flushed out the fine sands. For example, during June and July 2013, discharge rates in the river exceeded $3m^3 \cdot sec^{-1}$ (~ 100 $ft^3 \cdot sec^{-1}$) during more than 50% of the time. Discharge in June and July 2014 only exceeded $3m^3 \cdot sec^{-1}$ about 5% of the time (Figure 9).

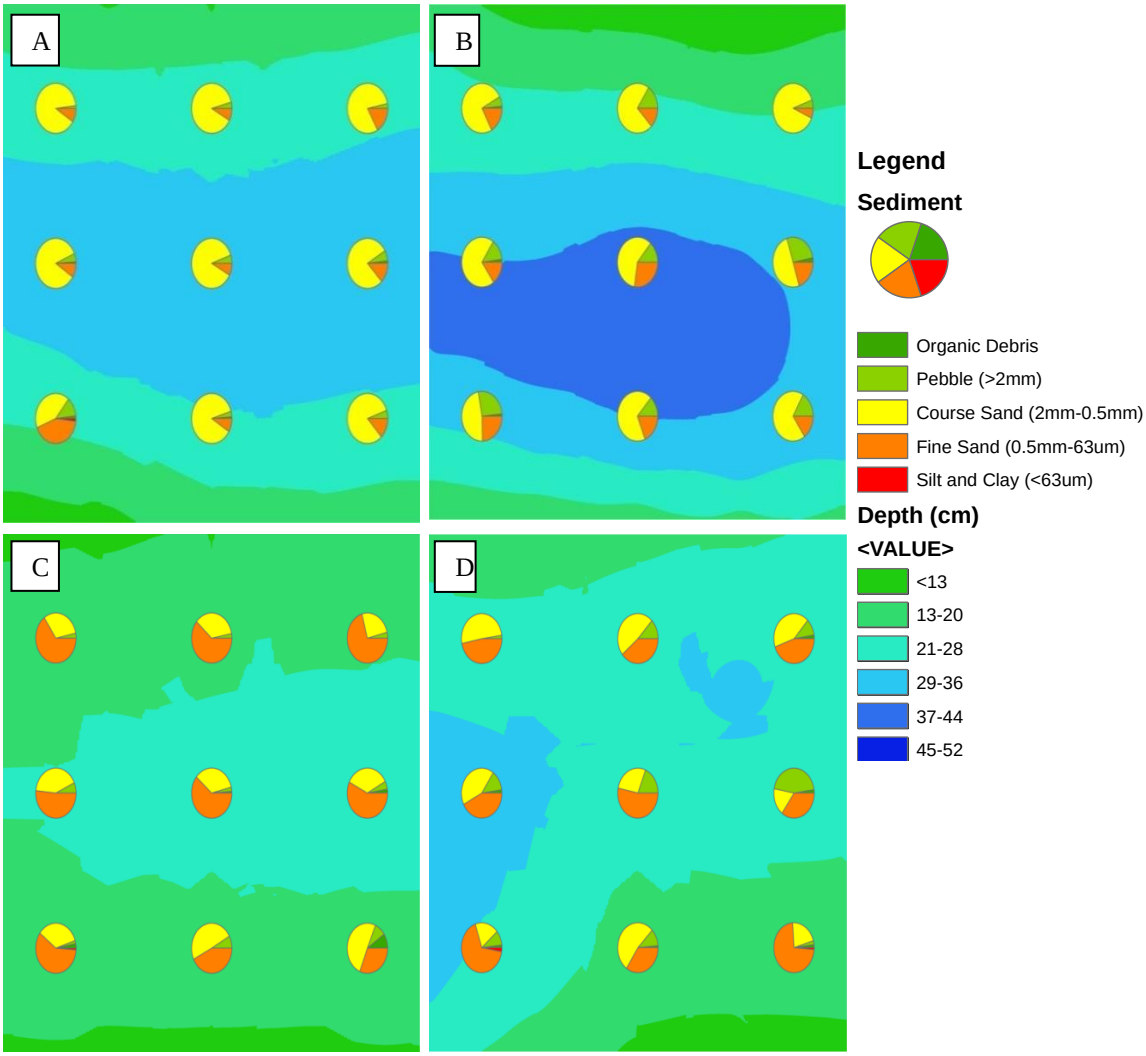

Figure 8: Sediment particle size distribution and coarse organic matter content (as a percent based on total dry weight of sediment core sample) of Impact Site 2 in 2013 (A) and 2014 (C) and Control Site 2 in 2013 (B) and 2014 (D) with depth (cm) interpolations. The top of each map represents the south bank of the river and water flows from right to left. All sites are located in reaches 2 – 5 of the Ottawa River on The University of Toledo campus in Lucas Co. Toledo, Ohio.

Table 9: Sediment texture, composition and coarse organic debris (expressed as percent of total weight with ranges in parentheses) in surface cores from the Ottawa River (Lucas Co.) on the UT Main Campus during 2013 and 2014 sampling.

2013	Control 1	Control 2	Control 3	Control 4	Impact 1	Impact 2	Impact 3	Impact 4
% Organic Debris	1.50% (0.4-7.9)	1.10% (0.1-2.9)	0.50% (0.1-1.4)	1.70% (0.5-3.6)	1.20% (0.3-2.7)	1.00% (0.4-1.7)	1.10% (0.2-3.7)	2.40% (0.1-11.5)
% Pebble	15.10% (7.6-22.3)	15.20% (5.4-27)	13.80% (3.2-52)	15.50% (6.3-31.2)	36.30% (17.8-75.8)	5.30% (1.8-12.4)	24.20% (6.6-45)	20.10% (4-55.9)
