AN ECOLOGICAL EVALUATION OF AN URBAN STREAM RESTORATION IN WEST CHICAGO, ILLINOIS

BY

AUSTIN TYLER RUNDUS

THESIS

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Adviser:

Adjunct Assistant Professor Jeffrey A. Stein

ABSTRACT

The consistent rise in urban population and expansion of urban centers in the United States over the last several decades has led to the need for preservation of natural resources in those areas, as well as degradation to those resources. Streams in urban systems are often highly degraded and may require restoration to mitigate negative effects of urbanization and restore ecosystem function. In this study, I analyzed the physical habitat, water quality, macroinvertebrate community, and fish community of a 13 km stream restoration on the West Branch of the DuPage in the suburban Chicagoland area, using the similar, unrestored East Branch of the DuPage as a reference. The restored West Branch had higher quality instream habitat than the East Branch, especially in regards to substrate, channel morphology, and pool and riffle quality. Water quality did not vary between the streams except for flow, which was higher on the West Branch. The macroinvertebrate community on the restored West Branch was more diverse, and included more sensitive species, and scored better on macroinvertebrate community metrics designed to indicate water quality. The fish communities did not differ between the streams; however, Smallmouth Bass (*Micropterus dolomieu*) were found in significantly higher numbers on the West Branch. My study indicates that there were some positive effects of the restoration, but also that pre- and post-restoration data together would allow for deeper insights into the effects of urban stream restoration.

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CHAPTER 1: GENERAL INTRODUCTION

Watershed Urbanization

Between the years 2000 and 2010, the United States population grew by almost 20% in 486 designated urban areas, with these urban areas averaging 44.25 square kilometers in growth (US Census Bureau 2010). Overall, population in the United States grew by over 27 million people during this period, with 98.4% of this growth occurring in urban areas. In contrast, human populations in rural areas have grown by less than half a million people in that same timeframe, demonstrating a clear shift in population growth to urban centers, despite modest migration out of urban centers (Schachter et al 2003). Growing human populations in urban centers and the resultant development and alteration of the landscape has been ecologically destructive (Zhao et al. 2011), negatively affecting the ecology of aquatic systems and their surrounding landscapes, leading to disturbances in ecosystem services. Urban population growth imparts an increased demand for ecosystem services such as drainage and flood control, noise reduction, recreation, fishing, and nature education (Bolund and Hunhammar 1999). This demand can be seen in the use of natural resources in urban areas for recreation in general, and for sport fishing in particular with sixty-eight percent of anglers in the U.S. residing in urban areas as of 2011 (USFWS 2011). This high percentage of anglers and overall resource users within urban watersheds requires that these areas be usable for recreation, as well as ecologically functional for both overall stream health and to support sustainable sport fisheries.

Rapid urbanization has been linked to negative abiotic and biotic impacts to stream ecosystems. Urban watersheds suffer negative impacts to hydrology, instream and riparian habitats, water chemistry, and ultimately biodiversity, and often require restoration to mitigate these effects (Paul and Meyer 2001; Bernhardt et al. 2005; Walsh et al. 2005). Understanding the negative effects of urbanization on rivers and streams has become increasingly important to meet the needs of local resource managers tasked with providing recreational opportunities based on sustainable natural resources, such as angling, in densely populated areas, as well as to slow or reverse the negative ecological and biological impacts of urbanization on the systems.

Physiochemical Effects of Urbanization

At a landscape scale, urbanization and the resultant hydrological changes in a watershed lead to physical alterations of instream habitats and chemical impairments in water quality (Walsh et al. 2005). Urbanization brings with it substantial changes in land cover that include shifts towards more impervious surfaces, which are considered one of the main driving forces behind hydrological shifts in urban stream systems (Finkenbine et al. 2000; Walsh et al. 2005). The construction of impervious surfaces increases surface run-off causing rapid, intense flooding events that quickly introduce sediments into the stream as exposed soils erode (Paul and Meyer 2001; Walsh et al. 2005). The combination of increased impervious surfaces and the advanced stormwater remediation infrastructure creating efficient drainage into streams creates flashier stream flow, with more rapidly ascending and descending hydrographs (Walsh et al. 2005). These changes in hydrology in turn cause shifts in the physical characteristics of riparian and instream habitats and structures (Paul and Meyer 2001; Walsh et al. 2005), disrupting natural filtration processes of undisturbed landscapes and increasing the amount and types of chemicals introduced into streams (Walsh et al. 2005). Increased imperviousness of the watershed also impedes natural infiltration of water into the soil, increasing runoff directly into the stream, which in turn shortens lag time to peak flows after precipitation events and causes higher peak flows (Paul and Meyer 2001). Lower base flows are also associated with higher impervious surface and urbanization in many stream systems, due to reduction in natural recharge of the stream via groundwater drainage, with some urban streams even becoming intermittent in the summer months (Finkenbine et al 2000).

Changes in hydrological regime, particularly higher variability and intensity of flows in the urban environment, have diverse effects on the physical and morphological characteristics of the stream environment. Urban streams with altered hydrology experience increases in pool depth, channel width, and increase in rates of erosion and scour, resulting in the homogenization of instream habitat (Walsh et al. 2005). Stream channel width and pool depth initially decrease due to the addition of sediments via erosion. Over time, frequent and intense flow events in urban watersheds (e.g., flash floods) eventually scour introduced erosional deposits resulting in wider and deeper channels. The influx of additional sediment also alters the channel shape, with the increased movement and scour of sediments often converting natural meanders into straight channels (Paul and Meyer 2001). Additionally, the increased habitat homogeneity can impact fish and macroinvertebrate communities through the reduction of specific habitats needed by a variety of fish, thereby reducing diversity (Gorman and Karr 1978). All of these physical changes to the stream can have a cascading effect leading to changes in fish and macroinvertebrate communities, ultimately impacting ecosystem services.

The changes in hydrology within urbanized stream systems, along with additional point sources of pollution, multiple land uses, and a lack of permeable surfaces for proper drainage increase the delivery of pollutants and nutrients into urban streams (Wang et al. 2001). Storm water drainage systems often flow directly into streams, eliminating the natural filtration provided by wetlands and other pervious habitats, and thus delivering additional pollutants directly into the system (Walsh et al. 2005). Hatt et al. (2004) found that urban areas tend to have efficient drainage systems, which more directly connect rainwater runoff and avoid natural filtration and groundwater recharge processes, leading to higher concentrations of dissolved organic carbon, total phosphorus, and ammonium. Urban streams also tend to have higher concentrations of nitrogen, and lower pH values than rural streams not impacted by human population centers (Hatt et al. 2004). Stream water temperature has also been found to be higher in the summer in urban watersheds and lower in winter months when compared to non-urban streams, with higher diel variations as well (Paul and Meyer 2001; Walsh et al. 2005). Instream temperature regimes in urban systems are impacted by higher temperatures of urban wastewater runoff during the summer, removal of riparian vegetation that provide shade from solar radiation, and decreased groundwater recharge (Kinouchi et al. 2007). Additionally, urban areas have increased need for dealing with runoff and storm water, which tend to have higher contaminant rates than in non-urban areas due to a higher density population, and the use of more chemicals such as road salt on the higher percentage of impervious surfaces (Kelly et al. 2012). These physiochemical impacts of urbanization in turn impact the biotic community within and around the stream environment.

