Benthic Macroinvertebrate Pre-assessment of a Proposed Restoration in the Grand River, Grand Rapids, MI, USA

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A Thesis Submitted to the Graduate Faculty of

GRAND VALLEY STATE UNIVERSITY

In

Partial Fulfillment of the Requirements

For the Degree of

Master of Science

Biology

December 2022

Thesis Approval Form

GRAND VALLEY STATE UNIVERSITY

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Acknowledgements

First and foremost, I would like to thank my friend and advisor, Dr. Eric Snyder, for all of his help, insight, and encouragement throughout the entire process of my graduate work. I am especially grateful for his assistance with, and full indulgence of, my taxonomic endeavors, including many hours spent working with specific keys and verifying diagnoses; encouraging and helping to fund taxonomic certification; and supporting, through time and funding, training in Chironomidae mounting/identification methodology. On the taxonomic front, a big thank you to Dr. James Dunn, a friend, mentor, and committee member, who taught me "bug" I.D., in general. I am also thankful for all of Dr. Dunn's advice and guidance leading up to grad school, which helped set me on this path initially. Thank you to my other committee members as well, Dr. Timothy Evans and Dr. Stephanie Ogren. Dr. Evans always took time out of his hectic schedule to work with me and I am especially grateful for his mentoring/guidance in my first teaching experience as a graduate student. Dr. Ogren has been critical in locating various local source material for my thesis that I was generally ignorant of and, in that way, played a major role in helping to round out my thesis arguments. I also want to thank my many excellent graduate professors - Dr. Meg Woller-Skar, Dr. Daniel Frobish, Dr. Jennifer Moore, Dr. Amy Russell, Dr. Mark Luttenton - and department faculty - Diane Laughlin and Barb Ellis - for all of your contributions to my work and education.

Many thanks to all student help, cohort or otherwise, that volunteered their time to work with me, including, but not limited to: Colin Assenmacher, Alexis Neff, Ben Gunnett, Taylor Suttorp, Paul Dingman, and Abigail Fischer. Thank you to Aaron Parker, of EGLE, for fielding numerous questions, over the years, and supplying requested data.

Thank you to my partner Mindy and daughter Emma for love, support, and understanding throughout the process. I am grateful to Mindy, especially, for providing positivity, humor, and balance behind the scenes. Thank you, also, to my mother Linda, father Darwin, and stepmother Chelle, all of whom have been consistently supportive and encouraging throughout my grad school career.

Finally, thank you to my first Biology mentor and "Bug" teacher, Dr. Matt Douglas. If you hadn't inspired me, initially, in Biology, I'm not sure where I'd be.

Abstract

River restoration is a dominant field of applied water resources management in the United States. Ecological improvement should be the goal of all river restorations, though many restoration projects fail to produce positive results due to limited scope or inadequate assessment methodologies. Pre-restoration, biotic data is essential for such projects as it can be paired with post-restoration data to gauge ecological outcomes. A major restoration effort is now underway in Michigan's longest river, the Grand River, where it flows through downtown Grand Rapids. The primary restoration measures will occur in-stream at local scale. We conducted a pre-restoration survey for the downtown reach of the Grand River with concomitant survey of a control reach using benthic macroinvertebrates. We predicted that the control reach would have higher ecological integrity than the downtown reach based on prior assessments. We found the downtown reach to have a dominant percentage of dipterans (39% vs. 3% in the control), and a high percentage of Hydropsychinae (83% of the total Trichoptera vs. 23% in the control). A dominance of dipterans and/or Hydropsychinae often points to impairment. We predict that the Grand River restoration is unlikely to improve macroinvertebrate community metrics without mitigating general out-of-channel/upstream sources of impairment.

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Abbreviations

BACIP – Before-After-Control-Impact-Pairs

BMPs – Best Management Practices

C-Site – Control Site

EGLE – Environment, Great Lakes, and Energy

EPT – Ephemeroptera, Plecoptera, Trichoptera

EtOH – Ethanol

FFG – Functional Feeding Group

GRWW – Grand Rapids Whitewater

Im-Site – Impact Site

KRRP - Kissimmee River Restoration Evaluation Program

LGROW – Lower Grand River Organization of Watersheds

LRPA – Lower Reach Project Area

LWD – Large Woody Debris

NMDS – Non-metric Multidimensional Scaling

USGS – United States Geological Survey

Chapter 1 – Thesis Introduction

Introduction / Background

Rivers and streams (flowing waters) are rich systems of biotic and abiotic interchange that make life on Earth, as we have come to know it, possible. Not only do they provide fresh water for all biotic processes, they also help maintain necessary biogeochemical cycles (Hauer & Lamberti 2017). Throughout history, humankind has sought proximity to flowing waters for various services such as nourishment (macroinvertebrates and fish), waste management, energy (hydropower), transportation, and, to a lesser extent, recreation and aesthetics (Haidvogl 2018). Unfortunately, our heavy usage of these services in tandem with exponential population growth and general ignorance of flowing systems, has led to various levels - absolute in some cases - of degradation (Wohl 2004; Haidvogl 2018). We have overfished flowing waters, causing extinction or mass reduction of species; impaired system processes by agricultural land-use alterations (e.g. riparian removal and wetland destruction); fouled flowing waters with human waste, livestock waste and myriad other point and nonpoint source pollutants (e.g. nutrients, sediment, litter); disconnected systems by damming longitudinal flows, paving floodplains, dredging alluvial beds; and altered biological communities by introducing invasive species via canals and shipping (Wohl 2004; Malmqvist & Rundle 2002). In the United States, the degradation of freshwaters reached a historic apex in the 1960s when two-thirds of our flowing waters were found to be polluted (Palmer & Allan 2006). This historic low, along with public outcry, culminated in the passage of the Clean Water Act of 1972 (Palmer & Allan 2006), which made discharging any point source pollutant into “navigable waters” illegal without a permit, prohibited discharge of toxic pollutants, and set standards for water quality/ecological integrity (Foster & Matlock 2001). As a necessary corollary, perhaps, river restoration became a dominant trend in natural resources management within the following decade (Wohl et al. 2015).

River restoration is a branch of water-resources management and currently supports a multibillion-dollar industry worldwide (Wohl et al. 2015). In the United States, alone, more than a billion dollars is spent annually on river and stream restoration (Palmer & Allan 2006). River restoration may be divided into two general types: ecological restoration, where some facet/s of the ecosystem are intended for improvement; or, restoration for other purposes such as recreational activities (Wohl et al. 2015). However, all river restoration should attempt ecological improvement as this inheres in the very term "restoration" (Palmer et al. 2005). River restoration projects may seek to modify single or multiple elements of a river or stream in-channel (e.g. substrate, flow regime) or out (banks, riparian zone, floodplains) (Wohl et al. 2015); the scope of restorations may vary as well from reach to catchment scale (Wohl et al. 2015). Typically, though, the goal of most restorations is to modify in-stream substrate at the reach scale (Palmer et al. 2014). In recent years, this reach-scale, in-channel focus has come under scrutiny from experts in the field (Bernhardt & Palmer 2011; Lorenz & Feld 2013). Multiple studies of mass data sets, both American and European, have shown little to no ecological improvement for river/stream restorations of this sort (Palmer et al. 2010; Feld et al. 2011; Haase et al. 2013). What accounts for this failure? The reason may be as simple as catchment degradations overriding reach restorations (Bernhardt & Palmer 2011; Lorenz & Feld 2013). In other words, unrestored upstream reaches and/or unrestored adjacent riparian zones/floodplains of the same watershed are impacting a restored reach. Conversely, the restoration measures may have caused further disturbance on their own (Tullos et al. 2009). Another consideration is the timescale of recovery, which is highly variable between different restoration methods, and whether a "failure" or non-significant result may be due to an inadequate timescale for post-restoration assessment/s (Felt et al. 2011; Bernhardt & Palmer 2011).   Further, the failure to ecologically improve a reach could be a combination of both scenarios. So how might river restoration tend toward ecological improvement or at least set ecological improvement as a reasonable hypothesis?

In 2005, top researchers in the field of water-resources management set forward five keys to ecological success in river restoration (Palmer et al. 2005): 1) have a guiding image at the outset of the project; 2) improve the ecosystem in a measurable way; 3) enable the ecosystem to be more self-sustaining; 4) do no harm during the restoration; 5) complete a pre-/post-assessment (Palmer et al 2005). The completion of a pre-/post-assessment is perhaps the most important key as it underlies the other four and allows for ecological impact of the restoration to be gauged. Completion of a pre-/post-assessment is also important for another reason: to inform future projects. As of 2005, 90% of restoration efforts had not conducted assessments and, therefore, were unable to quantify the results of their "restoration" whether ecological improvement was made or not (Bernhardt et al. 2005). This is unfortunate because even if a restoration is not ecologically successful, the analysis of a pre-/post-assessment may elucidate the reasons for failure or statistical insignificance and, thus, provide invaluable information for future restorations (Palmer et al. 2005).

Assessments of river restorations may focus on biotic groups (macroinvertebrates, fish, plants) and/or abiotic factors (habitat, water chemistry, hydromorphology) (See Morandi et al. 2014, Table 2). Biotic groups, especially macroinvertebrate communities, respond in predictable ways to various disturbances and thus become convenient proxies for ecosystems (Merritt et al. 2019). Biotic data is calculated as metrics of diversity, abundance, functionality, and percent sensitive taxa (e.g. percent EPT, percent Diptera), which measure ecological integrity of a site, location, etc. (Merritt et al. 2019). Ecological integrity of a restored/impact site is then compared, at least, to a single control site (usually a distant, upstream location); or compared to the same control *before* and *after* the restoration as part of a Before-After-Control-Impact-Pairs design (BACIP) (Palmer et al. 2005). The BACIP is ideal for ecological field studies as it does not rely on replicates of the perturbation (i.e. re-degrading, re-polluting a river) for statistical analysis but, rather, on replicates in time and space (Stewart-Oaten et al. 1986 and 1992; and modified by Underwood 1992). Replicates in time refers to the Before/After component, and replicates in space to the Control/Impact. Simply put, a BACIP is employed to weed out the natural variation seen over time. For example, a pristine control site will, over a period of years, be affected by climate change in a similar way as the impact site. By conducting a Before/After, Control/Impact assessment, the data can be paired, and the variance between sites analyzed to elucidate the actual results of the restoration.

Currently a major restoration effort is underway in Michigan's (MI, USA) longest river, the Grand River, where it flows through downtown Grand Rapids. The Grand River begins in Hillsdale County (southern MI) and flows in a northwesterly direction through the state capital (Lansing) and, eventually, Grand Rapids, toward its confluence with Lake Michigan in Grand Haven. Along the river's 252-mile (406 kilometer) course, forests and wetlands have been historically reduced by a combined 75%, and the river is impacted by agricultural lands (57% of the total watershed), residential areas, and recurring urbanization (9% total) (Hanshue & Harrington 2017). In downtown Grand Rapids, the Grand River is typical of other rivers that run through major urban centers: it is a channelized, dredged, and dammed river with nearly 100% impervious riparian. This uniform, high-gradient river channel once contained a series of rapids with a purported fall of 18 feet from which the city took its name (Baxter 1891). The rapids were demolished in the 1860s as part of a navigation project which was never completed (Hanshue & Harrington 2017). Grand Rapids Whitewater (GRWW), a not-for-profit organization spearheading the restoration, plans to restore the rapids by removing dams (4 low-head dams and 1 large weir dam) to achieve natural hydrology; adding an adjustable hydraulic structure to block upstream migration of invasive sea lamprey; and augmenting existing substrate with boulders and other large rocks to form riffle sequences (grandrapidswhitewater.org). In total, the restoration will encompass a 4 km stretch of the Grand River. The restoration will occur in two phases, the first to begin as early as summer 2023 (Burkman & Ogilvie 2022). Phase 1 involves demolition of four low-head dams and creation of in-stream hydrogeomorphologic features across an 840 meter stretch designated the Lower Reach Project Area (LRPA) (grandrapidswhitewater.org). GRWW's stated goals for the project are 1) to rebuild the rapids for recreational purposes and, 2) ecologically improve the Grand River's downtown stretch (grandrapidswhitewater.org, See Mission Statement).

Purpose

Rivers and streams provide essential ecosystem services to nearly all forms of life. And, to some extent, their functionality ensures that global, biogeochemical processes continue unabated. Up until the early 2000's, river/stream health had been trending upward; but a 2009 EPA report, based on ~564,000 assessed miles of flowing waters (16% of U.S. rivers and streams), showed a decline in river/stream health (Palmer and Allan 2006; EPA 2009). As of 2022, 51% of assessed U.S.A. river/stream miles were found to be impaired (Environmental Integrity Project 2022). Ecological river restoration, as those outlined by Palmer et al. (2005), is a best-practice methodology which can address and mitigate the impairment of our flowing water systems.

The downtown section of the Grand River has suffered historic degradation, namely, hydrogeomorphologic alterations (channelization, dredging, damming) and point source pollution (sewage, factory effluents) (Hanshue & Harrington 2017). Thanks, in part, to the Clean Water Act's regulation of point-source pollution and standardization of ecological integrity, the Grand River has improved, measurably, since its nadir. Grand Rapids Whitewater is taking the logical next step of restoring the original hydrogeomorphology, i.e. rebuilding the rapids. In-stream work may begin as early as July 2023 (= Phase 1) (Burkman & Ogilvie 2022). In order to measure the ecological impact of the Grand River restoration, a pre- and post-assessment must be completed.

To our knowledge, pre-assessments of the LRPA of the Grand River restoration section have been limited to fish (Neff 2022; Arnold 2021) and mussels (See https://www.nrcs.usda.gov/conservation-basics/conservation-by-state/michigan/news/public-comment-period%20LGR%20EA, Part 2). It is essential that benthic macroinvertebrates are also assessed because 1) unlike fish, they are poor dispersers with short life cycles and, thus, integrate, as a community, the effect of disturbances over time (Merritt et al. 2019); and 2) unlike mussels, benthic macroinvertebrate life stages, across orders and families, are highly variable in sensitivity and, as a group, benthic macroinvertebrates occupy nearly every trophic level (Merritt et al. 2019). Our project focuses on benthic macroinvertebrates and seeks to fill a gap in pre-assessment data.

Scope

This study provides a pre-assessment of the LRPA of the Grand River restoration section using aquatic macroinvertebrates and an upstream control. Our pre-assessment data is intended to be paired with future post-assessment data using the same methodology (See Chapter 2 Methods). The pre-assessment of both the LRPA and control were conducted in late July and should only be compared with an approximate seasonal post-assessment as taxa are likely to change in abundance and diversity over a given annual cycle (Merritt et al. 2019). As a taxonomic dataset, our inventory is limited to organisms occurring on benthic substrates. We chose to focus our sampling effort on benthic substrates as these will be directly altered by the restoration.

Assumptions

Given the consistent substrate type and taxonomic community composition across our five downtown sampling sites, we assume that our inventory is generally representative of the overall community composition in the LRPA (= Lower Reach Project Area as defined by grandrapidswhitewater.org) of the downtown restoration section. That being said, our selection of sites was stratified/ constrained by depth (our sampling depth ranged between ~ 40 and 60 centimeters), based on our sampling device, and are not truly random. This should not be an issue for pairing with a post-assessment as long as a similar sampling device and methodology are used (See Chapter 2 Methods).

We also assume that our subsampling methodology, as suggested by Dr. F. Richard Hauer (personal communication, 2021) and similar to that recommended by Barbour et al. (1999), yielded representative totals for each sample.

Finally, we assume that no major pollution event occurred at either of our two locations during or relatively previous to the sampling period that would affect our data. We are not aware of any such event for either of the two locations and have no reason to suspect that any occurred.

Objectives

The primary objective of this study was to complete a pre-assessment of the LRPA of the Grand River restoration section using benthic macroinvertebrates. This data provides a viable baseline of current benthic macroinvertebrate abundance, diversity, and functionality in the downtown Grand River that can be compared to a similarly derived post-restoration dataset. By having both pre- and post-assessment data, the ecological impact of the Grand River restoration can be gauged.