% Very Coarse-Coarse Sand	44.70% (23.5-61.6)	66.10% (47.6-88)	56.30% (28.5-87.5)	53.30% (37.7-70.3)	39.30% (10-62)	79.30% (41.9-88.3)	48.80% (32.1-79.1)	53.70% (27.4-85.4)
% Medium-Very Fine Sand	37.60% (17.4-52.8)	17.40% (6.5-27.2)	28.80% (8.7-62.8)	28.40% (18.8-43.4)	22.60% (11.1-37.4)	14.10% (7.7-42.2)	25.10% (13.7-39.3)	23.00% (5.4-47.2)
% Silt and Clay	1.10% (0.2-3.8)	0.10% (0.1-0.3)	0.70% (0.1-2.9)	1.10% (0.2-2.3)	0.60% (0.3-1.1)	0.40% (0.1-1.8)	0.90% (0.3-3)	0.80% (0.1-3.4)
2014	**Control 1**	**Control 2**	**Control 3**	**Control 4**	**Impact 1**	**Impact 2**	**Impact 3**	**Impact 4**
% Organic Debris	1.1% (0.4-3.0)	1.5% (0.4-2.3)	2.5% (0.7-6.8)	2.0% (0.5-4.7)	1.2% (0.6-2.7)	2.1% (0.3-9.5)	3.7% (0.2-23.0)	4.4% (0.1-20.9)
% Pebble	27.7% (7.0-61.5)	13.9% (2.2-44.6)	12.4% (2.2-29.8)	11.4% (1.2-21.0)	35.5% (0.7-61.3)	4.8% (1.9-8.9)	14.2% (1.4-29.9)	14.3% (2.3-45.7)
% Very Coarse-Coarse Sand	30.8% (16.3-55.2)	35.6% (17.7-52.6)	32.7% (20.0-54.4)	47.1% (24.8-77.5)	28.1% (19.6-48.6)	37.4% (25.4-50.5)	18.8% (3.7-34.0)	29.6% (17.4-42.0)
% Fine-Very Fine Sand	39.3% (17.6-57.9)	48.2% (34.3-67.1)	52.0% (30.4-73.0)	38.9% (8.1-63.5)	34.5% (17.1-75.4)	55.2% (31.1-70.1)	61.0% (40.5-92.8)	50.8% (29.4-72.0)
% Silt and Clay	1.0% (0.5-2.6)	0.8% (0.1-2.7)	0.4% (0.0-1.0)	0.7% (0.0-2.9)	0.8% (0.3-1.4)	0.5% (0.1-1.1)	2.3% (0.9-6.4)	0.9% (0.0-2.0)