These widespread and interconnected impacts of urbanization on the physiochemical characteristics of urban stream systems has far reaching impacts on ecosystem function as well as the ecosystem services available to humans in densely populated urban areas. For instance, stream temperature is an important variable in invertebrate life history and leaf decomposition (Paul and Meyer 2001), and many fish and macroinvertebrate species are sensitive to changes in water chemistry including pH, nitrates, and others (Paul and Meyer 2001; Walsh et al. 2005). Impacted water quality and physical changes to the stream environment can reduce abundance and diversity of macroinvertebrates and fish (Walsh et al. 2005), which in turn can limit the ecosystem services available to humans wishing to enjoy the fauna of local lotic systems.

Ecological Impacts of Urbanization

Shifts in macroinvertebrate communities are one of the most noticeable and dramatic effects of urbanization on streams, and can be tied to multiple abiotic changes caused by urbanization, including hydrology, riparian and physical stream habitat changes, and changes in stream chemistry, such as increased nutrient loads, altered temperature regimes, and increased toxins (Paul and Meyer 2001). Sensitive macroinvertebrate species in urban watersheds are typically completely absent or occur in very low abundance where fewer numbers of tolerant, generalist species comprise the macroinvertebrate community (Walsh et al. 2005), resulting in lower biodiversity and eliminating ecologically important taxa. Even though the changes to macroinvertebrate assemblages due to urbanization is one of the most widely and commonly studied effects of urbanization on streams, research is mostly based on gradients of urbanization through multiple watersheds, and studies assess the impacts over large areas, rather than within a single stream (Paul and Meyer 2001). Changes in macroinvertebrate community composition are the result of a slew of interconnected changes to urban watersheds that impact both abiotic and biotic characteristics of a stream, and these changes to the macroinvertebrate community in turn impact species richness, diversity, and abundance with the fish community. Physical and chemical changes impacting stream fish communities include higher sediment content and chemical input into urban streams (Paul and Meyer 2001), as well as reduction of instream habitat, greater channelization, and changes to flow regimes (Schwartz and Herricks 2007). Abiotic changes to the urban stream not only have direct negative impacts on fish species richness and diversity, but also have indirect negative effects through negative changes to the macroinvertebrate community, which are a vital part of the diet of many fish species (Taylor and Roff 1986). Within urban watersheds, sensitive fish species tend to be reduced in abundance and species number (Walsh et al. 2005) and overall fish abundance and species diversity tend to decline (Paul and Meyer 2001) resulting in an increase in relative abundance of many tolerant fish species (Paul and Meyer 2001; Walsh et al. 2005). Declines in the proportions of ecologically sensitive breeding guilds such as lithophilic spawners (Balon 1975), and declines of biotic integrity such as IBI values have been linked to increased urbanization (Paul and Meyer 2001; Helms et al. 2005). Fish communities in urban or urbanizing streams also have higher percentages of fish with deformities, erosion, lesions, and tumors (DELTs) than non-urban streams, suggesting that individual fish health is also decreased by urbanization (Helms et al. 2005). These impacts, along with abiotic changes due to urbanization and the need for management of urban streams and fisheries require remediation of the urban stream and surrounding environment. This is especially important in the context of shifting population demographics and the concomitant changes in land use across the U.S. Changes to stream fish communities can have major impacts on top predators (Paul and Meyer 2001) because piscivores rely on a diverse forage base. Cascading and accumulating effects of degraded instream habitat that top predators utilize for foraging, reproduction, and as a refuge from predation at juvenile life stages can result in low population sizes of sport fish targeted by anglers. Lost fishing quality in highly populated areas can be detrimental to the quality of ecosystem services available to the urban population.

Impairment of Ecosystem Services

Impairment of urban stream hydrology, chemistry, physical habitat, and biotic communities, identified as the Urban Stream Syndrome (Walsh et al. 2005), has diverse, mainly negative impacts on the ecosystem services available to human populations living in these urban watersheds. Negative impacts on the biotic community through the reduction of habitat for macroinvertebrate and fish species negatively affect ecosystem services such as water quality, flood control, and outdoor recreation activities such as recreational fishing (Bolund and Hunhammar 1999). The economic, social, and cultural importance of these ecosystem services can be very high, and the loss of them is an additional negative impact of urbanization on urban watersheds.

Densely populated urban areas result in significant demand for access to natural areas that, by virtue of extensive development, are in limited supply in the urban landscape. Because space available for development of new natural areas is limited, appropriate utilization and maintenance of existing natural areas, and the restoration of degraded areas that are unsuitable for development is necessary. Recreational use of streams and their surroundings can include canoeing, swimming, camping, hiking, and nature education, which can be among the most highly valued ecosystem services provided by urban stream environments, bringing social, cultural, and psychological values to urban areas (Bolund and Hunhammar 1999). These activities can also have value to citizens, bringing people and economic activity to natural areas for recreation (Bischel et al. 2013). Recreational ecosystem services lost to urbanization can have far reaching impacts on communities, including the physical and psychological benefits felt by those who have access to natural areas, especially in an urban or suburban setting (Lopez-Mosquera and Sanchez 2012).

Drastic changes in the hydrology of urban streams negatively affect the ecosystem services that specifically rely on flow and good water quality. Flood control, natural groundwater recharge, and rainwater drainage are all negatively affected by the increased impervious surface area and man-made drainages of urban areas (Bolund and Hunhammar 1999; Walsh et al. 2005). The decreased ability of streams in urban areas to control flooding and drainage causes problems for populated and developed areas where flood damage and lost economic activity during flood events can be costly to repair and difficult to mitigate. Groundwater recharge is important for sustainable subsurface sources of drinking water for human consumption, and a lack of it can also cause problems for cities, including loss of potable and non-potable water (Bischel et al. 2013). Poor water quality in urban streams can also damage ecosystem services available by creating health risks for people using the streams (Bischel et al. 2013).

Urban Stream Syndrome and its associated changes to biotic communities have made it difficult for sport fish, which are typically top predators in the aquatic ecosystem, to thrive in viable numbers in many urban streams (Walsh et al. 2005). Recreational fishing is one of the more financially important ecosystem services given the economic activity generated by businesses that sell bait and equipment related to fishing. Fishing license and gear sales, as well as local taxes associated with fishing activities provide funding for natural resource management and conservation programs (Balsman and Shoup 2008). However, over the past decade, the proportion of Americans living in urban areas who participated in angling decreased (USFWS 2011). These factors, along with rapid urban growth has created the need for natural resource agencies to reverse declines in fishing license sales and participation (Balsman and Shoup 2008) by providing quality angling opportunities near urban centers. The restoration of ecological function and ecosystem services in the urban watersheds has become an important goal of resource managers who, therefore, look to stream restoration as a way to achieve improvements to impaired urban systems.

Urban Stream Restoration

In response to the growing problem of urbanization as a leading cause of stream impairment and degradation (Karr 1981; Paul and Meyer 2001; Gleick 2003), restoration of degraded urban streams is becoming a widely-used method for restoring ecological function and stream ecosystem health in urban landscapes (Bernhardt et al. 2005; Palmer et al. 2010; Bernhardt and Palmer 2011; Jahnig et al. 2011; Doyle and Shields 2012; Haase et al. 2013). Given the wide array of causes of stream degradation in urban landscapes, restoration projects have sought to improve water quality, rehabilitate instream habitat, provide fish passage, establish effective riparian zone management, stabilize erosional stream banks, and provide flood control to urban communities (Bernhardt et al. 2005) with the overall goal of increasing habitat quality and heterogeneity (Palmer et al. 2010; Haase et al. 2013). Other restoration methods include debris and trash removal, reshaping of banks and channels, native vegetation addition, invasive vegetation removal, flow modifications, dam removal, and native fish and macroinvertebrate species reintroduction (Bernhardt et al. 2005). While some river restorations incorporate long segments of river, most restorations are smaller in scale, being executed over less than 1 km of stream length.