The secondary objective of this study was to characterize the downtown Grand River section, ecologically, by presenting a benthic taxonomic inventory along with selected abiotic metrics, substrate composition overview, and general habitat features.

We contend that both objectives have been met.

Significance

Flowing waters are imperiled across the United States owing to various anthropogenic impacts. River restoration describes a wide scope of methods which can potentially improve the ecologies of rivers and streams if certain criteria are followed (specifically, “the 5 keys” of Palmer et al. 2005). In order to determine the ecological outcomes of river restoration, pre-/post-assessments must be performed and analyzed. A major restoration effort is set to begin, by summer 2023, in Michigan's longest river, the Grand River, where it flows through Michigan's second largest city, Grand Rapids (census.gov). Our study performed a pre-assessment of the Grand River restoration section using benthic macroinvertebrates as the focal taxa. Ideally, our methodology will be repeated after the restoration (= post-assessment), and both datasets paired for analysis.

Chapter 2 - Benthic macroinvertebrate pre-assessment of a proposed restoration in the Grand River, Grand Rapids, MI, USA

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Abstract

River restoration is the dominant field of applied water resources management in the United States. Ecological improvement should be the goal of all river restorations, though many restoration projects fail to produce positive results due to limited scope or inadequate assessment methodologies. Pre-restoration, biotic data is essential for such projects as it can be paired with post-restoration data to gauge ecological outcomes. A major restoration effort is now underway in Michigan's longest river, the Grand River, where it flows through downtown Grand Rapids. The primary restoration measures will occur in-stream at local scale. We conducted a pre-restoration survey for the downtown reach of the Grand River with concomitant survey of a control reach using benthic macroinvertebrates. We predicted that the control reach would have higher ecological integrity than the downtown reach based on prior assessments. We found the downtown reach to have a high percentage of dipterans (39% vs. 3% in the control), and a high percentage of Hydropsychinae (83% of the total Trichoptera vs. 23% in the control). A dominance of dipterans and/or Hydropsychinae often points to impairment. We predict that the Grand River restoration is unlikely to improve macroinvertebrate community metrics without addressing general out-of-channel/upstream sources of impairment.

Keywords: Benthic macroinvertebrates - Bioassessment - River restoration - Quantitative sampling - Before after control impact pairs - Percent Diptera

Introduction

The restoration of flowing waters has been a dominant trend in United States (U.S.) natural resources management since the 1980s, largely driven by the Clean Water Act of 1972

(Wohl et al. 2015). At least 37,099 river restoration projects have been completed at a total cost of 14 to 15 billion dollars with greater than one billion dollars spent annually (Bernhardt et al. 2005; Palmer et al. 2007). Most projects have the stated goal/s of increasing biodiversity and/or enhancing water quality, and the most commonly employed methods to achieve these goals are channel reconfiguration and/or in-stream habitat modification, both of which increase habitat heterogeneity (Palmer et al. 2010; Palmer et al. 2014). Despite these efforts, a 2009 EPA report, based on ~564,000 assessed miles of flowing waters (16% of U.S. rivers and streams), showed a decline in river/stream health - the first downward trend since the passage of the Clean Water Act (Palmer & Allan 2006; EPA 2009). And as of 2022, 51% of assessed U.S. river/stream miles have been found to be impaired (Environmental Integrity Project 2022). What accounts for this discrepancy? Most river restoration projects occur at the reach scale, focusing on less than 1 kilometer (km) of river. A reach-scale focus, however, may neglect the source/s of impairment, which often span a greater distance into the watershed (Palmer et al. 2014). There has also been an emphasis on improving in-stream habitat heterogeneity. Palmer et al. (2010) found that many project managers assume an increase in habitat heterogeneity will enhance water quality and lead to an increase in biological heterogeneity (i.e. biodiversity). This assumption has been termed the “Field of Dreams” hypothesis (Palmer et al. 1997) and appears to be based on classic ecological theory, specifically that species diversity has a positive correlation to increased habitat heterogeneity (Palmer et al. 2010). However, there is scant evidence to support the theory that an increase in habitat heterogeneity via structural modifications will improve water quality or biological integrity in restored flowing water systems; on the contrary, a wealth of evidence supports no or insignificant effects (Roni et al. 2008; Haase et al. 2011; Lorenz & Feld 2013; Palmer et al. 2014). The reasons why this common type of river restoration fails to produce ecological improvement are myriad, though most fall into the category of overriding catchment degradation. In other words, impairment, from anthropogenic disturbance, current and historical, occurring upstream and throughout the immediate watershed, overrules local improvements (Feld et al. 2011; Palmer et al. 2014). Currently a major restoration effort is underway in Michigan's (MI, USA) longest river, the Grand River, where it flows through downtown Grand Rapids. The Grand River begins in southern MI and flows in a northwesterly direction through the state capital (Lansing) and, eventually, Grand Rapids, toward its confluence with Lake MI. Along the river's 406 km course, forests and wetlands have been historically reduced by a combined 75%, and the river is impacted by agricultural lands (57% of the total watershed), residential areas, and recurring urbanization (9% total) (Hanshue & Harrington 2017). In downtown Grand Rapids, the Grand River is typical of other rivers that run through major urban centers: it is a channelized, dredged, and dammed river with a nearly 100% impervious floodplain (Hanshue & Harrington 2017). This uniform, high-gradient river channel once contained a series of rapids with a purported fall of 18 feet from which the city took its name (Baxter 1891). The rapids were demolished in the 1860s as part of a navigation project which was never completed (Hanshue & Harrington 2017). Grand Rapids Whitewater (GRWW), a not-for-profit organization spearheading the restoration, plans to restore the rapids by 1) removing dams from the downtown section of the Grand River to achieve a more natural flow regime; and 2) augmenting the substrate with boulders, riffle-pool-glide sequences, and "diversified hydraulic structures" (grandrapidswhitewater.org; See also Burkman & Ogilvie 2022). An adjustable hydraulic structure will also be added to block upstream migration of invasive sea lamprey. The restoration will occur in two phases, the first to begin as early as summer 2023 (Burkman & Ogilvie 2022). Phase 1 involves demolition of four low-head dams (weirs) and creation of riffle-pool-glide sequences and "diversified hydraulic structures" across an 840 meter stretch designated the Lower Reach Project Area (LRPA) (grandrapidswhitewater.org); funds have also been allocated to address excess sediment loading (from agriculture) in two subwatersheds upstream of the LRPA (Burkman & Ogilvie 2022). GRWW's stated goals are 1) rebuilding of the rapids for recreational purposes and, 2) ecological improvement of the Grand River's downtown stretch (grandrapidswhitewater.org, See Mission Statement).

Ecological improvement should be the goal of all river restoration efforts as it inheres in the very term (Palmer et al. 2005). Ecological improvement, or success, is defined as "measurable changes in physicochemical and biological components of the target river…that move towards the agreed upon guiding image [of the project]" (Palmer et al. 2005: 211). In order to gauge ecological impact, improvement or otherwise, a pre- and post-restoration assessment of habitat and/or focal taxa is necessary (Palmer et al. 2005). The assessments may be used in a simple pre-/post-comparison or analyzed as part of a statistical design. Often, such assessments employ a Before-After-Control-Impact-Pairs design (BACIP) (Stewart-Oaten et al. 1986 and 1992; and modified by Underwood 1992) where a site upstream of the restoration (a control) is sampled along with the restoration site (the impact reach), before and after project completion. The data is then paired for statistical analysis. The BACIP is especially relevant to ecological field studies/biological assessments as it does not rely on replication of the perturbation (e.g. re-damming a river), but rather on replicates in time and space of perturbation and non-perturbation before and after restoration efforts. An exemplary model of a BACIP in action has been extensively documented by the Kissimmee River Restoration Evaluation Program (KRRP) (Bousquin et al. 2005).

We conducted a pre-restoration survey of the LRPA in downtown Grand Rapids (= impact reach) with concomitant survey of an upstream reach in Sunfield, MI (= control). Our Control reach was chosen because it was the highest scoring Grand River site, for macroinvertebrates and habitat, sampled by The Michigan Department of Environment, Great Lakes, and Energy (EGLE) in a recent water quality report (EGLE, 2012; see Station 24). Benthic macroinvertebrates were chosen as the focal taxa of our survey because they: 1) make up the base of the faunal food web; 2) occupy nearly every trophic level; and 3), as poor dispersers with a wide range of tolerance levels to various impacts, integrate disturbance over time as a community and, thus, serve as a proxy for ecological integrity of flowing water systems (Merritt et al. 2019). To our knowledge, comprehensive macroinvertebrate data does not exist for the LRPA. Our primary goal was to establish viable baseline data for this important indicator group. These data may serve as the before in a BACIP, which could help to ultimately assess the rapids restoration effort. Beyond the deliverable of robust baseline data, we herein attempt to fully characterize the impact reach with reference to its control. Finally, we make some predictions on the ecological status of benthic macroinvertebrates, post-restoration, and the ecological outcomes of the restoration, based on our data and the best available literature.

Methods

Study locations/reaches

Our study focused on two reaches within the Grand River Watershed, an impact reach and a control. Our impact reach was a section of the Grand River that flows through downtown Grand Rapids (Fig. 1), 831 meters in length. This reach spans nearly the entire area proposed for restoration during Phase I; it also includes four low-head dams (weirs) slated for removal (grandrapidswhitewater.org; Burkman & Ogilvie 2022). We refer to this sampling reach of the LRPA, hereafter, as the impact reach. The impact reach falls within the 12,690 km^2 drainage area of USGS gauge 04119000 (42°57'47", -85°40'38"), which is located 0.32 kilometers downstream of our first sampling site. Mean annual discharge for 2021 was 97 m^3/s; average daily discharge of the 7 day period preceding our sampling was 107 m^3/s. The floodplain of this reach is completely disconnected from the river and is composed primarily of impervious surfaces.

Our control reach encompassed a 230 meter (m) stretch of the Grand River in Sunfield, Michigan (Fig. 2) at the Portland State Game Area (length of the control reach was constrained by accessibility). This reach is 124.30 river km upstream of the impact reach and received high macroinvertebrate and habitat scores from prior state-administered sampling efforts (EGLE 2012). The USGS gauge nearest our control reach is gauge 04114000 (42°51'23", -84°54'44"), 9.7 km downstream of our first sampling site, with a drainage area of 3587 km^2. Mean annual discharge for 2021 was 25.5 m^3/s; average daily discharge of the 7 day period preceding our sampling was 33.9 m^3/s. The floodplain has a high degree of connectivity with the river and the riparian zone is densely forested, though large agricultural fields occur within 180 m upstream on the left bank.

Sampling design

We sampled five sites at the impact location (n=5) and five sites at the control (n=5) as replicates to represent the “Before” (n=10) in a Before-After-Control-Impact-Pairs design (BACIP). Sampling at the impact reach occurred on 27 July 2021, no significant rain event having occurred at least 7 days prior. To mitigate the possible effects of the low-head dams on macroinvertebrate occurrence, we chose a site approximately halfway between each dam and one of a similar distance upstream of the first low-head dam (n=5). Each site was no closer than 3 m from shore and no deeper than 60 cm (3 m from shore, generally, correlated to good sampling substrate (diverse rocks) and a level benthos for net placement. We took two velocity measurements (benthic and surface) using a OTT MF pro flow meter with 2 m velocity/depth sensor and 1.2 m wading rod before each sampling event.

Sampling at the control reach occurred on 28 July 2021, no significant rain event having occurred at least 7 days prior. Again, each site was no closer than 3 m from shore and no deeper than 60 cm; a distance of 10 m was used, initially, as an arbitrary spacing between sites; however, this varied when appropriate substrate was lacking, in which case the distance was doubled. Velocity measurements were taken as above. Our sampling dates coincided with, and occurred within, the sampling time-frame recommended by the state-government assessment methodology for nonwadeable rivers (EGLE 2013).

Benthos was sampled with a Stanford-Hauer kicknet (Fig. 3) composed of 250 micron Nitex mesh netting. The opening was 1 m x 1 m and tapered to a removable cod bucket with a 9 cm diameter; this device sampled 0.5 m^2 of stream bottom. Large black tick marks at the interior base of the kicknet allowed the operator to gauge the sampling area. At each site, sampling was performed by the same operator; for any moveable rock, all edges were scrubbed with gloved hands (neoprene) to reduce damage to specimens, and the rock set aside; all available surface area of boulders and macrophytes were scrubbed; and, finally, any small rocky substrate or detritus was disturbed lightly by hand and foot. The net was then closed and rinsed down from the outside to move as many organisms as possible into the cod bucket. The bucket was dumped and rinsed into a 12” diameter, 250 micron Gilson sieve; all remaining insects were picked from the net with forceps and added to the sieve; the sieve contents were then scooped into a Whirl-Pak, containing 95% EtOH, labeled, and sealed. After 24 hours, samples were decanted of EtOH in the laboratory and refilled with 70% EtOH to avoid desiccation and permit flexibility during identification.

A Wolman pebble count was conducted at each sampling location (n=100) to ascertain D_{50} particle size and percent rocky substrate (i.e. total substrate minus percent fines) (Wolman 1954).

Laboratory processing

Each sample was subsampled using a methodology suggested by Dr. F. Richard Hauer (personal communication, 2021) and similar to that recommended by Barbour et al. (1999), though simplified per equipment. First, samples were emptied into the Gilson sieve and rinsed in a large sink; any large or rare organisms were picked and placed in a 1:1 specimen cup with 70% EtOH. Second, samples were floated in a few inches of water and swirled to homogenize, the sieve being carefully lifted from the water. Third, the samples were quartered using a Plexiglass divider, and a quarter was selected at random; the quarter was then scooped into a tray and rough-sorted to order and/or family in multiple 4:1 specimen cups with 70% EtOH. Select chironomids from each sample were mounted using CMCP-10 after the methodology of Epler (2001). Chironomid slides were identified using a Nikon Eclipse Ci microscope; all other specimens were identified using a Nikon SMZ645 scope, and a Nikon DS-L3 when necessary. Chironomids were identified to subfamily, simuliids to family, and all other insects to genus with Merritt et al. (2019); non-insect taxa were identified to genus with Pennak (1989), excepting gastropods and annelids.

Data analysis

For each site, and each location, taxa richness, abundance, percent EPT, percent Diptera, percent Hydropsychinae of Trichoptera, and Shannon's Diversity Index were calculated. Taxa were also assigned to Functional Feeding Groups (FFGs) using Merritt et al. (2019) and Hauer and Lamberti (2017). FFGs were calculated as abundances and proportions per site and location. One-way analysis of variance (ANOVA), with location as a factor, was used to compare the metrics and index listed above, as well as velocity at the benthos. ANOVA was our preferred univariate method of analysis because, given our small sample size (n=5 at each location), our data was often not normal even after transformation; however, the assumption of homogeneity of variance was met in all cases using standard EDA techniques. Percent data was arcsine transformed, when necessary (Sokal & Rohlf 1987), prior to testing with ANOVA.

The Procedure 22 (P22) biotic index, from the EGLE assessment methodology of the same name, was also applied as a means of comparing our two locations (EGLE 2013). The P22 biotic index is the best, readymade index for scoring integrity of reaches in large, Michigan rivers (EGLE 2013). The P22 index includes 8 metrics (FFG diversity, % Trichoptera, Total Taxa Richness, etc.) which are scored, summed, and then compared to a range of ecological integrity levels (excellent, good, marginal, or poor), based on reference conditions, for rating (EGLE 2013). Ultimately, the sum of scores, as relates to the range of ecological integrity levels, is only valid if a requisite qualitative sampling methodology was performed. However, any benthic macroinvertebrate dataset, especially from a large river, could hypothetically be evaluated with the index so long as a control site was used for comparison. In this way, we used the P22 biotic index to compare our locations in the same way other diversity indices, such as

Shannon's or Simpson's, are used to compare sites, systems, locations, etc. (i.e. the higher score should equate to greater biodiversity/ecological integrity).