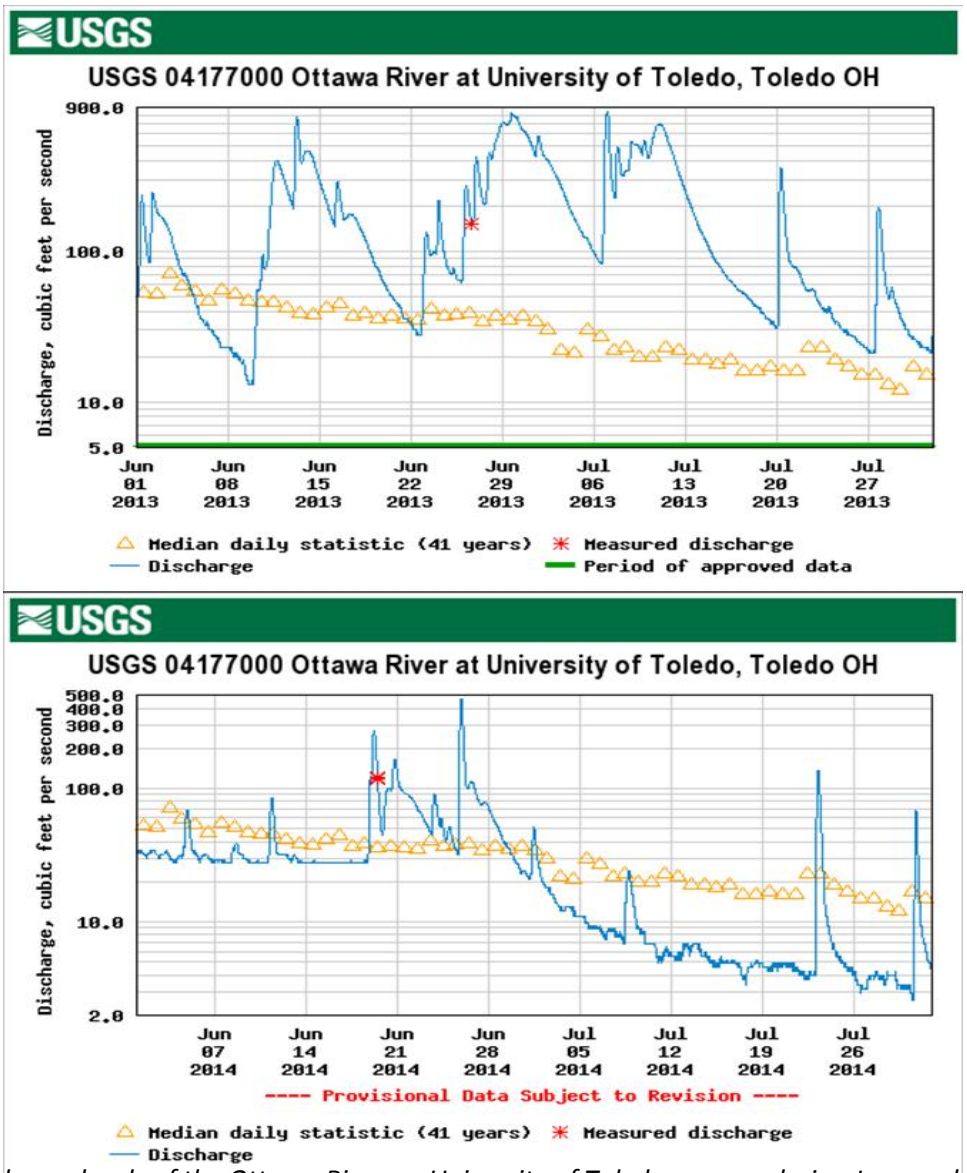

Figure 9: Discharge levels of the Ottawa River on University of Toledo campus during June and July of 2013 and 2014. Discharge expressed in cubic feet per second. Data collected from USGS gaging station 04177000 (USGS 2014).