With urbanization induced changes to hydrology being the most consistent impact on urban stream systems (Bernhardt and Palmer 2007), techniques that attempt to mitigate the changes to the hydrology of an urban stream are commonplace (Haase et al. 2013). Strategies designed to improve hydrology and hydromorphology have been shown to be effective techniques for physical restoration of streams (Jahnig et al. 2011; Bernhardt and Palmer 2011; Haase et al. 2013). Whether those physical improvements to urban streams result in benefits to ecological function, however, is less evident. In an analysis of 24 hydromorphological urban stream restorations, only four showed significant improvements in fish abundance and species richness metrics with similarly marginal improvements to macroinvertebrate diversity and abundance (Haase et al. 2013). These technique include dam removal, storm water runoff remediation, floodplain modification, reconnection of backwaters, creation of multiple channels, installation of flow deflectors, and the creation of wetlands to slow runoff flow into the stream

(Bernhardt and Palmer 2007; Haase et al. 2013). While hydrological restoration is very commonly used in urban restorations, there are other restoration methods that are often used in urban environments, due to the unique and added challenges of the urban environment, including higher impervious surface area and often more severely impacted stream reaches.

Restoration of instream habitat is one of the common restoration strategies used in urban streams. Instream habitat restoration can involve small-scale habitat improvements at local sites, or can involve broad improvements across large spatial scales. Bernhardt and Palmer (2007) found that in a study of urban stream restorations in Illinois and Washington, the main goal of instream restoration was to promote channel stability and to reduce erosion. However, restorations of instream habitats can also include the introduction of cover and microhabitats for fish and macroinvertebrate species, restoring meandering banks to naturalize flows, addition of pool-riffle sequences, and substrate enhancements (Bernhardt et al. 2005). In several post-restoration assessments, such instream enhancements have been shown to have a slightly positive effect on the macroinvertebrate community, though they have not shown many indications of positive effects on the fish community (Bernhardt and Palmer 2007). Physical restorations of streams are often coupled with other types of restorations in order to achieve stated restoration goals (Bernhardt and Palmer 2007).

Along with instream and hydrological restorations, the restoration of the riparian zone can also be of great importance in urban streams. Bernhardt and Palmer (2007) found riparian zone restoration to be one of the most common techniques utilized across the United States, both in urban and rural systems. Often times this involves the removal of exotic riparian plants, and the replanting of native flora. Urban riparian restoration often follows instream restoration activities like channel realignment, and is often required to repair damage done to the riparian zone during instream restoration work (Groffman et al 2003). While riparian zone restoration has been shown to help with hydrological management and erosion control, results are mixed when it comes to its effects on the biota of the stream (Bernhardt et al. 2005; Haase et al. 2013).

All of the aforementioned restoration techniques are not novel to urban watersheds, with many restorations of agriculture dominated and rural streams also utilizing a combination of techniques (Bernhardt et al. 2005; Doyle and Shields 2012). However, urban restorations are often more challenging than their counterparts, with the added hydrological, physical, chemical, and biotic problems associated with Urban Stream Syndrome, as well as the added challenge of stakeholder involvement and available land adjacent to streams (Paul and Meyer 2001; Walsh et al. 2005). Urban restorations require more stakeholder participation and planning, as well as a thorough integration of social and ecological sciences to be successful (Paul and Meyer 2001). Locating and gaining access sites suitable for restoration in urban areas is also more difficult because of higher land prices or scarcity of public lands (Bernhardt and Palmer 2007). Private land directly adjacent to the stream reduces the ability to restore a functional floodplain without participation and support from private landowners. In contrast to the restoration in non-urban areas, urban stream restorations often have an aesthetic element that may take priority over, and negate the effects of more ecologically oriented restoration approaches (Wolter 2010). Urban restorations are more costly in terms of financing and the resources needed to execute them than non-urban restorations.

Over $1 billion US dollars have been spent annually on restorations since 1990, the majority of which have been targeted on smaller, more expensive urban restorations averaging 60% of the stream length of rural restorations (Bernhardt et al. 2005; Bernhardt and Palmer 2007). However, despite the large number of urban restorations, and restorations in general, very little post project assessment and research has been conducted, with Bernhardt et al. (2005) estimating that only about between 5 and 20% of restoration projects utilizing post-restoration assessment. Additionally, many post restoration assessments have yielded conflicting results on the effectiveness of restoration techniques on the hydrological, physical, and biotic health of the restored streams (Bernhardt et al. 2005; Jahnig et al. 2011). With this being the case, there is a need for post-restoration assessment to inform decision-making on urban stream restorations.

The combination of continued rapid urbanization, its degradation of urban stream systems, the subsequent increase in restoration of those stream systems, and the lack of post-restoration assessment of restoration success leaves a critical knowledge gap. This knowledge gap becomes even more important to fill given the financial resources dedicated to restoration techniques that may not be ecologically effective. This overall goal of my thesis is to evaluate a substantial urban stream restoration project to evaluate the relationship between physiochemical rehabilitation and the restoration of ecosystem function to determine the potential for the improvement in ecosystem services, specifically the sustainability of a sport fishery.

West Branch of the DuPage River – A Case Study

The stream selected for this study is the West Branch of the DuPage River, which runs through the western suburbs of Chicago before joining with the East Branch of the DuPage River to become the DuPage River. The DuPage River then runs into Des Plaines River, which flows into the Illinois River. A 13 km stretch of the West Branch between West Chicago and Naperville was recently extensively restored as part of a $71 million superfund cleanup and restoration project conducted by USEPA and Engineering West Ltd. This project provides an opportunity to fill knowledge gap and evaluate a unique large-scale restoration in urban setting.

Between 1931 and 1973, low levels of radioactive thorium were dumped into the water and substrate from a rare earths facility run by the Kerr-McGee Chemical Company. After the facility was shut down in 1973, Kerr-McGee conducted approximately 120 residential property cleanups in the West Chicago area. In 1994, the EPA designated the area as a Comprehensive Environmental Response, Compensation, and Liability Act Superfund cleanup site due to the residual thorium in the soil and sediments in and around the river. Between 1995 and 2006, Kerr-McGee completed cleanup of 676 residential properties that had used contaminated sediments during construction (U.S. EPA 2009). In 2004, the EPA proposed a cleanup of the West Branch, as well as a tributary, Kress Creek. The $71.9 million cleanup stretched from about 2.4 km north of the confluence of the streams to about 6 km south at McDowell Grove (U.S. EPA 2009), and was approved in 2005. The restoration project was broken up into 8 reaches, with each done at a staggered rate between 2005 and 2012. Each reach was dewatered, contaminated sediment was removed, and then instream restoration was completed (U.S. EPA 2009). The restoration also involved the clearing of non-native riparian plants followed by the planting of native ones to restore a more natural riparian zone and floodplain around the stream. To restore a more natural flow and hydrology, as well as reduce flooding, the dam at Warrenville Grove was removed, meanders were recreated in several locations, and wetland areas were created to reduce the flooding and provide habitat. A variety of instream habitats including pool-riffle complexes, boulder fields, upturned root wads, and high quality substrates were constructed to create more heterogeneous habitat for fish and macroinvertebrates. Enhancement of riverbanks, floodplain vegetation restoration, addition of native fish and mussels from other areas of the stream, and drainage improvements were also completed (U.S. EPA 2009; FPDDC 2012). These strategies were incorporated to not only decontaminate the streambed, but also to regain ecological form and function that could support a thriving sport fishery. The actual instream restoration of the West Branch began in 2005, with heavy restoration finishing in 2012. The remediation of contaminated sediments was the catalyst for restoration of the West Branch, providing the opportunity to improve instream and riparian habitat.