Non-metric multidimensional scaling (NMDS) was performed to examine community structure at locations and sites with a Bray-Curtis Dissimilarity matrix. A one-way analysis of similarity (Anosim) was used to test for differences between locations. All statistical analyses were conducted in R, Version 4.1.1 (R Core Team 2021).

To understand how land cover may influence water properties within each study area, we used ArcGIS Pro 3.0 (Environmental Systems Research Institute, Redlands, California, USA) to classify imagery and compute the proportion of 4 land cover types within 300 meters laterally from the Grand River, and within a length extending 1 km downstream and 5 km upstream from the sampling sites. We used 4-band infrared imagery (1-m resolution) from the National Agriculture Inventory Program circa 2020 (National Agriculture Imagery Program (NAIP) Digital Object Identifier (DOI) number: /10.5066/F7QN651G) to classify land cover into 5 categories based on spectral signatures of training samples. We used the maximum likelihood estimator for the final classification, and improved accuracy of the classification by using the pixel editor to reclassify erroneous pixels for an overall classification accuracy of 91%. The categories included agriculture, open (vegetated with grass or herbaceous plants), forest (areas dominated by deciduous or coniferous trees), water, impervious (roads, concrete, buildings).

We then computed Euclidean distance laterally from the edges of the Grand River, and categorized them into distances 100, 200, and 300 m from the edge of the river. We delineated zones extending between the first and last sampling point in the study areas, 1 km downstream from each study area, and upstream 0-1 km, 1-2 km, and 2-5 km. The lateral distance categories were combined with each zone and then used to tabulate area of each land cover type within each distance category and zone. We ultimately computed percentages of land cover types to reveal different landscape characteristics that potentially may impact aquatic organisms within each system.

Results

Substrate / abiotic metrics

The substrates at our two locations differed in several ways. The impact reach was 95% rocky (i.e. greater than fines), with cobbles and small boulders being predominant (D_{50} = 97 mm); small bricks (medium cobble size range, 90-128 mm) were also occasionally present; macrophytes were uncommon, though macro-algae were present. The control reach was 82% rocky, with coarse gravel being predominant (D_{50} = 10 mm); rocky substrates were distributed nonuniformly among woody or detrital silt. Macrophyte beds were common, though macro-algae were rare.

The mean benthic velocity of the impact reach was 0.26 m^3/s, and 0.10 m^3/s for the control, though a one-way ANOVA, with location as a factor, yielded an insignificant p-value ($p > 0.05$) (See Table 1).

General taxonomic composition / biotic metrics, indices

Each reach was dominated by three taxa, respectively, which made up, on average, at least 64% of the total individuals/abundance at each site. The dominant impact reach groups were Chironomidae, Simuliidae, and the mayfly *Tricorythodes* (Leptohyphidae). *Tricorythodes* was also one of the three dominant taxa in the control reach, though the others were the amphipod *Gammarus fasciatus* and the riffle beetle *Stenelmis* (Elmidae) (Table 2). The impact reach contained 12 taxa not found in the control reach; the control contained 11 taxa not found in the impact (See Tables 3 and 4). Of those taxa distinct to the impact reach, *Caecidotea* (Asellidae) and Simuliidae were the most abundant, while *Corbicula* (Cyrenidae) and Sphaeriidae were the most abundant taxa unique to the control.

Initially, four metrics and one index were used to evaluate biotic integrity at our two locations: percent Diptera, percent EPT, abundance, taxa richness, and Shannon's Diversity (See Table 1). Comparison of percent EPT between locations, as well as a disproportionate number of Hydropsychinae (See Figure 4), lead us to include Percent Hydropsychinae of Trichoptera as an additional metric. Percent Hydropsychinae of Trichoptera (comprising genus *Hydropsyche* and *Cheumatopysche*) is a metric used to evaluate environmental stress (perturbation), with values greater than 85% indicating impairment (Stagliano 2020). Percent Hydropsychinae of Trichoptera was 83% for the downtown reach and 23% for the control. A One-Way ANOVA showed a statistically significant difference between locations (ANOVA: $p<0.05$) (Table 1). Percent Diptera is another metric used to evaluate environmental stress/biological impairment and is expected to increase as perturbation increases. Percent Diptera for the downtown reach was 38%, on average (See Figure 5), with two families, Chironomidae and Simuliidae, making up almost 100% of all dipteran taxa. Percent Diptera for the control was 3%. A One-Way ANOVA showed a statistically significant difference between locations ($p<0.05$) (Table 1). Taxa richness, percent EPT, and Shannon's Diversity were not statistically significant between locations ($p>0.05$) (See Table 1). Abundance was also not statistically significant between locations. However, the control reach was found to support 489 individuals per $0.5m^2$, on average, vs. the impact, which supported 388 individuals per $0.5m^2$. The most abundant control reach site (C-Site 1) supported 796 individuals vs. the most abundant impact reach site (Im-Site 4), which supported 790.

Percent FFGs differed, somewhat, between sites, though Percent Shredders was the only statistically significant group ($p < 0.05$) with 25% at the control vs. 3% at the impact (See Table 1). *G. fasciatus* made up 100% of the control reach shredder community and nearly 100% of the infinitesimal impact reach shredder community.

Initial P22 index scores, using a standard series of eight metrics, were 60 for the control and 58 for the impact (see Table 5). Considering the large difference in % Hydropsychinae between locations, we adjusted the third metric, % Trichoptera, by subtracting Hydropsychinae from total Trichoptera, and then recalculated the index. P22 index ratings, using the Hydropsychinae adjustment, were 60 for the control and 45 for the impact (see Table 5).

NMDS / Community structure

Non-metric multidimensional scaling (NMDS) was used to examine community structure between the control and impact reaches. Our ordination plot reveals a clear distinction between locations with impact and control sites segregating to separate sides (stress = 0.057) (See Figure 7). An Anosim was run and indicated that our two locations were significantly different ($p<0.05$).

Impact reach sites (abbreviated as "Im-Sites") formed a general *cloud* in the upper left corner of the ordination plot with Im-Sites 1 and 4, and Im-Sites 2 and 3, respectively, forming less distant pairs (See Figure 7). Im-Sites 1 and 4 both had similar numbers of simuliids and *Tricorythodes*, two of the three dominant impact reach taxa; while Im-Sites 2 and 3 shared similar numbers of *Tricorythodes* and chironomids, chironomids being the other dominant taxon. These taxon-abundance similarities between sites may explain the closer proximity of each less distant pair.

Three control reach sites (abbreviated as “C-Sites”) - C-Site 1 (the farthest downstream site) and C-Sites 4 and 5 (the farthest two upstream sites) - formed a loose cluster near the upper-right corner of the ordination plot (See Figure 7); the other two sites (C-Sites 2 and 3) were far removed from the cluster (bottom right), and at a moderate distance from each other. C-Sites 1, 4, and 5 had various rocky substrates along with sparse macrophytes and supported a high abundance of *Stenelmis* as well as three other genera of elmids, several mayfly genera, and one caddisfly genus (*Brachycentrus*). Presence of these taxa, as supported by said substrate, ultimately explains the clustering of C-Sites 1, 4, and 5, and distance from C-Sites 2 and 3, on the ordination plot. C-Sites 2 and 3 were composed, primarily, of detrital silt and supported low numbers of various taxa except *G. fasciatus*.

Habitat / Land-cover-type proportions

For brevity, we report only the land-cover-type proportion results for three zones - sampling area, 0-1 km upstream, and 1-2 km upstream - at a lateral distance of 300 m. For full land-cover-type proportion data, see Appendix 1 (Tables 3 and 4A). Land-cover-type proportions within the impact reach sampling area (.83 km), considering a 300 m lateral distance from the wetted bank, was 90.1% impervious, 5.9% forested, and 4.0% open. Land-cover-type proportions 0-1 km upstream of the impact reach sampling area, considering a 300 m lateral distance from the wetted bank, was 81.5% impervious, 9.8% forested, and 8.7% open. Land-cover-type proportions 1-2 km upstream of the impact reach sampling area, considering a 300 m lateral distance from the wetted bank, was 82.7% impervious, 6.8% forested, 10.5% open.

Land-cover-type proportions within the control reach sampling area (.23 km), considering a 300 m lateral distance from the wetted bank, was 0% impervious, 79.8% forested, and 20.2% open. Land-cover-type proportion 0-1 km upstream of the control reach sampling area, considering a 300 m lateral distance from the wetted bank, was .2% impervious, 83.7% forested, 3.9% open, and 12.2% agricultural. Land-cover-type proportion 1-2 km upstream of the control reach sampling area, considering a 300 m lateral distance from the wetted bank, was 3.8% impervious, 62.4% forested, 2.7% open, and 31.0% agricultural.

Discussion

Impact reach summary / ecological status

The floodplain habitat of the LRPA is 90.1% impervious and over 80% impervious for at least 2 river km upstream. There is a negligible amount of park space on the right bank (Ah-Nab-Awen Park), near the center of our 0.83 km sampling stretch with limited retention ability (Aaron Parker, EGLE senior aquatic biologist, personal communication). The nearest functional floodplain occurs, beyond a large weir dam, approximately 4 river km upstream at Riverside Park. The channel is constrained by concrete flood walls and earthen embankments throughout the LRPA and entire downtown section (Burkman & Ogilvie 2022). As all available bank is concrete or riprap, there is no natural refugia, such as those found throughout our forested control site (e.g. natural undercuts). This is yet another reason why benthic substrate sampling recommends itself to constrained urban reaches – there is little else to sample, so the numerous bank refugia of a forested reach/control would automatically be biased toward more and diverse taxa.

In contrast to the almost complete lack of riparian habitat and, by extension, lack of large woody debris (LWD), the impact reach has good, in-stream rocky substrate. We found a mix of, primarily, large rocky substrates with very low embeddedness across all sampling sites. The impact site supported a moderate amount of macroinvertebrate individuals per half square meter (388 on average), compared to the control (489). Considering that two of the control reach sites were detrital, with few rocks (C-Sites 2 and 3), the average difference is, perhaps, more impressive. For instance, if the three most abundant sites from each location were compared against each other, the average difference of individuals per half square meter would grow to 200, favoring the control. On the other hand, the unnatural shelter/shading presented by multiple bridges and overpasses may benefit impact reach statistics. For example, Im-Site 4, which is partially shaded by an overpass, contained the highest number of individuals at 790 total and was the only site with adult *Stenelmis*. This site has macrophytes, which can increase macroinvertebrate abundance and richness, depending on the species of macrophyte (Humphries 1996). *Stenelmis* are a facultative genus of riffle beetles (Hilsenhoff 1987) that function as collector-gatherers (via larvae) and scrapers (via adults) (Hauer & Lamberti 2017), so thermal preference and food source probably made the difference. Still, *Stenelmis* were ⅔ more abundant at C-Site 1 than Im-Site 4 where chironomids and *Tricorythodes* dominated. Despite this major difference in *Stenelmis* individuals between locations, overall macroinvertebrate abundance was not found to be significantly different. The two metrics that best differentiated the impact and control reach locations, statistically and descriptively, were percent Diptera and percent Hydropsychinae of Trichoptera.

Diptera was the dominant order of the impact reach with chironomids and simuliids averaging 38% of all taxa at each site (See Figure 6). A few larval dipteran families, especially

Chironomidae, are highly tolerant of nearly all perturbation and, when occurring in high numbers, indicate poor water quality and/or environmental stress (Voshell 2002). Thus, percent Diptera is predicted to increase as perturbation increases (Barbour et al. 1999; Mandaville 2002). Chironomidae is, however, a varied family. For example, the common subfamilies Chironominae, Orthocladiinae, and Tanypodinae all have slightly different tolerance levels to organic pollution, at least, and Tanypodinae occupy a different feeding group as predators (Mandaville 2002; Merritt et al. 2019). Orthocladiinae are considered somewhat tolerant collector-gatherers vs. the slightly more tolerant Tanypodinae, and tolerant Chironominae (Mandaville 2002). Our taxonomic survey found that 50% of impact reach chironomids belonged to Orthocladiinae; and greater than half of all impact reach Orthocladiinae occurred at Im-Site 4. We believe that the abundance of Orthocladiinae at Im-Site 4 is explained by the same factors explaining *Stenelmis* (namely, cooler temperatures and macrophytes). Excepting this site, Chironominae were found at greater numbers, on average, throughout the impact reach. Simuliids (black flies), a tolerant-to-facultative family depending on perturbation type (Voshell 2002; Mandaville 2002), made up nearly the entire remainder of dipterans (35%). Simuliids are, primarily, collector-filterers and known to increase with moderate amounts of organic and/or nutrient pollution (Voshell 2002). Taken together, chironomids (64%) and simuliids (35%) made up nearly 100% of impact reach dipterans (See Figure 6).

Hydropsychinae (comprising genera *Hydropsyche* and *Cheumatopsyche*) were not one of the three most dominant impact reach groups, though they were fourth in terms of relative abundance (10%). Hydropsychinae, a subfamily of Hydropsychidae (common net-spinning caddisflies), are facultative collector-filterers which are predicted to increase as perturbation increases (Barbour et al. 1999). Like other hydropsychids, Hydropsychinae produce a mesh netting which captures fine particulate organic matter (FPOM). Organic and/or nutrient pollution puts excess particles of FPOM into the water column (Voshell 2002). Hydropsychinae, unlike other hydropsychids, have been found to be tolerant of this sort of pollution and, thus, may be found in large numbers where moderate organic and/or nutrient pollution occur (Voshell 2002; Stagliano 2020). Hydropsychinae made up 83% of the total downtown reach Trichoptera and 23% of the total downtown EPT (See Table 1). Much of the downtown reach EPT was composed of a single genus of mayfly, *Tricorythodes*, from the family Leptohyphidae (little stout crawlers). *Tricorythodes* are also known to be tolerant of disturbance (Stagliano 2020), specifically, increased conductivity, turbidity, and sedimentation (Scherr et al. 2011; Hall 1975). *Tricorythodes*, like other leptohypids, have a pair of operculate gills which protect their functional gills from damage in silty waters (Voshell 2002); their small size is also an advantage in turbulent systems as it likely allows them to exploit a variety of interstitial spaces in and amongst refugia that would not be available to other, larger mayfly larvae. Percent *Tricorythodes* of Ephemeroptera can be used in a similar fashion to percent Hydropsychinae (particularly as a stand-in for family Baetidae when these are not found or found in low numbers) (Stagliano 2020). *Tricorythodes* accounted for 82% of the downtown reach mayfly community and, together with Hydropsychinae, account for 83% of the total downtown EPT.

To summarize, 38% of the impact reach taxa were facultative to very tolerant dipterans (Voshell 2002; Mandaville 2002); 10% were Hydropsychinae; and 26% were *Tricorythodes*. 8% of the remainder were other EPT. By way of comparison, 3% of the control reach taxa were dipterans; 1% were Hydropsychinae; and 24% were *Tricorythodes*. *G. fasciatus* was dominant at 25%, though expected, as a shredder, per River Continuum Theory (Vannote et al. 1980). 20% of the remainder were other EPT. Finally, our P22 index scores – based on similar metrics as those above and largely controlled by % Trichoptera (minus Hydropsychinae) and FFG Diversity – show a 15-point difference, favoring the control. We read this as the difference between a good (control) and a marginal (impact) location (EGLE 2013, see p. 17).

The downtown reach is obviously subject to, at least, moderate impairment; but we are not suggesting that this elucidation of data, or the downtown reach, be considered in isolation. The downtown reach is also the sum of its watershed / many upstream parts. In 2021, we took part in a season-long study of the Grand River that stemmed from headwaters to mouth (Assenmacher 2021). We used state-government procedures to evaluate river conditions via macroinvertebrates and habitat (EGLE 1990; EGLE 2013). The overall finding was a decrease in ecological integrity from upstream to downstream (Assenmacher 2021), consistent with prior state-survey data (EGLE 2016; EGLE 2017). Agriculture, urbanization, industrialization, and the myriad nonpoint source pollutants associated with each, take their exponential toll as the Grand River flows from its source in Somerset County, through the Greater Grand Rapids area, toward its confluence with Lake Michigan.