The significance of the Site:Year variable ($p<0.05$) for fine sands demonstrated a quantifiable impact of the rehabilitation structures on site habitat. The presence of the structures within impact sites may have resulted in an accumulation of fine sands around these structures. Rehabilitation structures such as locked logs may cause accumulation of fine sands (Shields *et al.* 2001) and over the course of one year between sampling events may have produced this difference.

Impact site depth variation did not increase after installation of the structures as shown by the Site:Year variable not identified as a significant predictor in the depth variation model. Depth

variation within each site was calculated using the coefficient of variation of 25 water depth measurements per site (Table 10), which were evaluated for both years and then analyzed for significance. For this analysis, we standardized the water depth measurement between sampling years using gage height data from the USGS gaging station in the river on campus.

Table 10: Average coefficient of variation measurements for 25 water depth measurements per site from control and impact sites in the Ottawa River (Lucas Co.) in 2013 and 2014.

	2013	2014
Control 1	0.45	0.40
Control 2	0.53	0.56
Control 3	0.24	0.25
Control 4	0.10	0.13
Impacted 1	0.25	0.41
Impacted 2	0.19	0.26
Impacted 3	0.23	0.16
Impacted 4	0.30	0.31

River width, canopy cover, dissolved oxygen (DO), pH, electrical conductivity (EC), and temperature were also recorded (Table 7). Overall, these data represented typical conditions of a Midwestern urban river (Gannon and Busse 1989, Tong and Chen 2002, Nedeau *et al*. 2002). Very little variance in river widths was observed, demonstrating the confined character of the homogenous channel of the river on campus. DO and temperature measurements showed no obvious differences between control and impact sites or between years. pH values in 2013 were somewhat higher than those in 2014, perhaps due to the higher rates of rainfall relative to 2014 adding carbonates from the drainage basin to the river. Despite the observed difference in pH between years, the ranges of pH documented were not outside tolerance levels of freshwater fish species (Ohio EPA 2008b). EC significantly increased in 2014 across control (paired t-test, $p=0.0001$) and impact (paired t-test, $p=0.0001$) sites, perhaps due to the lower water levels that year. With June and July 2014 receiving 15.8cm less rainfall than in 2013, the concentration of inorganic dissolved solids in the water may have been elevated relative to the previous year. The EC measurement for Control 4 in June 2013 was noticeably lower than the other measurements for 2013 (Table 7). This likely represents a measurement or probe error.

Conclusions

A significant improvement in impact site QHEI scores was observed in 2014. Coupled with a minimal fish community response, this suggests a delay between improved habitat and a fish community response. We expected an increase in depth variations within impact sites as well as a more heterogeneous distribution of surface sediment particle size classes, such as patches of fine sands interspersed with areas of coarser pebbles. We found neither. Gore and Shields (1995) noted that rehabilitation structures accelerated and redirected flow within a site, flushing away smaller sized sediments and creating areas comprised of larger particles. Large woody debris structures have also been found to promote sediment heterogeneity by increasing riverbed roughness (Moerke and Lamberti 2004). Instead, we observed a shift from coarse sands to fine sands throughout control and impact sites likely due to differences in rainfall and flow between years.

The impact sites, and thus the rehabilitation structures, were positively correlated with fish abundance. Significant responses to the structures were not observed for richness, IBI, and SDI. Our findings were constrained by the two-year study timeframe, one year pre and one year post-installation. Our data provide an important baseline for continued monitoring of future fish community responses to river rehabilitation. In a meta-analysis of six rehabilitation projects spanning seven years, Schmutz *et al.* (2013) reported a 55% increase in species richness across 19 rehabilitated sites of the Danube River relative to control sites. Similar results were found in a three year post-rehabilitation monitoring study that included 15 sites from seven large rivers (Jungwirth *et al.* 1995). Species richness and abundances significantly increased at each of these 15 sites after three years. Thus, the temporal scale of a rehabilitation effort plays a large role in determining the tempo and quality of a biological response. Conversely, no positive response from the fish community of the Ottawa River may be found by continued monitoring. In fact, Moerke *et al* (2004) monitored two rehabilitated sites and one control site of a 3rd order Indiana river over five years and found fish abundances of the rehabilitated sites to be similar to or even below abundances of the control site throughout the five year period.