In contrast to the West Branch, the East Branch of the DuPage River, while not polluted with radioactive Thorium mine tailings, is still a degraded, unrestored urban stream. The East Branch is considered by the EPA to be impaired due to excess nutrients, silt, chlorides, habitat alteration, and invasive aquatic plants (EPA 2004b). In the following chapters, I compare the restored West Branch to the unrestored East Branch to evaluate abiotic and biotic responses to habitat changes only, because Thorium contamination was restricted to the West Branch. The East Branch provides a reference for pre-restoration conditions on the West Branch due to it's similar habitat impairment, watershed size and land use, as well as its hydrologic proximity, regardless of the lack of Thorium contamination on the East Branch.

The West Branch of the DuPage River is an opportunity to study a large scale restoration on a small urban stream, making it unique compared to most restoration projects in the US, which are smaller in spatial scale and less costly (Bernhardt et al. 2005). Additionally, the proximity of the East Brach of the DuPage River provides a reference stream which to compare to the West Branch restoration activities in the same urban watershed with similar landscape development features. The West Branch watershed has drainage of 156 km^2, dominated by 33% residential land use, with 14% of the surfaces in the watershed being impervious, 17% being zoned as agricultural, 17% vacant, 5% commercial, and 4% industrial. The East Branch drains 122 km^2, with 40% of the land in the watershed designated as residential, 20% vacant land, and 16% covered by impervious surfaces, as well as 7% commercial and 4% industrial land (EPA 2008).

My thesis serves as a post-project evaluation of the restoration of the West Branch utilizing the East Branch of the DuPage as a reference stream, and is comprised of three major objectives:

Objective 1: Assess differences in habitat and water quality

Objective 2: Assess differences in the macroinvertebrate communities

Objective 3: Assess the response of fish community structure, abundance, and species diversity to the restoration of the West Branch.

Additionally, I will assess fine scale differences between the streams based on habitat, water quality, and macroinvertebrate community data. I also intend to use this data to examine the abundance, distribution, and habitat usage of a top predator Smallmouth Bass (*Micropterus dolomieu*) on both branches. In addition to providing a post-restoration assessment of the ecological impacts of the restoration on the West Branch, results of my thesis will inform managers and restoration ecologists on future restorations.

Research Strategy

To assess the impacts of the restoration on the West Branch of the DuPage River, using the East Branch as a reference system, I intend to complete a three-part project. The first (Chapter 2) is a comparison of physical, hydrological, and water quality variables between the two streams. Since they are both urban streams within the same watershed, the effect of the restoration of the West Branch on these characteristics should become apparent when compared to the East Branch. The second objective (Chapter 3) is to determine if there are differences between the macroinvertebrate communities in the two streams, as well as to determine what, if any, of the abiotic variables contributed to these differences. The final objective (Chapter 4) is to assess the differences in fish communities between the streams after restoration on the West Branch. Along with this assessment, I intend to link abiotic variables as well as macroinvertebrate community data to fish community metrics, as well as the population and habitat usage of a top predator and prized sport fish, Smallmouth Bass. Ideally, a before-after-control-impact study would take place with data from both streams both pre- and post-restoration, but the lack of consistent pre-restoration data makes the use of a BACI design impossible. The proximity, size, and comparable land use of both watersheds however, support the use of the East Branch as reference stream.

CHAPTER 2: PHYSICAL HABITAT AND WATER QUALITY

Introduction

The growth of human population and the resultant expansion of urban centers has led to the loss of natural areas and caused degradation to the ecology of urban watersheds (US Census Bureau 2010; Zhao et al. 2011). Urbanization has both direct and indirect effects on the ecology of urban streams, making aquatic ecosystems in urban areas particularly prone to degradation of stream hydrology, instream and riparian habitat, and water chemistry, all of which affect the distribution and abundance of stream biota (Paul and Meyer 2001; Bernhardt et al. 2005; Walsh et al. 2005). In recent years, stream restoration efforts have become a common practice to attempt to mitigate or reverse the negative impacts of urbanization on the watershed and its biota (Bernhardt et al. 2005; Walsh et al. 2005).

Watersheds at any stage of urbanization undergo physical changes that directly impact aquatic ecosystems. As human population density increases in urban centers, physical alterations to streams for flood control, conveyance, and erosion control, including the straightening and hardening of stream channels, the addition of dams to manage flows, and the reduction of stream banks and riparian zones to facilitate human use are commonplace due to societal needs (Booth and Jackson 1997). The result is a highly altered stream ecosystem lacking meanders, stream edge wetlands, and a functioning floodplain.

As urban areas develop, land surfaces become increasingly impervious, resulting in rapid run-off and intense flooding events (Finenbine 2000; Walsh et al. 2005). The development of efficient man-made storm water and treated waste water infrastructure impedes natural filtration into the soil causing shorter lag time to peak flows and higher peak flows following precipitation events (Paul and Meyer 2001; Walsh et al. 2005). Increases in areal extent of impervious surfaces can restrict natural groundwater recharge, causing lower base flows and, in some urban landscapes, converting streams from perennial to seasonally intermittent lotic systems during the summer (Finkenbine 2000). These changes in hydrology have profound impacts on the physical characteristics of the stream environment. Increased flashiness of the hydrograph increases erosion and scour of the streambed, intensifying sedimentation in depositional zones, altering pool depth and channel width of the stream by the rapid movement of large amounts of sediment. The addition of sediment from runoff also changes channel shape, and many times converts natural meanders into more straight channels (Paul and Meyer 2001). Dams and other manmade additions to streams change the physical characteristics by altering flow as well (Walsh et al. 2005). These alterations accumulate to reduce instream habitat heterogeneity, which includes the reduction of important refuges and habitats for macroinvertebrates and fish (Walsh et al. 2005).

Densely populated areas generate both point and non-point sources of pollution due to a variety of human activities such as land development, which increases in impervious surface and storm water drainage, negatively impacting stream water chemistry (Wang 2001; Walsh et al. 2005). Increased density of roads in the urban environment and the use of road salt, coupled with lack of natural drainage, commonly cause increased salinity in urban and urbanizing streams (Morgan et al. 2012; Wu et al. 2015). Urban streams tend to have larger daily temperature variation than non-urban streams, as well as higher summer and lower winter temperature, due to heated runoff from impervious surfaces, removal of riparian zone vegetation, and decreased recharge of groundwater (Kinouchi et al. 2007). Urbanization, specifically the influx of organic material and effluent, serve to reduce dissolved oxygen content in many urban streams as well as increase biological oxygen demand needed to break down these organic materials (Walsh et al. 2005; Herringshaw et al. 2011; Miskewitz et al 2013).