Restoration measures and potential outcomes

It should be noted that GRWW's first stated goal is rebuilding of the rapids for recreational purposes (grandrapidswhitewater.org, see Mission Statement). This is obviously an anthropocentric, socioeconomic goal that could only impact the long-term ecological health of the Grand River in one way: successful beautification of the river with concomitant recreational utilization could, potentially, garner public support for further restoration efforts/future projects. Although it is outside the scope of our study, we hope this becomes the case.

GRWW's second stated goal is ecological improvement of the Grand River's downtown stretch (grandrapidswhitewater.org, see Mission Statement). In order to support the hypothesis of ecological improvement/success, some increase in biotic index scores, ratings, etc., should be demonstrated (Palmer et al. 2014). Considering our P22 index scores with the Hydropsychinae adjustment, this would simply mean an increase in the current score of 45 to anything greater. If the score more closely resembled the control score of 60 (or the mean of pre-/ post-restoration control scores) this would be ideal. However, achieving this sort of increase is, apparently, the exception to the rule as concerns in-stream, structural restorations.

Palmer et al. (2014) reviewed 149 studies covering a total of 644 stream/river restoration efforts, all of which provided quantitative assessments. Most studies sought to increase biodiversity, habitat, and/or water quality by manipulating channel and/or in-stream hydrogeomorphology. Though structural variables (habitat, channel, velocity, etc.) improved in many cases, 0% of in-stream hydrogeomorphic projects showed improvement in water quality, and only 16% of all projects showed evidence of increased biodiversity (as gauged by biotic indices) (Palmer et al. 2014). Taxa richness was found to increase for projects that only manipulated in-stream hydrogeomorphology, though the added taxa were tolerant species that were characteristic of degraded/urban streams and not characteristic of reference locations (Palmer et al. 2014). In a more recent study, Muhar et al. (2016), synthesizing 10 small and 10 large "good-practice" river restoration efforts across Europe that manipulated in-stream hydrogeomorphology, reported little to no effect on biodiversity or richness of taxa.

The removal of weirs can restore abiotic factors in streams such as longitudinal connectivity and upstream habitat diversity (Feld et al. 2011). However, Feld et al. (2011), considering 31 weir and dam removals, concluded that positive effects of such measures were short-lived and occurred only at the local scale. Lorenz & Feld (2013), analyzing 46 river restorations which employed in-stream hydrogeomorphic measures, including weir removals, found no improvement in aquatic macroinvertebrate metrics. Considering only fish, Roni et al. (2008), investigating 14 published papers on dam removal, stated that various types of dam removals, including weir removals, and diversions around weirs, can be beneficial for "migratory and lotic fishes" (p. 866). However, Roni et al (2008), in speaking of in-stream habitat modifications, added the caveat that the benefits of these projects would be short-lived unless "process-based restoration activities," such as riparian management, were performed in tandem with structural modifications.

It should be noted that, as a secondary restoration measure, funds have been allocated to implement Best Management Practices (BMPs) in two subwatersheds (of the Lower Grand River watershed) upstream. These subwatersheds are Indian Mill Creek, which drains 45 km^2, and the Rogue River, which drains 1210 km^2. These BMP's are intended to reduce sediment loading in agricultural areas in upper portions of both watersheds. Implementation of these measures is necessary and may be impactful for both watersheds if enacted thoroughly. That being said, both watersheds are relatively small compared to the ~12,720 km^2 cumulative drainage of the LRPA, and the larger of the two, the Rogue is 15 river km upstream of the LRPA. Given the size of the Indian Mill Creek watershed and the distance of the Rogue River watershed, we do not believe that these actions will affect the ecological integrity of the LRPA, especially as evaluated by a biotic index such as P22.

As the aforesaid restoration measures are unlikely to increase ecological integrity of the LRPA, this raises the question, what would? Assuming our first goal was to improve ecological integrity via restoration, what measures would be effective in this endeavor? We know that the

LRPA is governed by the same watershed factors that govern other large river reaches (flow regime, longitudinal connectivity, floodplain interconnectedness). We also know that the LRPA, as an impaired urban reach, is impaired by the same factors (degraded processes) as other impaired urban reaches, and that these factors often override local restoration measures/improvements (Feld et al. 2011). These factors include riparian degradation, loss of large woody debris, hydrologic alteration, nonpoint source pollution (especially nutrient enrichment and sedimentation), and point source pollution (Feld et al. 2011; Allan 2004). The only exception, concerning the LRPA, would be large-scale point source pollution, which appears to have been successfully mitigated by the city of Grand Rapids (see www.grandrapidsmi.gov/Government/Departments/Engineering-Department/Grand-River-Water-Quality-Monitoring). Knowing the general factors which would ultimately limit ecological success of a local restoration work, it would be logical to first address riparian degradation since a healthy, complete riparian, alone would help to mitigate, if not potentially solve, the other factors. A healthy riparian would become a source of large woody debris; work as a buffer for runoff/floods; become a potential, if small, floodplain providing refugia; and create a filter for nonpoint and point source pollution. Riparian zone management has in fact been shown to be a highly successful restoration technique. For example, of the 644 restoration projects reviewed by Palmer et al. (2014), approximately 110 were riparian restorations; and 88-100% of these projects reported improvements in system functioning (i.e. restoration of processes). Feld et al. (2018), in a more recent literature review/modelling study, concluded that riparian zone management, in general, offered a “no-regrets management option” with scale-independent benefits, which include reduced light, reduced temperature, increased LWD, and increased coarse particulate organic matter (CPOM). Based on their conceptual model, Feld et al. (2018) offer recommendations for riparian zone width, composition, and coverage range. The authors do stipulate that full knowledge of catchment issues is necessary, via assessment, in order to mitigate impaired upstream conditions which could overrule local restoration benefits. They further state that, if biodiversity is the chief concern, "…riparian restoration should start upstream in the network and then continue further downstream, to aid the subsequent recolonization of restored reaches" (Feld et al. 2018, p. 390). In parallel, but speaking of restoration success in general, Roni et al. (2008) recommends beginning at high quality habitats by first ensuring distinct protection of these natural sources of biological integrity. After this step, water quality and quantity issues (flow restrictions) need to be addressed/mitigated; then watershed processes should be restored (for example, by managing riparian zones, reconnecting floodplains); and, finally, in-stream habitat should be improved. This sequencing of events for successful restorations is based on a literature review of 345 restoration projects and places in-stream hydrogeomorphic work at the end of a concerted restoration effort (Roni et al. 2008). Thus, if GRWW wants to be successful at increasing the ecological integrity of the LRPA, work should begin upstream at the nearest and best source of ecological integrity and continue downstream to the LRPA. Currently, we recognize a potential starting point 44 river km upstream at Lowell, MI (EGLE 2021, Station F). This site received a score of excellent for macroinvertebrates in the most recent EGLE water quality report (2021) and is the closest, best Grand River mainstem source of ecological integrity we are aware of.

Predictions

In its current trajectory, all documents considered, we make the following predictions concerning the rapids restoration:

1) The rapids restoration will improve in-stream habitat heterogeneity of the restored reach.

2) The rapids restoration will not improve indices of macroinvertebrate biodiversity, though it may decrease dominance/increase evenness.

3) The rapids restoration may increase macroinvertebrate taxa richness, though the added taxa will be tolerant species representative of degraded systems.

Conclusions

Our study has collected initial, pre-restoration data from the impact site as well as an upstream control. Ideally, this study will be replicated at least three years after the restoration. Having pre- and post-restoration data from both a control and impact site allows for an accounting of natural variation over time. Whether the restoration is a success or failure, in terms of ecological improvement, will ultimately depend on the methods employed by GRWW and their partners. Results notwithstanding, completing a post-restoration survey to pair with the pre-restoration survey will at least provide data of ecological significance for future restoration efforts. To that end, we hope our sampling methodology is replicated and utilized appropriately.

Acknowledgements

We are grateful to the Graduate School of Grand Valley State University, and Grand Valley State University in general, for funding and support of this endeavor. We thank Dr. F. Richard Hauer and Dr. Jack Stanford for providing our primary sampling apparatus and the training thereof. We thank EGLE senior aquatic biologist, Aaron Parker for sharing data and fielding questions, and EGLE senior GIS analyst, Christopher Vandenberg for providing river km measurements. We would also like to thank our entire lab – Alexis Neff, Taylor Suttorp, and Benjamin Gunnett – for help and support in sampling and laboratory processes. Special thanks to Dr. Meg Woller-Skar for critiquing our NMDS data analysis; to Dr. Ali Locher for creating our maps and providing land-use percentage data; and Dr. Amy Russell for critiquing an initial draft of this manuscript. Last, but certainly not least, we would like to thank our "Student Scholar," Colin Assenmacher, who worked with us every step of the way.

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Table 1. Abiotic and biotic characteristics of the Lower Reach Project Area (= impact) in Grand Rapids, MI, USA, with reference to an upstream control. **Bold** values were statistically different between locations via One-Way ANOVAs (p-value <0.05). All percent data was arcsine transformed when necessary. (%HofT = Percent Hydropsychinae of Trichoptera).

Characteristic	Control reach	Impact reach	p-value (ANOVA)
Abiotic			
Substrate > 2mm	82%	95%	N.A.
D50 (mm)	10	97	N.A.
Avg. depth (cm)	52	58	N.A.
Avg. velocity "b" m/s	0.1	0.26	>0.05
Biotic (totals)			
Taxa richness	29	30	N.A.
Shannon's (H)	2.23	2.25	N.A.
Abundance	2435	1938	N.A.
% EPT	45	44	N.A.
% HofT	23	83	N.A.
% Diptera	3	39	N.A.
% Shredders	25	3	N.A.
Biotic (means)			
Taxa richness	14	15	>0.05
Shannon's (H)	1.91	1.94	>0.05
Abundance	388	489	>0.05
% EPT	39.2	48.2	>0.05
% HofT	**17**	**74.6**	**<0.05**
% Diptera	**4.2**	**38.4**	**<0.05**
% Shredders	**32**	**3**	**<0.05**

"b" = benthic, FFG = functional feeding group

Table 2. The 10 most abundant taxa from the control and impact reach by percentage. *Tricorythodes* sp.(a genus of mayfly) nearly dominated both locations.

Control reach (Sunfield, MI)		Impact reach (G.Rapids, MI)	
Taxa	%	Taxa	%
Gammarus fasciatus	25.2	***Tricorythodes* sp.**	26.7
***Tricorythodes* sp.**	24.2	**Chironomidae**	25.1
***Stenelmis* sp.**	21.8	Simuliidae indet.	13.9
***Stenonema* sp.**	7.4	***Cheumatopsyche* sp.**	6.9
Baetidae indet.	5.2	***Stenelmis* sp.**	6.3
Brachycentrus sp.	4.2	*Caecidotea* sp.	4.6
Chironomidae	2.6	***Stenonema* sp.**	3.5
Ephoron sp.	1.6	*Hydropsyche* sp.	3.2
***Cheumatopsyche* sp.**	1.1	***Gammarus fasciatus***	2.5
Corbicula sp.	1	Oligochaeta indet.	1.7

Bold taxa occurred in both reaches. [indet., indeterminate]

Table 3. Benthic macroinvertebrate inventory for the Lower Reach Project Area of the Grand River, Grand.Rapids, MI (= impact reach). Taxon ID represents the lowest level of taxonomic identification. *Tricorythodes* sp., Chironomidae, and Simuliidae were the three most abundant taxa.

Phylum	Class	Order	Family	Genus	Taxon ID	Count
Annelida	Oligotchaeta	indet.	indet.	indet.	Oligochaeta	33
Arthropoda	**Arachnida**	**Trombidiformes**	**indet.**	**indet.**	**Hydracarina**	**1**
Arthropoda	Crustacea	Amphipoda	Gammaridae	*Gammarus*	*G. fasciatus*	49
Arthropoda	Crustacea	Decapoda	Cambaridae	indet.	Cambaridae	3
Arthropoda	**Crustacea**	**Isopoda**	**Asellidae**	***Caecidotea***	***Caecidotea* sp.**	**91**
Arthropoda	Insecta	Coleoptera	Elmidae	*Dubiraphia*	*Dubiraphia* sp.	4
Arthropoda	Insecta	Coleoptera	Elmidae	*Macronychus*	*Macronychus* sp.	7
Arthropoda	Insecta	Coleoptera	Elmidae	*Stenelmis*	*Stenelmis* sp.	92
Arthropoda	Insecta	Diptera	Chironomidae	indet.	Chironomidae	487
Arthropoda	**Insecta**	**Diptera**	**Empididae**	**indet.**	**Empididae sp.**	**4**
Arthropoda	Insecta	Diptera	indet.	indet.	Unknown Dipteran	4
Arthropoda	**Insecta**	**Diptera**	**Simuliidae**	**indet.**	**Simuliidae**	**269**
Arthropoda	Insecta	Ephemeroptera	Baetidae	indet.	Baetidae	13
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	*Stenacron*	*Stenacron* sp.	21
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	*Stenonema*	*Stenonema* sp.	69
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	*Tricorythodes*	*Tricorythodes* sp.	518
Arthropoda	Insecta	Ephemeroptera	Polymitarcyidae	*Ephoron*	*Ephoron* sp.	10
Arthropoda	**Insecta**	**Lepidoptera**	**Cramidae**	***Petrophila***	***Petrophila* sp.**	**5**
Arthropoda	**Insecta**	**Odonata**	**Coenagrioninae**	**indet.**	**Coenagrioninae**	**4**
Arthropoda	Insecta	Trichoptera	Brachycentridae	*Brachycentrus*	*Brachycentrus* sp.	4
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Cheumatopsyche*	*Cheumatopsyche* sp.	137
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Hydropsyche*	*Hydropsyche* sp.	64
Arthropoda	**Insecta**	**Trichoptera**	**Hydropsychidae**	***Potamyia***	***Potamyia flava***	**4**
Arthropoda	**Insecta**	**Trichoptera**	**Hydroptilidae**	**indet.**	***Hydroptilidae***	**4**
Arthropoda	**Insecta**	**Trichoptera**	**Leptoceridae**	***Ceraclea***	***Ceraclea* sp.**	**12**
Arthropoda	**Insecta**	**Trichoptera**	**Leptoceridae**	***Nectopsyche***	***Nectopsyche* sp.**	**4**

Arthropoda	Insecta	Trichoptera	Leptoceridae	*Oecetis*	*Oecetis* sp.	4
Arthropoda	**Insecta**	**Trichoptera**	**Polycentropodidae**	***Cyrnellus***	***C. fraternus***	**4**
Arthropoda	**Insecta**	**Trichoptera**	**Polycentropodidae**	***Polycentropus***	***Polycentropus* sp.**	**4**
Mollusca	Neotaenioglossa	Gastropoda	Pleuroceridae	indet.	Pleuroceridae	13

Taxon ID represents the lowest level of taxonomic identification. **Bold** taxa occurred in the impact reach only.

[indet., indeterminate]

Table 4. Benthic macroinvertebrate inventory for Sunfield, MI (= control reach). Taxon ID represents the lowest level of taxonomic identification. *G. fasciatus, Tricorythodes* sp., and *Stenelmis* sp. were the three most abundant taxa.