The spatial scale of rehabilitation is also of considerable importance. River rehabilitation is more than the installation of in-stream structures throughout a specific small stretch of river. The focus of project managers needs to include floodplain rehabilitation, storm water management, non-native species control and hydrological alterations (Gottgens *et al.* 2009, Booth 2005). In the case of the Ottawa River, a continued focus on the implementation of woody debris is important as it was likely abundant historically throughout the Great Black Swamp area (Sampson 1930). Future rehabilitation efforts should continue to encompass these important aspects along with the structural improvements of a river system.

Outreach and Broader Impacts

This project provided practical field experience for more than 300 University of Toledo students in the following courses during 2014. This experience is continuing in subsequent semesters.

- EEES 1020 (Introductory Geology lab): Thirteen sections, 150 students, 1 laboratory exercise on stream discharge.
- EEES 1050 (Geological Hazards and the Environment): Two sections, 82 students, 1 class visit, discharge measurements.
- EEES 1140 (Environmental Problems lab): Eighteen sections, 2 visits per semester, 300+ students, seining around structures.
- EEES 3060 (General Ecology lab): Two sections, 30 students, 1 visit, seining for fish community composition.
- EEES 4740 (Aquatic Ecology lab): One section, two lab exercises covering QHEI and IBI scoring, water quality, and stage discharge curves.

This grant formed the basis of the Master of Science thesis research of a University of Toledo graduate student (Aaron Svoboda). In addition, this grant provided an opportunity for an honors thesis project by James Moriarty who compared the effectiveness of the two different fish sampling techniques we used. This honors project was sponsored under a separate student grant from the University of Toledo's Office of Undergraduate Research and resulting in a poster presentation at a regional symposium, an oral presentation at the 2014 National Council of Undergraduate Research annual meeting (Lexington, KY) and a honors thesis presentation.

Posters/presentations/reports resulting from this project:

- Svoboda, A.D., J.F. Gottgens and B.J. Zimmerman. Shrinking Distribution of the Least Darter (*Etheostoma microperca* Jordan and Gilbert): A State Listed Species of Concern in Ohio. *American Midland Naturalist (in revision)*
- Gottgens, J.F., T.D. Crail, A. Svoboda and J. Moriarty. 2014. Fish community composition pre- and post-installation of in-stream structures to assess improvement to ecosystem function: Phase I (pre-installation). Final Report to the U.S. Fish and Wildlife Service: Award: F13AP00516: 14pp.
- Svoboda, A., J.F. Gottgens and T.D. Crail. 2014. Effects of Urban River Rehabilitation Structures on the Fish Community in the Ottawa River, Ohio. EcoStream Conference, NC State, Charlotte North Carolina.
- Moriarty, J., J.F. Gottgens and T.D. Crail. 2013. Benefits and downfalls of two different fish capturing methods used to evaluate fish communities of the Ottawa River, Lucas County, Ohio. Final report, University of Toledo Office of Undergraduate Research: 18 pp.
- Svoboda, A., J. Moriarty, J.F. Gottgens and T.D. Crail. 2013. The Fishes of the Ottawa River, Ohio. Presented to the UT campus community during the Clean your Streams week, Toledo, OH.

- Moriarty, J., J.F. Gottgens and T.D. Crail. 2013. Comparing two different fish capturing techniques in the Ottawa River, Ohio. University of Toledo Undergraduate Research Symposium, Toledo, Ohio. (poster)
- Moriarty, J., J.F. Gottgens and T.D. Crail. 2013. Comparing two different fish sampling methods to evaluate fish communities of the Ottawa River, Lucas County, Ohio. 27th Annual National Conference on Undergraduate Research, Lexington, Kentucky. (oral presentation)
- Research updates provided monthly to the University of Toledo's President's Commission on the River.