Restoration of urban streams has become increasingly common as the negative effects of urbanization have become clearer, and as the desire to protect clean drinking water and attractive riverfront aesthetics from the negative effects of urbanization on water quality has become more prevalent (Bernhardt and Palmer 2007). Restoration of urban streams most commonly seeks to mitigate the impacts of urbanization and restore hydrological, physical, and ecological function to lotic ecosystems (Bernhardt et al 2005; Palmer et al. 2010; Bernhardt and Palmer 2011; Doyle and Shields 2012), and are undertaken with a wide variety of goals in mind, including improvement of water quality, physical stream habitat restoration, fish passage, riparian zone and flood plain management, erosion control, dam removal, and aesthetic improvements for public use (Bernhardt et al. 2005). Though restorations of streams are not exclusive to urban environments, they are often more difficult than in non-urban watersheds due to the challenges presented by urbanization as well as the addition of human attention and stakeholders in urban environments (Paul and Meyer 2001; Walsh et al. 2005). The complex nature of improvements combined with the rapid increase in both urbanization and the increase in restorations has created a need to evaluate physical restoration projects and their effects on stream biota.

Though there has been a drastic increase in both the number and cost of urban stream restorations (Bernhardt et al. 2005), post restoration assessment of streams has been limited to non-existent (Small 2012; Stranko et al. 2012). The few post restoration assessments that have been completed have often yielded conflicting results regarding the effectiveness and results of the restorations (Bernhardt et al. 2005; Jahnig et al 2011), leaving a critical knowledge gap in the assessment of restoration effectiveness. Proper post restoration assessment is vital to inform both future restoration and management decisions.

The goal of this study was to evaluate the restoration of an urban stream with respect to its physical and chemical characteristics. I assessed physical habitat metrics that would indicate improvements to the restored stream including instream and riparian zone habitat, channel morphology, pool and riffle quality, substrate quality, and gradient, as well as water quality variables including temperature, dissolved oxygen, flow, and conductivity. I hypothesized that the physical restoration of the West Branch would improve stream habitat quality and result in improvements in water quality metrics when compared to an unrestored reference stream in the same watershed.

Methods

Study Sites

The West Branch of the DuPage River is a second order stream within the Chicago, IL metropolitan area that underwent a major instream habitat restoration project from 2005 – 2012. I assessed the effects of that restoration by making comparisons to a relevant reference stream, the unrestored East Branch of the DuPage, which is a sister stream of the West Branch (Figure 2.1). Both streams are first to second order streams flowing from north to south in northeastern Illinois. The West Branch watershed encompasses approximately 329 square km, nearly all in DuPage County, with small portions in Kane, Cook, and Will Counties. Thirty-three percent of the land in the watershed is classified as residential, and 14% of the watershed is covered by impervious surfaces (Illinois EPA 2004a). The East Branch watershed covers 212 square km immediately to the east of the West Branch, with 40% of the land use in the watershed being residential, and 16% covered by impervious surfaces (Illinois EPA 2004b). Eight sites each on the West and East Branches of the DuPage River were selected for this study. Sample sites on the West Branch were located within a 13 km restored reach of the river from the cities of West Chicago, IL to Naperville, IL. Sample sites on the East Branch were selected along a 22.5 km degraded reach between Glendale Heights, IL, and Woodridge, IL (Figure 2.1). Each site was 45.7 m in length and located in a wadeable reach adjacent to Forest Preserve District of DuPage County (FPDDC) property to permit access for sampling of fish and invertebrate communities. In consultation with Illinois Department of Natural Resources (IDNR) fisheries biologists as well as personnel from the FPDDC, sites were selected to be representative of the diversity of habitats occurring throughout the entirety of each stream.

Habitat Quality

Instream habitat assessments were conducted once at each site during July and August of 2014, while the streams were within 25% of base flow (USGS 2015). The Qualitative Habitat Evaluation Index (QHEI) for Midwestern streams developed by the Ohio EPA (2006) was used to characterize instream habitat, and the same observer scored all 16 sites to ensure consistent scoring among sites. QHEI is a qualitative evaluation of instream and riparian physical habitat, where observers assign scores for each of six physical characteristics of a stream segment: substrate, instream cover, channel morphology, bank erosion and riparian zone, pool/glide and riffle/run quality, and gradient. Each of the six metric scores of the QHEI has a maximum ranging from 10 to 20, and is the sum of several components. Six metric scores are then summed to determine the total QHEI score, which has a maximum of 100 (Table 2.1). The Ohio EPA methodology provides a general characterization of the quality of stream habitat based on the total QHEI score (Table 2.2). In the current study, total QHEI scores were used to compare habitat quality between the restored West Branch and the degraded East Branch, and the six individual metric scores were used to determine specific habitat characteristics responsible for any differences in overall habitat quality. Individual metric scores, as well as their components, were used as covariates in analyses of biotic indicators of stream health conducted in Chapters 3 and 4.

Water Quality

Water quality data were collected at each site in the fall of 2013, and the spring, summer, and fall of both 2014 and 2015, coinciding with fish community sampling. For each sampling event, water quality data were collected at the downstream end, middle, and upstream end of each sampling site, and the mean of the three data points was calculated for analysis. Conductivity (microSiemens/cm), water temperature at the middle of the water column (°C), and dissolved oxygen (mg/L) were measured at all sites using an YSI Professional Plus Multimeter Instrument (Yellow Springs, OH). USGS streamflow gauges (USGS 2015) were utilized to determine stream flow (cubic meters/second) at the time of each sampling event, with sampling occurring when flow was within 25% of base flow.

Statistical Analyses

A series of analyses were used to identify differences in abiotic conditions between streams and among seasons prior to the analysis of macroinvertebrate and fish community metrics. QHEI scores for all site visits were treated as independent samples and a Wilcoxon rank sum test was used to test for differences in total QHEI and metric scores between the restored West Branch and the degraded East Branch. Sinuosity of each stream was calculated by dividing channel length by downvalley length from the sampling site furthest upstream to the most downstream site. Conductivity, temperature, and dissolved oxygen content were determined to be non-normally distributed, so a Kruskal-Wallis test was used to test for differences among sampling sites to determine upstream-downstream differences in abiotic conditions both within the West and East Branches and among all sites combined. Generalized linear models with sites as random effects were utilized to test for differences in water quality variables for both streams and seasons. Regression analysis was used to test water quality variables against Julian Date and distance from the confluence of the streams, as stream water quality can often vary temporally (Ouyang et al 2006) and spatially (Vannote et al. 1980).

Results

Habitat Quality

QHEI scores ranged from 37.5 to 76.0 ($\bar{x}$ = 61.9; SE = 3.2) for all sites, with only two “excellent” sites, eight “good” sites, and the remaining six sites ranking either “fair” or “poor”. The West Branch ($\bar{x}$ = 70.3; SE = 2.0) had higher quality habitat based on overall QHEI scores (Wilcoxon rank sum $W = 57$; $p < 0.01$) than the East Branch ($\bar{x}$ = 53.1; SE = 4.4). All eight sites on the West Branch were categorized as either "excellent" (n=2) or "good" (n=6), while only two sites on the East Branch were categorized as "good", with the remaining classified "fair" (n=3) or "poor" (n=3). The difference in total QHEI scores between the restored West Branch and the degraded East Branch was the result of differences in the substrate, channel morphology, and pool/glide riffle/run quality metrics (Table 2.2).