Phylum	Class	Order	Family	Genus	Taxon ID	Count
Annelida	Oligotchaeta	indet.	indet.	indet.	Oligochaeta	33
Arthropoda	**Arachnida**	**Trombidiformes**	**indet.**	**indet.**	**Hydracarina**	**1**
Arthropoda	Crustacea	Amphipoda	Gammaridae	*Gammarus*	*G. fasciatus*	49
Arthropoda	Crustacea	Decapoda	Cambaridae	indet.	Cambaridae	3
Arthropoda	**Crustacea**	**Isopoda**	**Asellidae**	***Caecidotea***	***Caecidotea* sp.**	**91**
Arthropoda	Insecta	Coleoptera	Elmidae	*Dubiraphia*	*Dubiraphia* sp.	4
Arthropoda	Insecta	Coleoptera	Elmidae	*Macronychus*	*Macronychus* sp.	7
Arthropoda	Insecta	Coleoptera	Elmidae	*Stenelmis*	*Stenelmis* sp.	92
Arthropoda	Insecta	Diptera	Chironomidae	indet.	Chironomidae	487
Arthropoda	**Insecta**	**Diptera**	**Empididae**	**indet.**	**Empididae sp.**	**4**
Arthropoda	Insecta	Diptera	indet.	indet.	Unknown Dipteran	4
Arthropoda	**Insecta**	**Diptera**	**Simuliidae**	**indet.**	**Simuliidae**	**269**
Arthropoda	Insecta	Ephemeroptera	Baetidae	indet.	Baetidae	13
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	*Stenacron*	*Stenacron* sp.	21
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	*Stenonema*	*Stenonema* sp.	69
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	*Tricorythodes*	*Tricorythodes* sp.	518
Arthropoda	Insecta	Ephemeroptera	Polymitarcyidae	*Ephoron*	*Ephoron* sp.	10
Arthropoda	**Insecta**	**Lepidoptera**	**Cramidae**	***Petrophila***	***Petrophila* sp.**	**5**
Arthropoda	**Insecta**	**Odonata**	**Coenagrioninae**	**indet.**	**Coenagrioninae**	**4**
Arthropoda	Insecta	Trichoptera	Brachycentridae	*Brachycentrus*	*Brachycentrus* sp.	4
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Cheumatopsyche*	*Cheumatopsyche* sp.	137
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Hydropsyche*	*Hydropsyche* sp.	64
Arthropoda	**Insecta**	**Trichoptera**	**Hydropsychidae**	***Potamyia***	***Potamyia flava***	**4**
Arthropoda	**Insecta**	**Trichoptera**	**Hydroptilidae**	**indet.**	***Hydroptilidae***	**4**
Arthropoda	**Insecta**	**Trichoptera**	**Leptoceridae**	***Ceraclea***	***Ceraclea* sp.**	**12**
Arthropoda	**Insecta**	**Trichoptera**	**Leptoceridae**	***Nectopsyche***	***Nectopsyche* sp.**	**4**
Arthropoda	Insecta	Trichoptera	Leptoceridae	*Oecetis*	*Oecetis* sp.	4

Arthropoda	**Insecta**	**Trichoptera**	**Polycentropodidae**	***Cyrnellus***	***C. fraternus***	**4**
Arthropoda	**Insecta**	**Trichoptera**	**Polycentropodidae**	***Polycentropus***	***Polycentropus* sp.**	**4**
Mollusca	Neotaenioglossa	Gastropoda	Pleuroceridae	indet.	Pleuroceridae	13

Taxon ID represents the lowest level of taxonomic identification. **Bold** taxa occurred in the impact reach only.

[indet., indeterminate]

Table 5. Prerestoration assessment data evaluated via Procedure 22 metrics. Traditional series metrics/calculations (left) vs. metrics with a Hydropsychinae adjustment (right). Adjusting for Hydropsychinae (subtracting Hydropsychinae from % Trichoptera) makes a significant change between values, scores, and totals for the Impact Location (=downtown reach). (Note: table data represent Location totals).

Traditional series

METRIC	Impact Location		Control Location	
	Value	Score	Value	Score
1. FFG diversity	1.3	8	1.59	16
2. FFG surrogate	0.43	8	0.2	8
3. % Trichoptera	**0.13**	**20**	**0.06**	14
4. EPT taxa richness	15	8	12	8
5. Total taxa richness	29	7	28	7
6. Diptera taxa richness	3	2	3	2
7. Plecoptera taxa richness	0	0	0	0
8. % Dominance	0.27	5	0.24	5
Total		**58**		60

w/ Hydropsychinae adj

METRIC	Impact Location		Control Location	
	Value	Score	Value	Score
1. FFG diversity	1.3	8	1.59	16
2. FFG surrogate	0.43	8	0.2	8
3. % Trichop w/Hyd adj	**0.02**	**7**	**0.05**	14
4. EPT taxa richness	15	8	12	8
5. Total taxa richness	29	7	28	7
6. Diptera taxa richness	3	2	3	2
7. Plecoptera taxa richness	0	0	0	0
8. % Dominance	0.27	5	0.24	5
Total		**45**		60

Gray fill / bold highlights differing scores between series

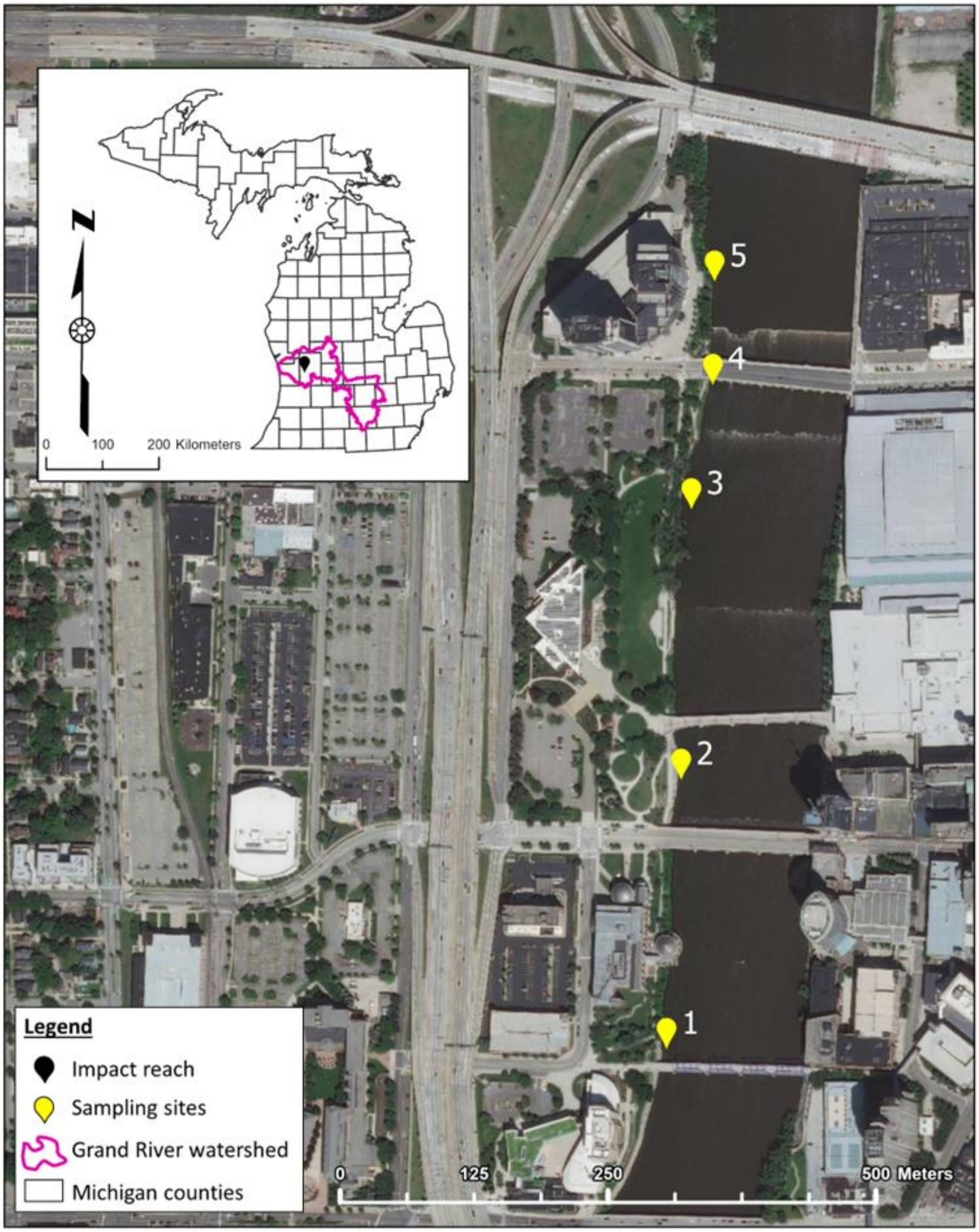

Figure 1. Pre-restoration benthic macroinvertebrate sampling in the Grand River (impact reach). Our Impact Reach encompassed nearly the entire Lower Reach Project Area (840 m) of the Grand River, downtown Grand Rapids, MI, including four low-head dams.

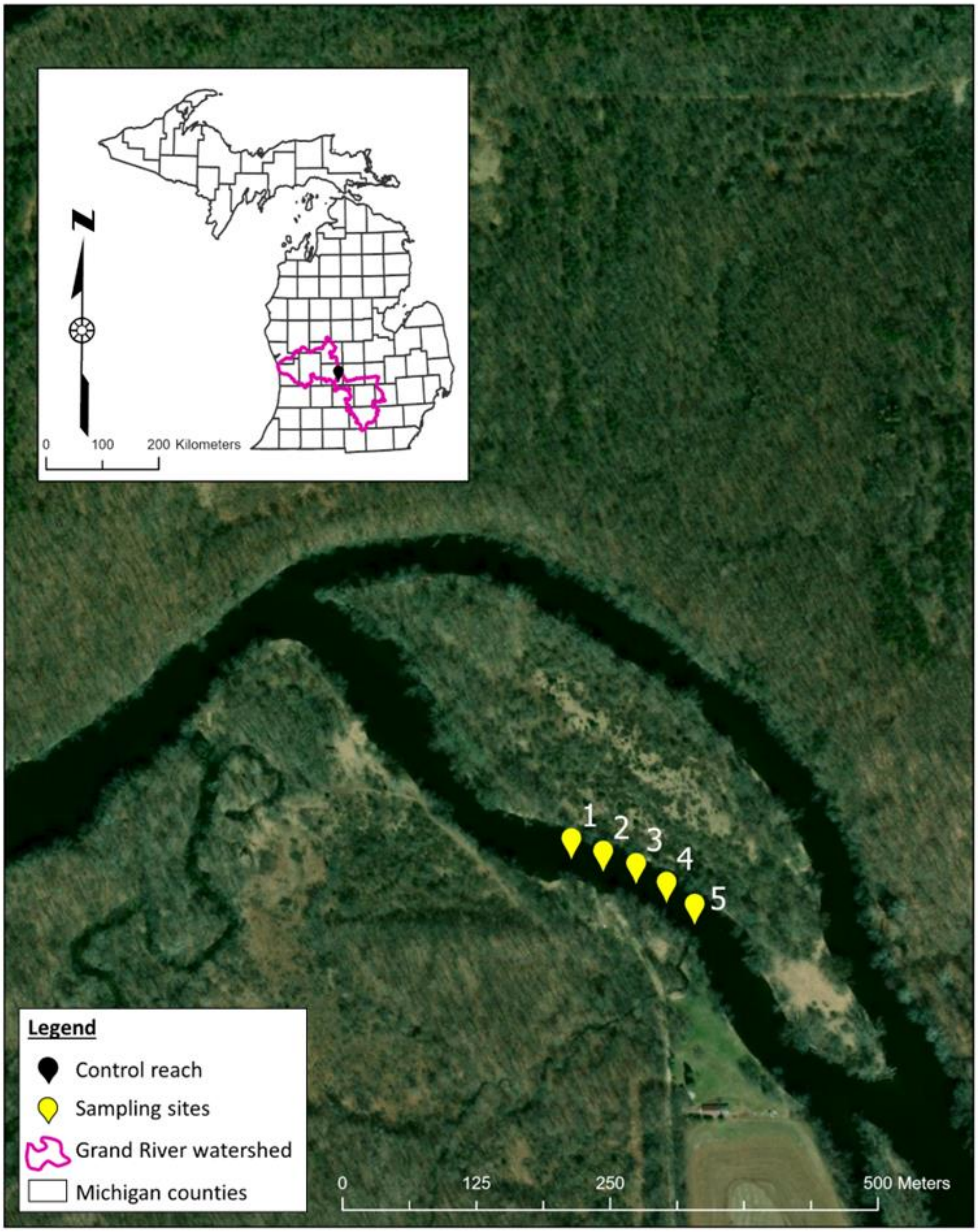

Figure 2. Pre-restoration benthic macroinvertebrate sampling in the Grand River (Control reach). Our Control reach encompassed a 230 m stretch of the Grand River within the Portland State Game Area.

Figure 3. Stanford-Hauer kicknet in operation in the Impact reach of the Grand River, Grand Rapids, MI. 0.5 m^2 of benthos was sampled: first, medium to large substrate was sampled with gloved hands and set aside; next, immovable substrate was hand-scrubbed; and, finally, remaining substrate was kick-sampled for approximately 30 seconds.

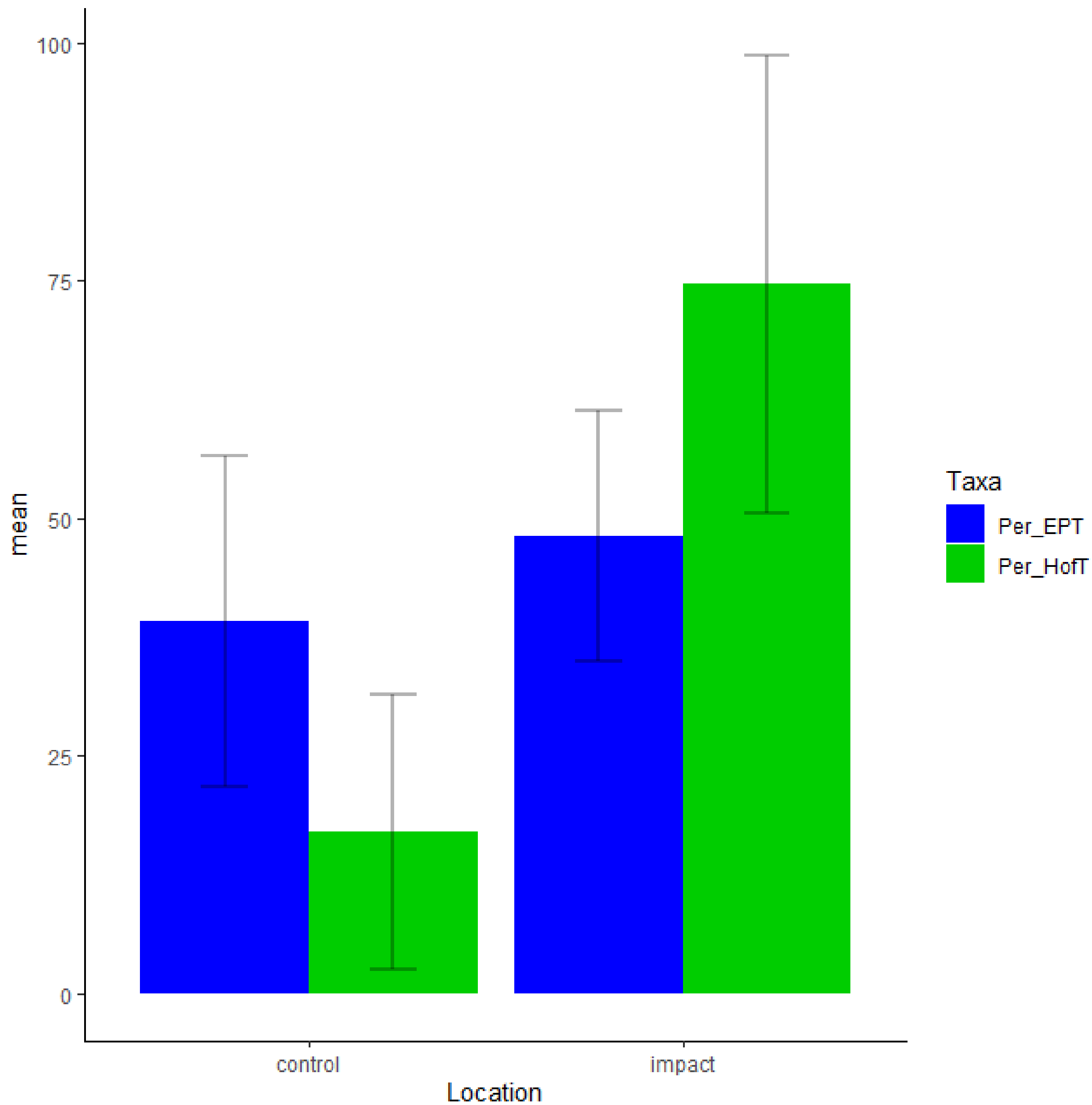

Figure 4. Percent EPT (Ephemeroptera, Plecoptera, Trichoptera) and Percent Hydropsychinae of Trichoptera in the impact and control reaches (mean ± s.d.). Percent Hydropsychinae of Trichoptera was significantly different between locations via One-Way ANOVA ($p<0.05$), while Percent EPT ($p>0.05$) was not.