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Appendix A: Scientific and common names of fish species recorded from the Ottawa River (Lucas. Co.) during the 2013 and 2014 summer sampling.

Scientific Name	Common Name
Catostomidae	
Carpoides cyprinus Lesueur	Quillback carpsucker
Catostomus commersonii Lacepede	White sucker
Minytrema melanops Rafinesque	Spotted sucker
Moxostoma erythrurum Rafinesque	Golden redhorse
Centrarchidae	
Micropterus salmoides Lacepede	Largemouth bass
Lepomis cyanellus Rafinesque	Green sunfish
Lepomis gibbosus Linnaeus	Pumpkinseed sunfish
Lepomis humilis Girard	Orangespotted sunfish
Lepomis macrochirus Rafinesque	Bluegill sunfish
L. cyanellus x L. macrochirus	Green sunfish/Bluegill hybrid
L. gibbosus x L. humilis	Pumpkinseed/Orangespotted sunfish hybrid
Pomoxis annularis Rafinesque	White crappie
Pomoxis nigromaculatus Lesueur	Black crappie
Clupeidae	
Dorosoma cepedianum Rafinesque	Gizzard shad
Cyprinidae	
Campostoma anomalum Rafinesque	Central Stoneroller
C. auratus x C. carpio	Goldfish/Common carp hybrid
Cyprinus carpio Linnaeus	Common carp
Cyprinella spiloptera Cope	Spotfin shiner
Luxilus cornutus Mitchill	Common shiner
Lythrurus umbratilis Girard	Redfin shiner
Notropis atherinoides Rafinesque	Emerald shiner
Notropis stramineus Cope	Sand shiner
Phenacobius mirabilis Cope	Suckermouth minnow
Pimephales notatus Rafinesque	Bluntnose minnow
Pimephales promelas Rafinesque	Fathead minnow
Semotilus atromaculatus Rafinesque	Creek chub
Esocidae	
Esox lucius Linnaeus	Northern pike
Gobiidae	
Neogobius melanostomus Pallas	Round goby
Ictaluridae	
Ameiurus nebulosus Lesueur	Brown bullhead
Noturus gyrinus Mitchill	Tadpole madtom
Percidae	
Perca flavescens Mitchill	Yellow perch
Etheostoma nigrum Rafinesque	Johnny darter
Etheostoma microperca Jordan and Gilbert	Least darter
Percina caprodes Rafinesque	Logperch darter
Percina maculata Girard	Blackside darter
Sciaenidae	
Aplodintus grunniens Rafinesque	Freshwater drum

Appendix B: Regional species pool for the Ottawa River (Lucas Co.). Sites are the Maumee River, Swan Creek, and the Lake Erie western basin. Sites were chosen based on their proximity to the Ottawa River and interconnectedness and represent documented species from collections spanning 1997 to 2012 (Grabarkiewicz and Crail 2009, Ohio EPA 2008a).