Substrate metric scores on the West Branch ($\bar{x}$ = 16, SE = 0.4) were significantly higher ($W = 55.5$, $p = 0.01$) than scores on the East Branch ($\bar{x}$ = 11.7, SE = 1.4), indicating higher amounts of high quality substrate types such as boulder, cobble, and gravel on the West Branch (Figure 2.2). All 16 sites had substrate material that was derived from tills (glacially deposited sediment), leaving substrate type, silt cover, and embeddedness to explain the differences between the West and the East Branches. Higher quality substrate types on the West Branch were predominately boulder (19%), cobble (35%), and gravel (30%), with a smaller proportion of lower quality sand (11%), silt (3%) and muck (0.1%). On the East Branch, high quality boulder (6%) and cobble (16%) were less common than on the West Branch, while gravel (49%), sand (14%), silt (10%) and muck (5%) were much more common, indicating lower overall substrate quality. Boulder/slab, the highest quality substrate type, was absent on both branches. Silt cover on the West Branch was considered to be in the normal range (Ohio EPA 2006) at all sites, while five sites on the East Branch had higher than normal amounts of silt, with three sites having moderate silt coverage, and two having heavy silt coverage. Overall silt coverage on the West Branch was significantly lower than the East Branch ($W = 12$, $p = 0.01$). The five sites that had higher than normal amounts of silt on the East Branch were the five most upstream sites. All sites on the West Branch had a normal amount of substrate embeddedness, while four of the sites on the East Branch had higher than normal amounts of embeddedness, making the substrate on the West Branch significantly less embedded ($W = 16$, $p = 0.03$). Extent of embedded substrate covering between 25% and 50% of the sampling reach is considered to be normal (Ohio EPA 2006).

Channel morphology metric scores (Figure 2.3) on the West Branch ($\bar{x} = 12.6$, SE = 0.6) were also significantly higher ($W = 58$, $p = 0.007$) than scores on the East Branch $\bar{x} = 8.3$, SE = 1), indicating a higher quality stream channel, and more stable microhabitats. Channelization ($W = 3.5$, $p = 0.0015$) differed significantly between the branches, with the East Branch being more channelized, contributing to lower overall metric scores. The development of pool and riffle complexes was higher on the West Branch ($W = 10.5$, $p = 0.017$), contributing to higher metric scores on the West when compared to the East Branch. Neither stability ($W = 19$, $p = 0.13$) nor sinuosity ($W = 25.5$, $p = 0.46$) differed significantly between the streams. However, sinuosity indexes calculated for the each stream between the northernmost and southernmost sampling sites showed the West Branch with a sinuosity of 1.33 and the East Branch with a sinuosity of 1.18, with 1.3 being the threshold between sinuous and meandering streams (Mueller 1968).

Pool/glide and riffle-run metric scores on the West Branch ($\bar{x}$ = 15,2, SE = 1.4) were significantly higher ($W = 53$, $p = 0.03$) than on the East Branch ($\bar{x} = 8.8$, SE = 1.7), indicating better quality pools, glides, and riffle-run complexes. Only two sites on the East Branch scored greater than 10 (out of 20) for this metric, while all but one site on the

West Branch scored greater than 10. Pool depths were fairly consistent across all sites on both streams, but riffles and runs were considered higher quality on the West Branch. High quality riffles had areas deeper than 10 cm, fair quality riffles had areas between 5 and 10 cm, and low quality riffles had no areas deeper than 5 cm. High quality runs had maximum depths of over 50 cm, and low quality runs had no depths over 50 cm. The scores for the riffle-run portion of the metric differed significantly between branches, with the West Branch receiving higher scores ($W = 9.5$, $p = 0.018$), with larger, higher quality substrate, lower amounts of substrate embeddedness, and deeper riffles and runs. Instream cover, riparian zone erosion, and gradient metric scores did not differ significantly between the two streams. Instream cover was similar in type and amount on both branches ($W = 34.5$, $p = 0.83$) where scores for the West Branch ($\bar{x} = 14.8$, SE = 0.9) were not significantly different than the scores for the East Branch ($\bar{x} = 13.3$, SE = 1.4). Additionally, amount of erosion, riparian zone width, and flood plain quality did not significantly differ between branches ($W = 37.5$, $p = 0.57$), explaining the lack of difference in the bank erosion and riparian zone metric between the West ($\bar{x} = 9$, SE = 0.5) and East ($\bar{x} = 9$, SE = 0.3) branches. The gradient did not vary significantly between streams ($W = 44$ $p = 0.07$), with similar scores on the West Branch ($\bar{x} = 2.3$, SE = 0.4) and East Branch ($\bar{x} = 2.0$, SE = 0).

Water Quality

Conductivity on the restored West Branch ranged between 778 mS/cm and 1486 mS/cm (median = 1008), and conductivity on the East Branch ranged between 678 and 1377 (median = 986) (Table 2.3). Conductivity did not differ by site on the West Branch, the East Branch, or amongst all sites (Table 2.4). There were no differences in the overall conductivities between West and East Branch samples ($t = 0.9$; $p = 0.35$). However, conductivity across seasons differed significantly, with conductivity values decreasing throughout the year (all p-values < 0.01). Additionally, using linear regression for each stream separately, conductivity was found to be highly negatively correlated to Julian Date on the West Branch (adjusted $R^2 = 0.68$, $p < .001$) and the East Branch (adjusted $R^2 = 0.73$, $p < .001$), as well as overall (adjusted $R^2 = 0.69$, $p < .001$; Figure 2.3).

Water temperature ranged between 9.3° and 27°C (median = 19.4°) on the West Branch, and 12.4° and 29.7°C (median = 19.8°) on the East Branch (Table 2.3), and did not differ among sites between streams nor within each stream (Table 2.4). Temperature did not significantly differ between the two streams ($t = -1.7$; $p = 0.09$). Temperature was not significantly different between streams in the spring or fall, but was higher on the West Branch during the summer sampling periods ($p < 0.01$). Temperatures on both the West (adjusted $R^2 = 0.76$, $p < 0.01$) and East (adjusted $R^2 = 0.51$, $p < 0.01$) branches also correlated with Julian date, and were fit using separate quadratic functions (Figure 2.5), indicating that temperatures tend to be highest during the middle of the year, and decrease towards both spring and fall. Temperature did not significantly correlate with distance from the confluence of the branches (adjusted $R^2 < 0.01$, $p = 0.17$), indicating that temporal variation, not spatial variation, is driving the differences in temperature.

Dissolved oxygen content (mg/L) ranged between 2.4 and 15.7 on the West Branch (median = 5.8), and 2.1 and 18.0 on the East Branch (median = 5.9) (Table 2.3). Dissolved oxygen did not differ between the West and East Branches ($t = 0.1$; $p = 0.97$). However, dissolved oxygen was significantly lower during the summer when compared to both the spring and fall ($p < 0.001$). Dissolved oxygen did not differ significantly by Julian Date ($t = -0.47$; $df = 106$; $p = 0.7$),

Stream flow on the West Branch ranged between 23 and 2470 cubic feet per second between September of 2013 and October of 2015 ($\bar{x} = 142$), and from 9 to 880 cf/s on the East Branch during this same time period ($\bar{x} = 58$). Flow was significantly higher on the West Branch when compared to the East Branch ($t = 7.5$; $p < 0.001$). Flow was significantly related to season as well, decreasing each season ($p < 0.001$). Similarly, flow on both branches was negatively correlated to Julian Date (Figure 2.6). Additionally, I utilized the Richards-Baker Flashiness Index (Baker et al 2004) to quantify the flashiness of both streams. During our sampling period, flashiness did not differ between the branches, with the mean R-B Index on the West Branch being 0.33 (SE = 0.01), and the mean on the East Branch being 0.32 (SE = 0.02). Both of these are within the middle 50% for the watershed size of the West and East Branches, and flashiness was similar to years prior to the restoration.