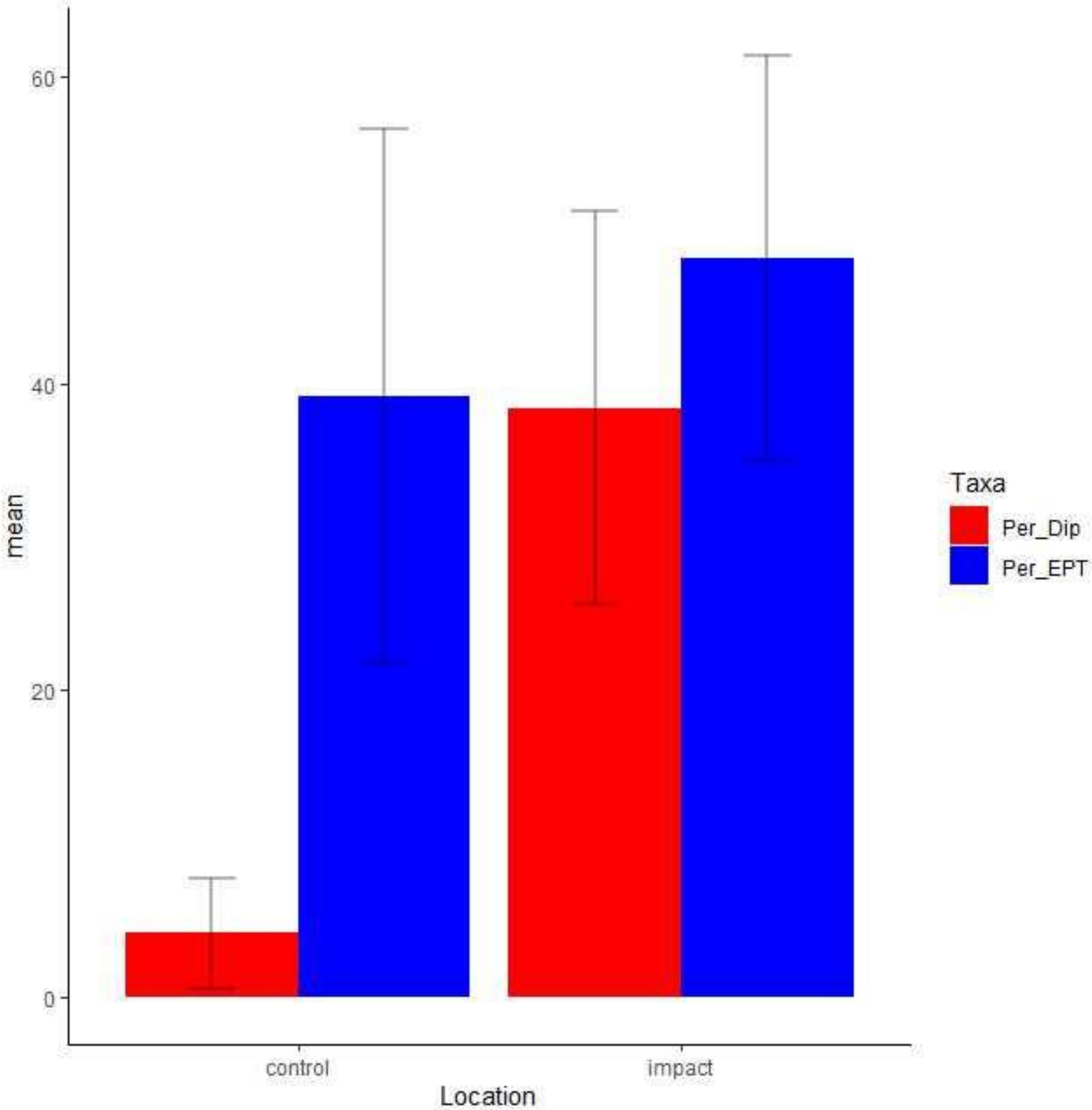

Figure 5. Percent EPT (Ephemeroptera, Plecoptera, Trichoptera) and Percent Diptera in the impact and control reaches (mean ± s.d.). Percent Diptera was significantly different between locations via One-Way ANOVA ($p<0.05$), while Percent EPT ($p>0.05$) was not.

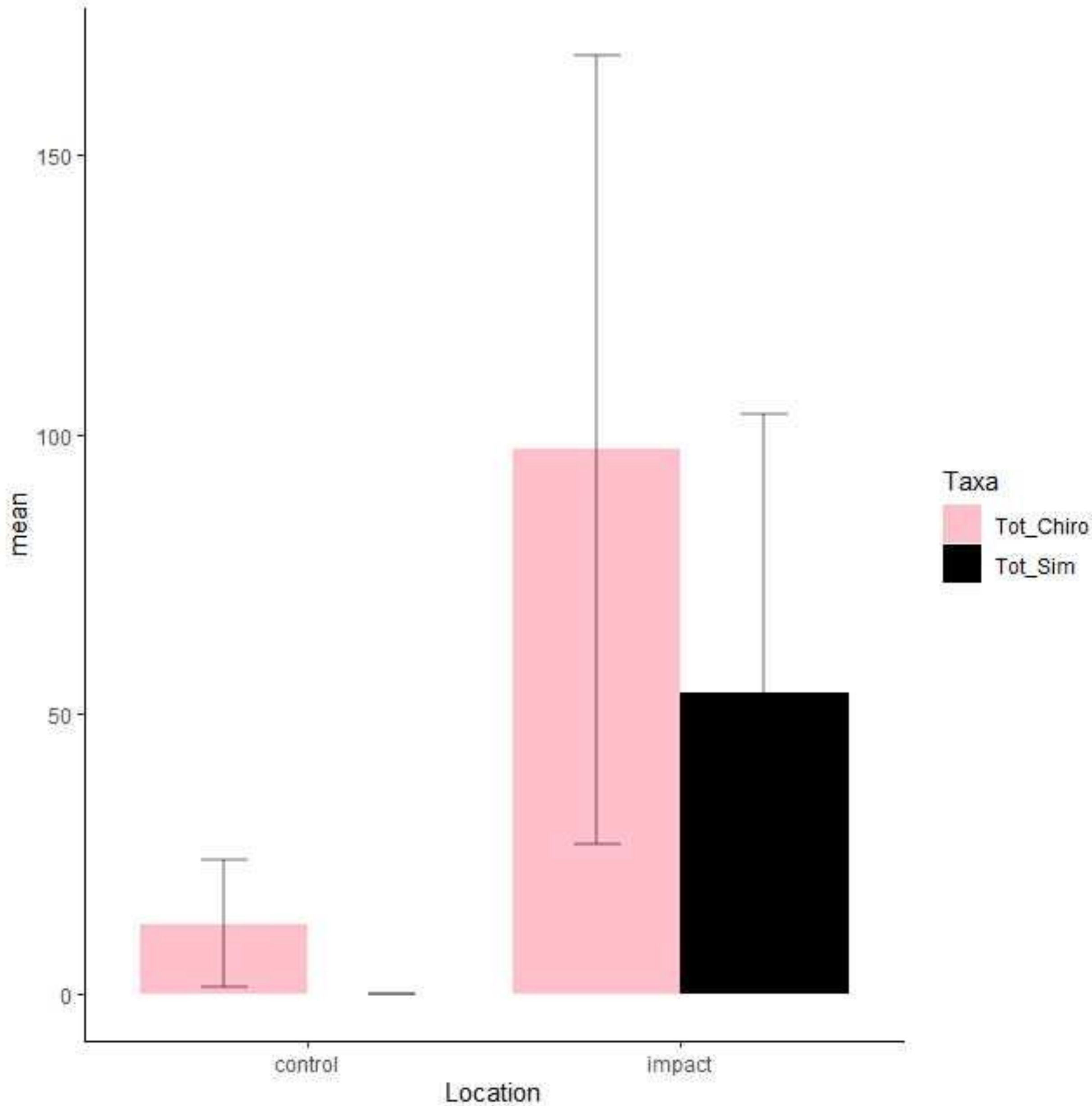

Figure 6. Total chironomids (Tot_Chiro) and total simuliids (Tot_Sim) in the impact and control reaches (mean ± s.d.). Total chironomids was significantly different between locations via One-Way ANOVA ($p<0.05$); total simuliids was 0 for the control reach so no statistic could be run.

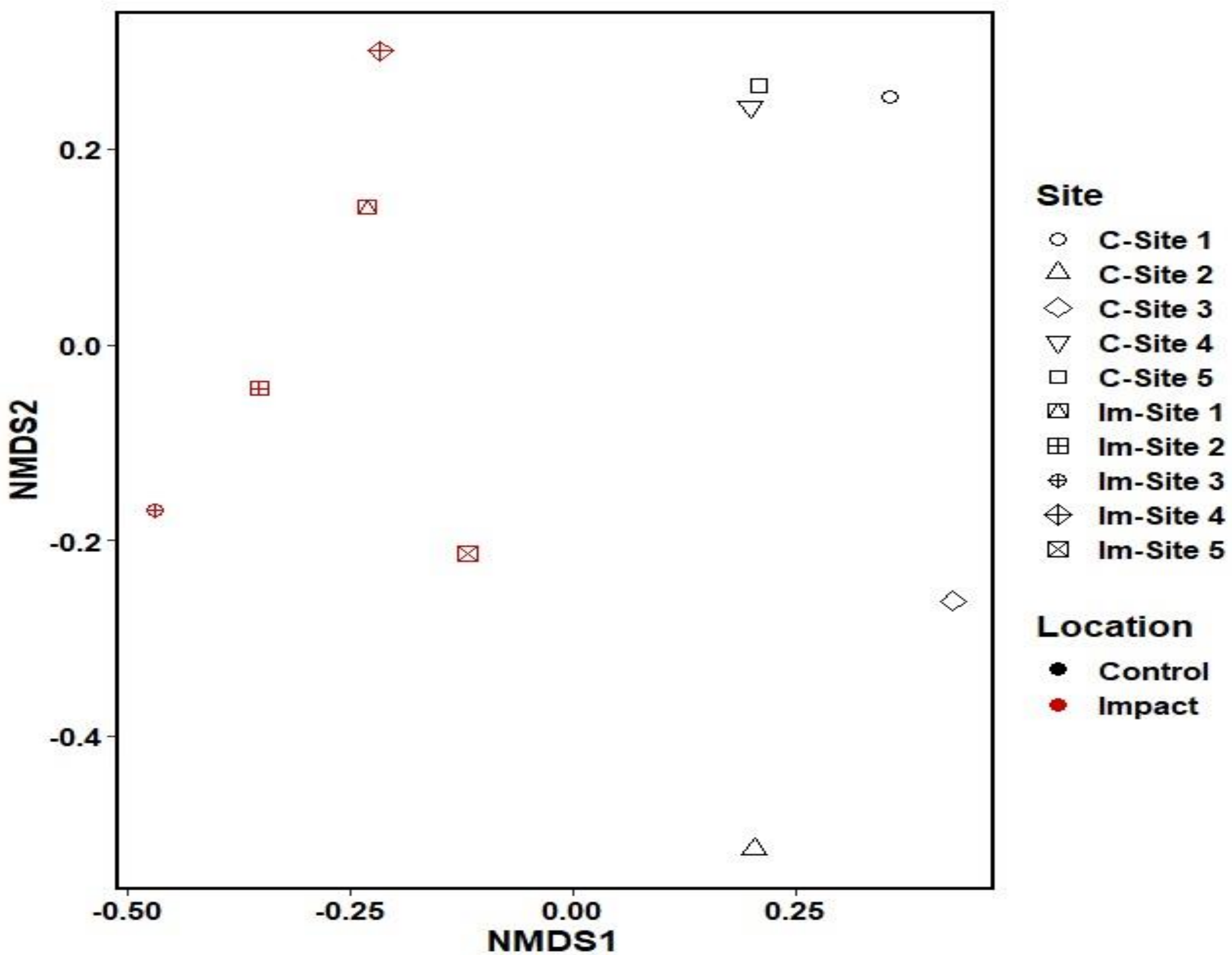

Figure 7. Ordination plot for NMDS examining differences in community structure between the impact reach (LRPA) and control. The ordination plot reveals a clear distinction between locations with impact and control sites segregating to separate sides (stress = 0.057; Anosim p-value = 0.007).

Chapter 3 – Extended Literature Review and Extended Methodology

Extended Literature Review

Introduction

Rivers are dynamic freshwater bodies which integrate the four dimensions of longitude (advective flow), laterality (floodplains), verticality (the hyporheic zone), and temporality (time) (Ward 1989). Along with groundwater flows, and stream coalescence, rivers, to use the metaphor of Stanford et al. (as presented in Hauer and Lamberti, 2017), provide the “plumbing of the continents,” as they flow to the oceans. Human civilization, from the apparent dawn of that enterprise through at least the pre-industrial age, has sought close proximity to rivers as a source of nutriment, power, and transport (Fang and Yawitz 2019). It is no surprise or accident that most major cities have a river at their heart. Fang and Yawitz (2019) describe a “coevolution” of human civilization and water-resources in the U.S., in which populations sought close residency to rivers through 1870. During the Industrial Age, new methods of land-borne transport (highways and railroads), along with the diverting of water from its source (canals and groundwater pumping), dense populations of humans were able to increase their distance from rivers (Fang and Yawitz 2019). However, by the mid 20th century, exponential population growth, complete industrialization of large riparian zones, and a willful ignorance of ecology, caused many major rivers, where they flowed through urban centers, to become de facto sewers (Burian et al. 2000). Citizen-led reform culminated in the Clean Water Act of 1972, under which many reclamation and restoration activities find their genesis (Hines 2013). Since that time, the understanding of river-health, as measured scientifically and/or aesthetically, in relation to human-health/well-being, has fostered the water resource management strategy of river restoration (Wohl et al. 2015).

River Restoration is a branch of applied water-resources science which seeks to restore a river, either for aesthetic/recreational and/or ecological purposes (Wohl et al. 2015). Currently in the U.S., greater than a billion dollars a year support the River Restoration Industry (Bernhardt 2005; Palmer and Allan 2006). Unfortunately, benchmarks for “success” are often arbitrary, ambiguous, or not at all (Palmer and Allan, 2006; Wohl 2005). Although, ecological success should be, at least, a goal of restoration efforts for more reasons than it is inherent in the term itself (Palmer et al. 2005). Palmer et al. (2005) provide five criteria for ecological success of river restoration efforts, which have been embraced internationally by experts in the water-resources field (Wohl et al. 2005). Perhaps the most important criterion is the fifth: “Some level of pre- and post-assessment is completed” (Palmer et al. 2005). Without pre- and post-assessment of the restoration reach, no ecological goals can be truly assessed, little can be learned, scientifically, from the endeavor (to guide future efforts), and the results cannot be said to be restorative or otherwise. Pre-/Post-assessments can range from simple qualitative sampling of biotic and/or abiotic targets to robust statistical analyses.