Ottawa River	Maumee River	Swan Creek	Lake Erie (Western Basin)
Black redhorse	Bigmouth buffalo	Blackside darter	Bigmouth buffalo
Blackside darter	Black buffalo	Bluegill sunfish	Black bullhead
Bluegill sunfish	Black bullhead	Bluntnose minnow	Black crappie
Bluntnose minnow	Black crappie	Brindled madtom	Black redhorse
Brook silverside	Blackside darter	Brook silverside	Bluegill sunfish
Central mudminnow	Blackstripe topminnow	Brown bullhead	Bluntnose minnow
Central stoneroller	Bluegill/Orangespotted hybrid	Central mudminnow	Bowfin
Channel catfish	Bluegill sunfish	Central stoneroller	Brook silverside
Common carp	Bluntnose minnow	Channel catfish	Brown bullhead
Carp/goldfish hybrid	Brook silverside	Common carp	Brown trout
Common shiner	Brown bullhead	Creek chub	Channel catfish
Creek chub	Central mudminnow	Emerald shiner	Common carp
Emerald shiner	Central stoneroller	Fathead minnow	Emerald shiner
Fathead minnow	Channel catfish	Freshwater drum	Flathead catfish
Freshwater drum	Common carp	Gizzard shad	Freshwater drum
Gizzard shad	Carp/Goldfish hybrid	Golden redhorse	Ghost shiner
Golden redhorse	Common shiner	Golden shiner	Gizzard shad
Golden shiner	Creek chub	Goldfish	Golden redhorse
Goldfish	Dusky darter	Grass pickerel	Golden shiner
Grass pickerel	Emerald shiner	Greater redhorse	Goldfish
Green sunfish	Fathead minnow	Green sunfish	Greater redhorse
Johnny darter	Flathead catfish	Greenside darter	Green sunfish
Largemouth bass	Freshwater drum	Johnny darter	Largemouth bass
Least darter	Ghost shiner	Largemouth bass	Logperch darter
Logperch darter	Gizzard shad	Logperch darter	Longnose gar
Longnose gar	Golden redhorse	Northern hogsucker	Mimic shiner
Mosquitofish	Golden shiner	Northern pike	Northern hogsucker
Northern pike	Goldfish	Orangethroat darter	Northern pike
Orangespotted sunfish	Greater redhorse	Pumpkinseed sunfish	Orangespotted sunfish
Orangethroat darter	Green sunfish hybrid	Rainbow trout	Pumpkinseed sunfish
Pumpkinseed sunfish	Green/bluegill hybrid	Redfin shiner	Quillback carpsucker
Quillback carpsucker	Green/Orangespotted hybrid	Rock bass	Rainbow trout
Rainbow trout	Green/Pumpkinseed hybrid	Round goby	Rock bass
Redfin shiner	Green sunfish	Shorthead redhorse	Round goby
Round goby	Greenside darter	Silver lamprey	Sand shiner
Sand shiner	Johnny darter	Silver redhorse	Shorthead redhorse

Shorthead redhorse	Largemouth bass	Silver shiner	Silver chub
Spotfin shiner	Logperch darter	Silverjaw minnow	Silver redhorse
Spottail shiner	Longnose gar	Smallmouth bass	Smallmouth bass
Spotted sucker	Mospuitofish	Smallmouth buffalo	Smallmouth buffalo
Striped shiner	Northern hogsucker	Spotfin shiner	Spotfin shiner
Suckermouth minnow	Northern pike	Spottail shiner	Spottail shiner
Tadpole madtom	Orangespotted/Pumpkin seed hybrid	Spotted sucker	Spotted sucker
White sucker	Orangespotted sunfish	Stonecat madtom	Trout-perch
Yellow bullhead	Orangethroat darter	Striped shiner	Walleye
Yellow perch	Pumpkinseed sunfish	Tadpole madtom	White bass
	Quillback carpsucker	White bass	White crappie
	Rainbow trout	White perch	White perch
	Redear sunfish	White sucker	White sucker
	Redfin pickerel	Yellow bullhead	Yellow bullhead
	Redfin shiner	Yellow perch	Yellow perch
	Rock bass		
	Round goby		
	Sand shiner		
	Shorthead redhorse		
	Silver chub		
	Silver lamprey		
	Silver redhorse		
	Silver shiner		
	Silverjaw minnow		
	Smallmouth bass		
	Smallmouth buffalo		
	Spotfin shiner		
	Spottail shiner		
	Spotted sucker		
	Stonecat madtom		
	Striped shiner		
	Striped/Common shiner hybrid		
	Suckermouth minnow		
	Tadpole madtom		
	Trout-perch		
	Walleye		
	Warmouth sunfish		
	White bass		
	White crappie		
	White perch		
	White sucker		
	Yellow bullhead		
	Yellow perch		