Discussion

Urbanization of the landscape has wide ranging negative effects on the physical and chemical characteristics of streams (Paul et al 2001; Walsh et al 2005). Addition and expansion of impervious surfaces increases surface runoff rates during rainfall events, and increases flow volume via wastewater treatment effluent, altering the hydrology of streams by changing the frequency, magnitude, and timing of flows (Knouft and Chu 2015). These changes in flow regime lead to changes in the physical characteristics of the stream such as increased pool depth and channel width, higher rates of erosion and scour, reduction of channel sinuosity, and reduction of instream habitats (Walsh et al. 2005).

The changes to a stream precipitated by urbanization also commonly cause reductions in dissolved oxygen (Walsh et al. 2005; Herringshaw et al. 2011), and both daily and seasonal temperature fluctuations are greater than in non-urban streams (Kinouchi et al. 2007). Increased density of roads in the urban environment and the use of road salt during the winter season are also linked to higher conductivity levels in urban waterways (Morgan et al. 2012; Wu et al. 2015).

In the current study, physical restoration of the West Branch resulted in improved instream habitat quality relative to the East Branch. Specifically, the West Branch has higher quality substrates, more natural stream meanders, and more frequent and higher quality riffle habitats. The addition of riffle habitats has been shown to benefit multiple macroinvertebrate and fish species, increasing species richness and diversity (Taylor 1999; Wang et al. 2006). Pools and riffle-run complexes installed in the West Branch created more habitat diversity, which should result in increased dissolved oxygen production (Burke 2006; Higashino and Stefan 2011). The removal of low-head dams, dredging, and floodplain wetlands restoration on the West Branch resulted in less embedded substrates and generally higher quality streambed habitat. The removal of dams helps aid in the reduction of sediment and nutrient blockage, and allowing for more natural flow downstream. Dam removal also increased connectivity of upstream and downstream areas on the stream, allowing for fish passage to restored sections above Warrenville dam.

Restoration of the riparian zone on the West Branch involved large-scale efforts to add natural riparian plants and floodplain wetland areas adjacent to the stream; however, riparian zones remained similar on the West and East Branch other than the addition of more native vegetation and wetland areas. Further, all study sites were adjacent to forest preserve district property where the width of the riparian zone and land use categories the floodplain were very similar throughout, leading to the lack of observed significant differences in riparian zone quality between the two streams. Stream restoration commonly includes a mixture of instream habitat and riparian zone improvement (Palmer et al. 2010), as was the case on the West Branch (Burke 2006). As a rapid assessment tool, QHEI can be effective to determine changes in habitat quality over time; however, the utility of QHEI to evaluate the effects of stream restoration activities may be limited due to its qualitative nature. If used in conjunction with more quantitative and expansive methods, would be more useful to detect changes in habitat quality over shorter time scales (Somerville 2010).

Restoration activities within an urban stream as well as in the riparian zone are expected to have indirect effects on water quality. Conductivity tends to be higher in urban landscapes due the nature of urban land cover, high density of impervious surfaces, and extensive road networks which require winter snow removal (Wang et al 2011; Wu et al 2015). Reports generated by the DuPage River Salt Creek Workgroup (2007) concluded that areas around both streams received nearly 1,300 lbs. of road salt per lane-mile per storm, which is 500 lbs. above the average for most major cities. High conductivity, which is indicative of dissolved organic and inorganic solids in a stream, can harm stream biota (Walsh et al. 2005; Kelly et al. 2012), by altering fish reproduction and larval development as well as reducing fish abundance and species richness (Morgan et al. 2012). High conductivity is associated with low macroinvertebrate richness, specifically causing reductions in EPT and other intolerant taxa (Johnson et al. 2013). Conductivity on the West Branch was well above expected Midwestern stream conductivity levels during all seasons, but consistent with recently reported conductivities on the both branches (Murphy and Willis 1996; ILEPA 2004a; ILEPA 2004b) and to levels observed prior to rapid urbanization (Kelly et al. 2012). Restoration activities on the West Branch do not appear to have lowered conductivity, likely because instream improvements have limited effect on land-based factors affecting conductivity in urban streams. The addition of riffles provides turbulent water, which improves dissolved oxygen for the stream. Temperature and dissolved oxygen together are critical factors in determining species richness and abundance of macroinvertebrate communities (Collier and Clement 2011), as well as movement patterns and distributions of fish and macroinvertebrate species within a system (Caissie 2006; Kaller and Kelso 2007; EPA 2008). In this study, the West Branch experienced higher temperatures in the summer than the East Branch, which may be the result of newly created floodplain and wetland areas allowing more water to be heated prior to seeping into the stream (Poole and Berman 2001). Though the restoration may have been intended to improved dissolved oxygen content in the stream, it is likely that continuing inputs of polluted runoff from anthropogenic sources are impeding the recovery of dissolved oxygen in the West Branch, but further monitoring of abiotic conditions at finer spatial and temporal scales is needed. Data generated by this study do not show that there were water quality improvements due to restoration, but do show that both streams have temperature and dissolved oxygen content within the expected range for sustaining aquatic life. With these results, I expect that physical habitat improvements on the West Branch will be drivers of differences between the macroinvertebrate and fish communities on the West and East Branches.

As with instream restoration activities, riparian zone restoration is similarly designed to improve water quality through indirect effects. For example, Bernhardt et al. (2005) noted that the restoration of riparian zones during stream restoration is often used to create a buffer zone to reduce the organic chemical input into the stream. Wetland buffers are utilized to reduce flooding during rain events, as well as a buffer for chemical inputs (Castelle et al. 1994). The water quality parameters that I measured did not include chemicals that would be affected by riparian zone improvements, preventing a more in depth analysis into the impact of restoration in the riparian zone and flood plain. Future research into the water quality in the two streams could allow for more insights into riparian zone restoration.

Landscape scale anthropogenic alterations to flow are common, especially in urban settings (Poff et al. 1997, Walsh et al. 2005), and can have wide ranging impacts including increased flooding, high flashiness, scouring and erosion of stream banks and habitat, and the disturbance of fish and macroinvertebrates (Walsh et al. 2005). Restoration actions designed to support a more natural hydrology in urbanized systems have been shown to mitigate flooding and flow issues, but does not necessarily lead to restored ecosystem health (Haase et al 2013). Restoration of the West Branch did not appear to mitigate or impact flooding to any discernable degree, likely due to the fact that landscape scale changes to the stream hydrology were not addressed during the restoration. In this study, flow was higher on the West Branch consistently across all seasons due to the slightly larger size of the West Branch watershed and slightly larger stream size. Both the East Branch and West Branch experienced highest flows in spring during the study period, consistent with historical data for both branches (USGS 2015). Flashiness on both streams was consistent with Midwestern streams of their size (Baker et al. 2004); regardless of urban or rural land use. It is possible that the flood intensity (i.e., maximum flood height) and not the overall flashiness (i.e., rate of flood level rise and fall) could have a greater impact through the rapid introduction of chemicals into the stream and scouring of the streambed. Further research into organic contaminants in the stream may help to quantify this impact.