A common method of Pre-/Post-assessment employed in ecological studies is the Before-After-Control-Impact-Pairs design, or BACIP, of Stewart-Oaten et al. (1992). The BACIP is especially relevant to ecological studies where true replicates may not be feasible. The replicates of a BACIP, rather, involve replicates in time, where approximately simultaneous sampling events occur at a control and impact site, and the mean is evaluated. This is done pre- and post-restoration, and the means for both are compared (Stewart-Oaten et al. 1992). Beyond creating the “dynamic vision” of the completed restoration - Criterion 1. of Palmer et al. (2005) - the gathering of pre-restoration data is the first and essential piece of fieldwork required.

Pre-restoration assessments establish the baseline by which to guide the restoration project, revise goals, and, ultimately, evaluate the success or failure of the project. Pre-restoration assessments may focus on a single abiotic or biotic target, or they may focus on many (Morandi et al. 2015). The gold standard of pre-restoration efforts is that collected for the Kissimmee River Restoration Project (KRRP), Kissimmee River, Florida, in “Establishing a Baseline: Pre-Restoration Studies of the Channelized Kissimmee River.” This 487 page volume details the assessment of myriad abiotic and biotic targets including geomorphology, stream metabolics, and myriad taxa of plants and animals (Bousquin et al. 2005). 143 pages of the Volume are devoted to biotic communities. Aquatic macroinvertebrates take up nearly a quarter of this space.

Aquatic or Benthic Macroinvertebrates are the target group of many freshwater bioassessments for numerous reasons, including their ability to integrate ecosystem disturbances as a community within certain spatio-temporal ranges (Merritt et al. 2019, USEPA 1999). They are currently used in biotic indices by 49 states of the U.S.A., as well as across Europe (Merritt et al. 2019). Methods of macroinvertebrate bioassessment can be grouped into two major categories: Traditional/Morphological methods and Molecule/Genetic-based Methods. Traditional/Morphological methods are further divided into qualitative and quantitative methods (Barbour et al. 1999). Molecular/Genetic-based Methods revolve around the concept of “DNA barcoding” (Hebert et al. 2003). The upshot of either type of method is computation of data as metrics which are fitted into an index, or indices, to score the assessed system’s ecological integrity (Barbour et al. 1999). An outstanding example of the traditional/morphological method, is the qualitative Procedure 51 (EGLE 1990). Procedure 51 is the primary bioassessment tool used by The Michigan Department of Environment, Great Lakes, and Energy, or EGLE, to assess streams and rivers (EGLE 1990).

The Grand River Watershed is one of Michigan's largest watersheds (Hanshue and Harrington 2017), and flows through the state capital of Lansing, to one of Michigan's largest cities, Grand Rapids, to eventually empty into Lake Michigan (Hanshue and Harrington, 2017). The city of Grand Rapids takes its name from a series of extinguished rapids once active in the Grand River (Baxter, 1891). The building of dams for logging and industry, and the removal of natural substrates, spelled the end of the "rapids" (Hanshue and Harrington, 2017). To date, the downtown reach of the Grand River has suffered nearly every anthropogenic degradation possible - including its allocation as a municipal sewage dump (abiolch1@mlive.com 2010). Public outcry, the Clean Water Act of 1972, and the dedicated work of numerous agencies have allowed for rehabilitation of the Grand to its present state of recovery (abiolch1@mlive.com, 2010). Currently, a major restoration effort - Restore the Rapids - is now underway, spearheaded by Grand Rapids Whitewater, a not-for-profit organization (GRWW.org).

River Restoration: Standards and Methods of Evaluation

A river restoration project, or any ecological restoration project for that matter, is only as good as its evaluation; and the evaluation is only as good as the evaluation method. Palmer and Allan (2006) distill the problem into a statement of simple facts: no national standards exist to gauge river restoration success and, thus, no unified system for evaluation exists. This problem is made plain by Morandi et al. (2014) in which 44 river restoration projects were evaluated throughout France. Each of the 44 projects had an evaluation strategy, though these differed across projects; Morandi et al. separated the evaluation strategies into 4 classes, Class 1 being the simplest to Class 4 the most rigorous. Those projects using a simple, Class 1 strategy (<20 biophysical surveys focused on a few target groups over a >10 year period) reported the highest rate of success of all classes; whereas the most extensive and robust strategy, Class 4 (extensive monitoring period, well-devised framework, at least four biophysical target groups), often showed no effects, or negative ones (Morandi et al. 2014). The different evaluative Classes and their successes/failures raises a quandary: which of the four Classes from "poor" to "robust" represent the best method of evaluation? Do any represent the best method of evaluation? Assumedly the most robust, but if this is so, did they fail because they failed outright, or because they set their standards too high? Morandi et al. (2014) conclude that evaluation strategy must sync up with evaluation standards; i.e. the monitoring framework, metrics, and reference must be designed in such a way to support or not support a pre-set standard for success. Considering the above mentioned statement of Palmer and Allan (2006), one can clearly see the problems inherent with the lack of unified standards and evaluations. Which of the 44 examples in Morandi et al. were successes? And which were failures? By whose standards? By what criteria? Would those restoration projects of Class 1 be deemed failures by the standards of Class 4, and/or vice versa? Palmer and Allan recommend that federal agencies "adopt and abide by standards" for ecological success in river restoration (2006). To that end, Palmer, Allan, and many of the top practitioners and professionals in the field, present five standards for ecological success (2005): 1. Define a "dynamic ecological endpoint" - a "guiding image" for the project; 2. Improve the ecosystems of the river, and those interrelated; 3. Increase the resilience of the ecosystem/ its ability to self-sustain; 4. Ensure that no irreparable harm is done by the restoration process; and 5. Complete a pre-/post- assessment to evaluate the four previous standards and inform future projects. Standard 5 may be the most important step as it underlies each of the previous four standards by providing guidance, via pre-assessment, for 1 and supporting and/or failing to support the hypotheses of 2 through 4.

The Before-After-Control-Impact-Pairs Design

A Before-After-Control-Impact-Pairs design (BACIP) is a an evaluative method often applied to ecological studies in which the sampling of pre- (Before) and post- (After) conditions at a control and impact site are paired to provide a mean (Stewart-Oaten et al. 1986, 1992). The BACIP design was originally devised as a response to Hurlburt's problem of pseudoreplication (Hurlburt 1984; Stewart-Oaten et al. 1986). The BACIP avoids the problem of pseudoreplication by performing simultaneous sampling events in time at a control site and an impact site and taking the mean of both (Stewart-Oaten et al. 1986, 1992). Such simultaneous sampling events are taken pre- and post- condition of interest and the means are then compared/evaluated, via t-tests, to determine the degree of impact. Underwood (1992) advocates for modifications to the BACIP by adding multiple control sites, each paired simultaneously, during respective sampling events, to the single impacted site. Underwood also suggests ANOVA as a method for statistical analysis when sampling at both locations cannot be done simultaneously; though non-simultaneous sampling events must be randomly distributed in time (Underwood 1992).

Methods of Bioassessment using Benthic Macroinvertebrates

Macroinvertebrates are the focal group of many freshwater bioassessments for numerous reasons, including their small, collectible size which is yet large enough to allow Family level identification with the naked eye; poor dispersal rate; ubiquity of taxa diversity in nearly all freshwater habitats; wide range of disturbance sensitivity values across and within taxa; and their ability to integrate ecosystem disturbances as a community within certain spatio-temporal ranges (Merritt et al. 2019, Barbour et al. 1999). This last reason, especially, speaks to the importance of using macroinvertebrates to assess water body integrity as this community of organisms displays, within its metrics of richness and abundance, an accurate “moving picture” of ecological conditions (Rosenberg, as quoted in Mandaville, 2002) Methods of macroinvertebrate bioassessment can be grouped into two major categories: Traditional/Morphological methods and Molecule/Genetic-based Methods.

Traditional/Morphological methods of bioassessment

Traditional, Morphological Methods (referred to, for the remainder of this review, as TMMs) of macroinvertebrate bioassessment are myriad and vary, substantially, across Nations and States (Barbour et al. 1999, Birk et al. 2012). The legislated mandates above are simply drivers that allow for development of protocols by the respective governmental agencies. There are, however, necessary commonalities in the manner of approaches. The Environmental Protection Agency, for example, in its recommendations to various bureaucratic entities, sets forward specific Rapid Bioassessment Protocols (RBPs) and a framework to work within (Barbour et al. 1999). Nearly every state uses RBPs (singular or plural), and, although there is diversity of method between RBPs, all share the following aspects: 1. Sampling in the field involves D-nets, Kick nets, Surber Samplers, Hess Samplers, respectively or in combination; 2. All available habitats and microhabitats (substrates, undercut banks, etc.) are sampled for a given area i) with specific number of net jabs/sweeps, ii) for a set duration of time, iii) with the goal of gathering a specific number of organisms (e.g. ~300 for Procedure 51 (EGLE 2020); 3. Macroinvertebrates are identified in the field (usually to Family), and/or in the lab (Family-Genus-Species, if applicable); data are then computed as metrics which are fitted into an index, or indices, to score the assessed system's ecological integrity. Metrics vary, but most fit the following four categories: Taxonomic Richness, Composition (community and/or %individual taxon), Tolerance/ Intolerance, Functional Feeding Group. The synthesis of these metrics, via index, gives an accurate read of biotic conditions (Barbour et al. 1999).

Molecular methods of bioassessment

Molecular/Genetic-based Methods (referred to, for the remainder of this review, as MGbMs) of macroinvertebrate bioassessment revolve around the concept of "DNA barcoding" (Hebert et al. 2003). DNA barcoding sequences a genetic fragment of DNA (a barcode) from unknown or known organisms and compares it to others in DNA libraries (e.g., Barcode of Life, aka BOLD, and GenBank). Barcode sections for different taxa are those genetic regions which are highly conserved across phyla, class, etc. while still offering enough sequence diversity to discriminate conspecifics from each other with up to 99.9% accuracy (Hebert et al. 2003). Metabarcoding, when used as a method of bioassessment, is the DNA barcoding of entire samples. Sampling is performed via TMMs, as discussed above, or by taking bulk water samples to return to the lab. In the laboratory, a typical eDNA workflow consists of DNA concentration and isolation/purification; PCR amplification of a target region (Cytochrome Oxidase I for macroinvertebrates); the generation of unique nucleotide sequences which are pooled together to form a library; and high throughput sequencing. High throughput sequencing creates output files which are processed (edited to create clean sequence data called "reads") using a bioinformatics pipeline; reads are then clustered into molecular operational taxonomic units (MOTUs) based on sequence similarity of the targeted barcoding region; each MOTU represents a distinct taxonomic group (e.g. Family, Genus, Species, depending on the research question); MOTUS are compared to barcodes in DNA reference libraries such as BOLD and GenBank to identify the specific taxonomic group to which they align most closely (Deiner et al. 2017). Once identifications of taxa are made, the data is fitted to metrics to produce indices as above. Metabarcoding of bulk specimen samples is currently being used by the EPA (USEPA 2019) to conduct various bioassessments, though not as routine practice by individual states. Metabarcoding from bulk water samples, however, is the newest frontier of MGbMs and involves the use of Environmental DNA (eDNA). eDNA is defined as genetic material (extracellular castings, feces, gametes, tissues, fluids, etc.) left by an organism in the non-living sinks of its environment (Stewart 2019). As regards aquatic macroinvertebrates, this could be any genetic trace of an organism held within environmental continua, as the various inputs described by Vannote (The River Continuum Concept, 1980). eDNA metabarcoding for bioassessment, therefore, avoids the use of any TMM, field or laboratory. By eliminating TMMs from the workflow, eDNA metabarcoding could greatly reduce the cost and time of macroinvertebrate bioassessments (Deiner et al. 2016). In fact, a few studies have already suggested its supplanting, sometime in the future, of TMMs altogether (Machler et al. 2014).

"Restore the Rapids": a river restoration in progress

The Grand River basin is the second largest watershed of Michigan's Lower Peninsula (Hanshue and Harrington 2017). The Grand River itself is Michigan's longest river where it flows from Hillsdale County, across Southwest Michigan, to empty into Lake Michigan (Hanshue and Harrington 2017). The Grand River flows through Michigan's second largest city, Grand Rapids (Michigan Cities By Population). Grand Rapids was incorporated as a "village" in 1838 (Baxter 1891) and takes its name from a series of "rapids" that ran through the village center, for about a mile, with a purported fall of 18 feet (abiolch1@mlive.com, 2017). Beginning in 1866, dams were built to divert water for saw and grist mills, and natural substrates were removed as part of an excavation project for a proposed canal (Hanshue and Harrington, 2016). These operations effectively spelled the end of the "rapids" (Hanshue and Harrington, 2016). Since that time, the downtown reach of the Grand River has suffered nearly every anthropogenic degradation possible - including its allocation as a municipal sewage dump (abiolch1@mlive.com 2010). Water quality standards driven by public outcry, mid-century, and the enactment of the Clean Water Act of 1972, have slowly rehabilitated the Grand River to its present state of recovery (abiolch1@mlive.com, 2010). In 2011, the next logical step was taken, and the City's Master Plan was updated to include restoration of the "rapids" (Green Grand Rapids Report, 2012). A major restoration effort is now underway, spearheaded by Grand Rapids Whitewater, a not-for-profit organization (SEE https://grandrapidswhitewater.org). The Restore the Rapids project proposes to restore approximately 7 miles of the Grand River, where it flows through the city's center, and adjacent river banks (GRWW.org). The project comprises four key elements: 1. build a barrier in the upper reach to exclude invasive sea lamprey; 2. remove a dam in the upper reach to expose natural limestone substrates; 3. remove four low-head dams of the lower reach; 4. add large, rocky substrate to recreate natural riffle sequences (GRWW.org). Currently, the timeline for this project extends to 2026 which, if completed then, would represent a 15 year process from plan conception through completed construction (Experience Grand Rapids 2020). Post-restoration sampling of benthic macroinvertebrates typically begins 6 months post-restoration and continues, at regular intervals, for at least 2 years (Nuttle et al. 2017, MacCoy and Short 2017). Therefore, 2028 would be the earliest that Restore the Rapids could be evaluated on an ecological basis.

Extended Methodology

Subsampling

Great care should be exercised when subsampling as this dataset - the subsample - becomes the basis for analysis. Our subsampling methodology was suggested by Dr. F. Richard Hauer (personal communication, 2021) and similar to that recommended by Barbour et al. (1999), though simplified per equipment. First, EtOH was decanted from a Whirl Pak into a 250 micron (#60) Gilson sieve. Next, the Whirl Pak contents were poured into a large white pan and sorted for large and/or rare taxa; these taxa were placed in a specimen cup with 70% EtOH and labeled with the sample name, date, and ratio 1:1; the Whirl Pak was rinsed for remaining organisms into the sieve; finally, the remaining contents were poured into the sieve and rinsed. The sieve was then floated (within a stoppered lab sink) in ~2 inches of water, swirled to homogenize the sample, and lifted with care to keep level. Using a Plexiglass splitter (personal construction from two notched pieces of Plexiglass), the sample was quartered. Numbers 1-4 were assigned to each quarter and a virtual random generator was used to select a quarter. The random quarters were then isolated with a metal cookie cutter (personal construction from sheet metal) and transferred, with a chemistry spatula, to a large white pan for sorting.

Sub-subsampling

Occasionally, a dominant taxon made it necessary to sub-subsample. In this case, a gridded pan was used. All taxa excepting the dominant taxa were picked from the subsample. Next, the subsample was poured into the gridded pan with sixteen grids; the contents were stirred and allowed to settle. Using a virtual random number generator, four of the grids were selected.

The taxon was counted in the four selected grids and multiplied (x4) to approximate the total number of the dominant taxon in the sample.

Sorting

A Realspace© magnifier task-lamp with 22-watt bulb and 1.75x magnification was used during the sorting process. Four specimen cups were labeled with the sample name, date, and ratio 4:1. The four specimen cups were further labeled by group as EPT, Dipterans/worm-like, Beetles/True bugs, and Other; the cups were filled with 70% EtOH. Taxa were rough-sorted to the appropriate specimen cup for later identification to lower taxonomic level.

Chironomid subfamily sorting/I.D.

Much is made of the difficulty in IDing chironomids. Often, a noxious mounting chemical, such as CMCP-10, is necessary. We did use CMCP-10 to initially diagnose chironomids, and this is definitely necessitated when diagnosing to genus or species. However, subfamily can, with practice, be diagnosed using a good dissecting scope. We had the benefit of a Nikon DS-L3 with 135x magnification. We found that, under this level of magnification, subfamilies could be readily discriminated. For example, if no ligula is present, Tanypodinae is eliminated as a diagnosis. If premandibles are present, Podonominae is eliminated. If ventromental plates with well-developed striations are observed, the specimen belongs to subfamily Chironominae. If ventromental plates with well-developed striations are lacking, specimens most likely belong to Orthocladiinae (Note: the common Grand River specimen - probably *Orthocladius* sp. - has a triangular mentum with an odd number of teeth).

Appendix

Table 1A. Complete prerestoration assessment dataset, with metrics and indices, for the impact reach in downtown Grand Rapids, MI, USA.

	Impact Reach/Location						Trait 0	Trait 1
Taxa	Dwntwn1	Dwntwn2	Dwntwn3	Dwntwn4	Dwntwn5	Row Totals	Feeding group	Voltinism
Ancylidae	0	0	0	0	0	0	scr	na
Baetidae	8	0	0	4	1	13	cg	3
Brachycentrus	4	0	0	0	0	4	cf	2
Caecidotea	0	22	0	69	0	91	cg	na
Cambaridae	1	1	0	1	0	3	cg	na
Ceraclea	0	4	0	4	4	12	cg	2
Cheumatopsyche	18	49	5	57	8	137	cf	2
Chironomidae	97	71	77	215	27	487	cg	2
Coenagrioninae	0	0	4	0	0	4	prd	
Corbicula	0	0	0	0	0	0	cf	na
C. fraternus	0	0	0	0	4	4	cf	2
Dromogomphus	0	0	0	0	0	0	prd	1
Dubiraphia larva	4	0	0	0	0	4	cg	1
Empididae	0	0	0	4	0	4	prd	2
Ephoron	3	2	5	0	0	10	cg	2
G. fasciatus	14	19	1	6	9	49	shr	na
Helicopsyche	0	0	0	0	0	0	scr	2
Hydracarina	1	0	0	0	0	1	prd	na
Hydropsyche	19	15	0	29	1	64	cf	2
Hydroptilidae	0	0	0	0	4	4	scr	2
Macronychus adult	0	0	1	0	1	2	cg	1
Macronychus larvae	0	0	0	1	4	5	cg	1
Nectopsyche	0	0	4	0	0	4	shr	2
Oecetis	0	4	0	0	0	4	prd	2

Oligochaeta	8	8	4	8	5	33	cg	na
Optioservus adult	0	0	0	0	0	0	cg	1
Optioservus larvae	0	0	0	0	0	0	scr	3
Petrophila	5	0	0	0	0	5	scr	2
Pleuroceridae	12	1	0	0	0	13	scr	na
Polycentropus	0	0	0	0	4	4	prd	2
Potamyia flava	0	4	0	0	0	4	cf	2
Psephenus	0	0	0	0	0	0	scr	1
Setodes	0	0	0	0	0	0	cg	2
Simuliidae	108	52	4	100	5	269	cf	3
Sphaeriidae	0	0	0	0	0	0	cf	na
Stenacron	0	7	1	9	4	21	cg	2
Stenelmis adult	0	0	0	8	0	8	cg	1
Stenelmis larvae	4	1	0	79	0	84	cg	1
Stenonema	17	10	0	12	30	69	scr	2
Tabanidae	0	0	0	0	0	0	prd	2
Tricorythodes	184	51	42	180	61	518	cg	2
Unknown Dipteran	0	0	0	4	0	4		2
Unknown Gastro	0	0	0	0	0	0		na
Unknown Trichop	0	0	0	0	0	0		2
~Limnephilid	0	0	0	0	0	0		2
Unknown Plecop	0	0	0	0	0	0		2
Neureclipsis	0	0	0	0	0	0	cf	3
						Total Dwntwn organisms		
Total individuals:	507	321	148	790	172	1938	387.6	
#Diptera	205	123	81	319	32			
%Diptera	40	38	55	40	19			
#EPT:	253	146	57	291	121	868		
%EPT:	50	45	39	37	70	44		
#Hydropsychinae/Trichoptera	37	64	8	86	9	204		
%Hydropsychinae of Trichoptera	90	84	67	96	36			

Shannon Diversity (H)	1.9	2.23	1.42	2.08	2.08			
Evenness	0.672	0.789	0.591	0.707	0.75			
Taxa Richness	17	17	11	17	15			
					Total H =	2.25		
					Total Evenness =	0.643		
					Richness =	30		

Chironomidae Subfamilies								
Taxa	Dwntwn1	Dwntwn2	Dwntwn3	Dwntwn4	Dwntwn5	Row Totals	Feeding group	Voltinism
Chironominae	25	16	53	40	29	163	cg	3
Orthocladiinae	44	37	16	116	5	218	cg	2
Tanypodinae	8	8	0	35	1	52	prd	2
FFGs						Totals		
Shredders	14	19	5	6	9	53		
Scrapers	34	11	0	12	34	91		
Collector-filterers	149	120	9	186	18	482		
Collector-gatherers	309	167	130	578	107	1291		
Predators	1	4	4	4	4	17		
P22 metrics			**Values**					
1. FFG diversity	1.38	1.51	0.72	1	1.58			
2. FFG surrogate	0.57	0.7	0.07	0.34	0.45			
3. % Trichoptera	0.08	0.24	0.06	0.11	0.15			
4. EPT taxa richness	7	9	5	7	10			
5. Total t.r.	17	17	11	16	15			
6. Diptera t.r.	2	2	2	3	2			
7. Plecoptera t.r.	0	0	0	0	0			
8. % Dominance	0.36	0.22	0.52	0.27	0.35			
X. % Trichop w/adj	0.008	0.04	0.03	0.005	0.09			
P22 metrics			**Scores**					
1. FFG diversity	8	16	0	8	16			
2. FFG surrogate	8	8	0	8	8			
3. % Trichoptera	20	20	14	20	20			

4. EPT taxa richness	6	6	3	6	8		
5. Total t.r.	2	2	0	2	2		
6. Diptera t.r.	2	2	2	2	2		
7. Plecoptera t.r.	0	0	0	0	0		
8. % Dominance	4	5	2	5	4	mean	s.d.
Totals	50	59	21	51	60	48.2	
1. FFG diversity	8	16	0	8	16		
2. FFG surrogate	8	8	0	8	8		
X. % Trichop w/adj	0	14	7	0	20		
4. EPT taxa richness	6	6	3	6	8		
5. Total t.r.	2	2	0	2	2		
6. Diptera t.r.	2	2	2	2	2		
7. Plecoptera t.r.	0	0	0	0	0		
8. % Dominance	4	5	2	5	4	mean	s.d.
Totals	30	53	14	31	60	37.6	18.68957

Table 2A. Complete prerestoration assessment dataset, with metrics and indices, for the control reach in Sunfield, MI, USA.

	Control Reach/Location						Trait 0	Trait 1
Taxa	Sunfield1	Sunfield2	Sunfield3	Sunfield4	Sunfield5	Row Totals	Feeding group	Voltinism
Ancylidae	4	0	0	4	4	12	scr	na
Baetidae	28	0	0	84	16	128	cg	3
Brachycentrus	30	5	0	56	13	104	cf	2
Caecidotea	0	0	0	0	0	0	cg	na
Cambaridae	1	0	0	0	0	1	cg	na
Ceraclea	0	0	0	0	0	0	cg	2
Cheumatopsyche	21	1	0	0	4	26	cf	2
Chironomidae	8	5	5	32	13	63	cg	2
Coenagrioninae	0	0	0	0	0	0	prd	
Corbicula	4	0	1	2	17	24	cf	na
C. fraternus	0	0	0	0	0	0	cf	2
Dromogomphus	0	5	0	0	0	5	prd	1
Dubiraphia larva	0	4	4	0	0	8	cg	1
Empididae	0	0	0	0	0	0	prd	2
Ephoron	1	3	6	4	26	40	cg	2
G. fasciatus	186	27	101	177	125	616	shr	na
Helicopsyche	0	0	0	0	4	4	scr	2
Hydracarina	0	0	0	0	0	0	prd	na
Hydropsyche	0	0	0	9	0	9	cf	2
Hydroptilidae	0	0	0	0	0	0	scr	2
Macronychus adult	4	0	4	5	0	13	cg	1
Macronychus larvae	4	0	0	0	0	4	cg	1
Nectopsyche	0	0	0	0	0	0	shr	2
Oecetis	4	0	0	0	0	4	prd	2
Oligochaeta	0	8	0	0	0	8	cg	na
Optioservus adult	0	0	0	4	0	4	cg	1
Optioservus larvae	4	0	0	0	0	4	scr	1

Petrophila	0	0	0	0	0	0	scr	2
Pleuroceridae	0	0	0	0	4	4	scr	na
Polycentropus	0	0	0	0	0	0	prd	2
Potamyia flava	0	0	0	0	0	0	cf	2
Psephenus	6	0	0	0	4	10	scr	1
Setodes	0	0	0	4	0	4	cg	2
Simuliidae	0	0	0	0	0	0	cf	3
Sphaeriidae	0	0	0	13	4	17	cf	na
Stenacron	11	0	0	0	0	11	cg	2
Stenelmis adult	177	0	8	88	58	331	cg	1
Stenelmis larvae	141	4	0	12	45	202	cg	1
Stenonema	78	13	23	30	37	181	scr	2
Tabanidae	0	4	0	0	0	4	prd	2
Tricorythodes	80	4	4	180	322	590	cg	2
Unknown Dipteran	0	0	0	0	0	0		2
Unknown Gastro	0	0	0	4	0	4		na
Unknown Trichop	0	0	0	0	0	0		2
~Limnephilid	0	0	0	0	0	0		2
Unknown Plecop	4	0	0	0	0	4		2
Neureclipsis	0	4	0	0	0	4	cf	3
						Total Sunfield organisms		
Total individuals:	796	87	156	708	696	2443	488.6	
#Diptera	8	9	5	32	13			
%Diptera	1	10	3	5	2			
#EPT:	257	30	27	367	422	1103		
%EPT:	32	34	17	52	61	45		
#Hydropsychinae/Trichoptera	21	1	0	9	4	35		
%Hydropsychinae of Trichoptera	38	10	0	13	24			
Shannon Diversity (H)	2.12	2.23	1.27	2.08	1.83			
Evenness	0.708	0.871	0.576	0.733	0.661			
Taxa Richness	18	13	9	16	15			

					Total H =	2.23		
					Total Evenness =	0.643		
					Richness =	29		
Chironomidae Subfamilies								
Taxa	Sunfield1	Sunfield2	Sunfield3	Sunfield4	Sunfield5	Row Totals	Feeding group	Voltinism
Chironominae	8	4	4	28	9	53	cg	3
Orthocladiinae	0	0	0	4	0	4	cg	2
Tanypodinae	0	1	1	0	0	2	prd	2
FFGs						Totals		
Shredders	186	27	101	177	125	616		
Scrapers	92	13	23	34	53	215		
Collector-filterers	55	10	1	80	38	184		
Collector-gatherers	455	28	31	413	480	1407		
Predators	4	9	0	0	0	13		
P22 metrics								
1. FFG diversity	1.62	2.16	1.32	1.52	1.33			
2. FFG surrogate	0.23	0.42	0.18	0.19	0.15			
3. % Trichoptera	0.07	0.11	0	0.1	0.03			
4. EPT taxa richness	8	6	3	7	7			
5. Total t.r.	17	13	9	15	15			
6. Diptera t.r.	1	2	1	1	1			
7. Plecoptera t.r.	0	0	0	0	0			
8. % Dominance	0.4	0.31	0.65	0.23	0.46			
X. % Trichop w/adj	0.04	0.1	0	0.08	0.02			
P22 metrics			**Scores**					
1. FFG diversity	16	25	8	8	8			
2. FFG surrogate	8	8	8	8	8			
3. % Trichoptera	20	20	0	20	7			
4. EPT taxa richness	6	3	0	6	6			
5. Total t.r.	2	0	0	2	2			
6. Diptera t.r.	0	2	0	0	0			

7. Plecoptera t.r.	0	0	0	0	0		
8. % Dominance	4	5	0	5	4	mean	s.d.
Totals	56	63	16	49	35	43.8	
1. FFG diversity	16	25	8	8	8		
2. FFG surrogate	8	8	8	8	8		
X. % Trichop w/adj	14	20	0	20	7		
4. EPT taxa richness	6	3	0	6	6		
5. Total t.r.	2	0	0	2	2		
6. Diptera t.r.	0	2	0	0	0		
7. Plecoptera t.r.	0	0	0	0	0		
8. % Dominance	4	5	0	5	4	mean	s.d.
Totals	50	63	16	49	35	42.6	17.86897

Table 3A. Land-cover-type proportions for the impact reach in downtown Grand Rapids, MI, USA. Each of 5 zones is described with three different lateral distances (widths from the wetted bank).

Description	Lateral distance (m)	Impervious	Forest	Open
2 - 5 km upstream	100	22.71359957	59.61703563	17.6693648
	200	44.03605463	36.59476759	19.36917778
	300	53.1564457	35.01899361	11.82456069
1 - 2 km upstream	100	78.39068476	13.43541102	8.173904218
	200	88.58437816	4.989038599	6.426583245
	300	82.67022565	6.82533798	10.50443637
0 - 1 km upstream	100	80.59191623	10.92066735	8.487416421
	200	88.02777284	2.46343341	9.508793747
	300	81.46072714	9.825040328	8.714232535
Sampling area	100	78.61594953	13.31136428	8.072686188
	200	92.66594321	3.835719354	3.498337438
	300	90.1046463	5.886005302	4.009348402
1000 m downstream	100	81.09837768	11.50270386	7.398918454
	200	90.15912756	4.267164559	5.573707884
	300	89.13568039	3.884007995	6.980311617

Table 4A. Land-cover-type proportions for the control reach in Sunfield, MI, USA. Each of 5 zones is described with three different lateral distances (widths from the wetted bank).

Description	Lateral distance (m)	Impervious	Forest	Open	Agriculture
2 - 5 km upstream	100	0.786703388	80.98899306	5.139098141	13.08520541
	200	1.927073939	78.99930325	1.393255831	17.68036698
	300	2.449542867	77.0917699	2.69531697	17.76337026
1 - 2 km upstream	100	0.826083518	80.39228092	14.20001582	4.581619741
	200	3.393751544	65.80681146	4.195209966	26.60422703
	300	3.800272194	62.44550741	2.71702565	31.03719474
0 - 1 km upstream	100	0.254548666	71.02637247	14.89858367	13.8204952
	200	0.236979229	76.00371172	2.561088772	21.19822028
	300	0.145259676	83.70098099	3.906307509	12.24745182
Sampling area	100	0.276674086	87.2792328	12.44409311	0
	200	0	95.3163817	4.683618296	0
	300	0	79.78205594	20.21794406	0
1000 m downstream	100	0.003230299	88.64801229	11.34875741	0
	200	0.006421488	90.48197855	9.51159996	0
	300	0.079202952	86.45276642	13.46803063	0

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