Despite evidence that the restoration of instream and riparian habitat on the West Branch of the DuPage resulted in significant improvements in habitat quality, I observed no water quality improvements with respect to temperature, dissolved oxygen, or flow regime. Specifically, restoration of the West Branch has had a positive impact on substrate quality, diversified channel morphology, and improved instream habitat features, which should provide benefits to macroinvertebrate and fish communities despite a lack of water quality improvements. Further study of anthropogenically derived water quality parameters, including nitrates, ammonium, and inorganic pollutants may further clarify whether filtration function of riparian habitat will improve water quality. Restoration efforts in upland areas throughout the watershed would be required to address storm water inputs, pollutants, and sediment that may more strongly impact flow regime, and water quality of the stream may be necessary for full ecological recovery of the stream.

Figures

Figure 2.1. 16 sampling sites, eight each on the West Branch of the DuPage River (left, W1-W8) and East Branch of the DuPage River (right, E1-E8).

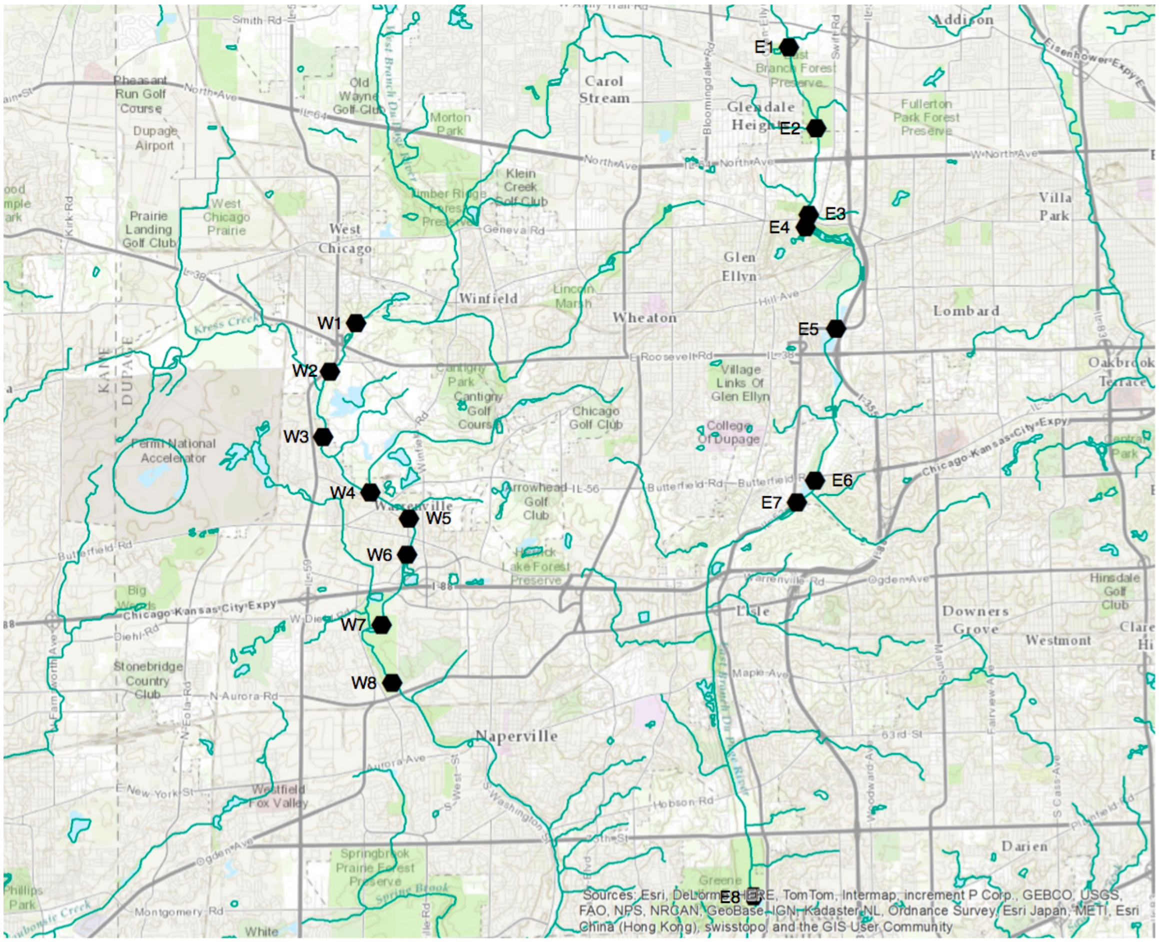


Figure 2.2. Mean proportions (with standard error) of substrate types found at all 16 sites on the West and East Branches of the DuPage River. Substrate types are in descending order of quality (i.e., QHEI component score).

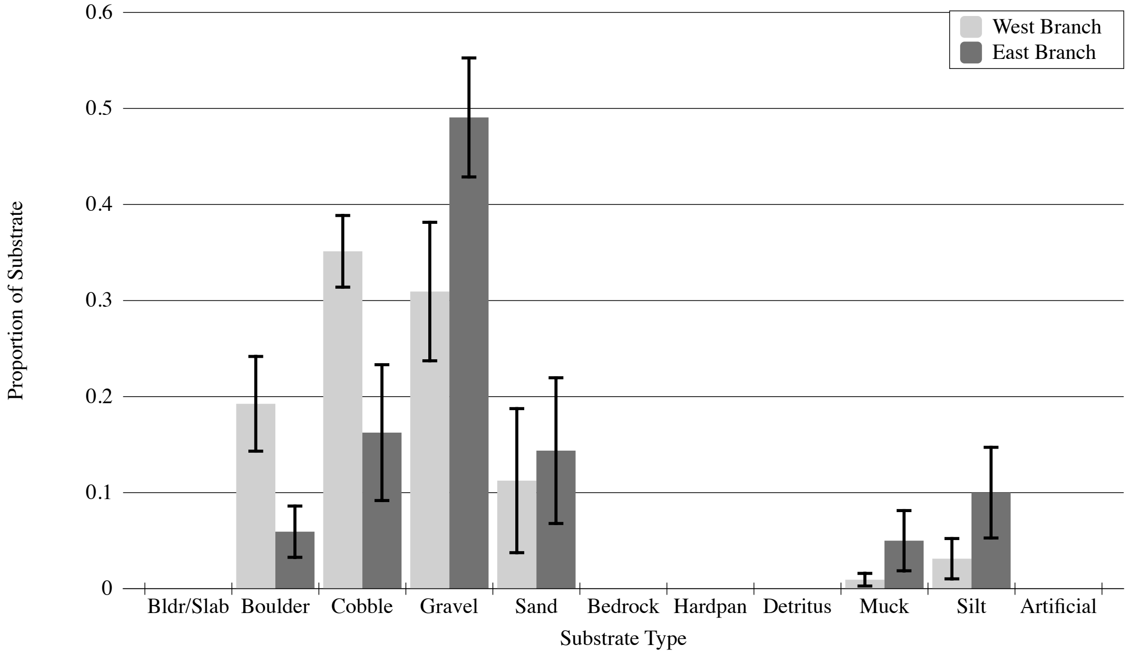


Figure 2.3. Submetric scores for all channel morphology for all sites on the East and West Branch of the DuPage. Channelization and development scores were significantly higher on the West Branch.

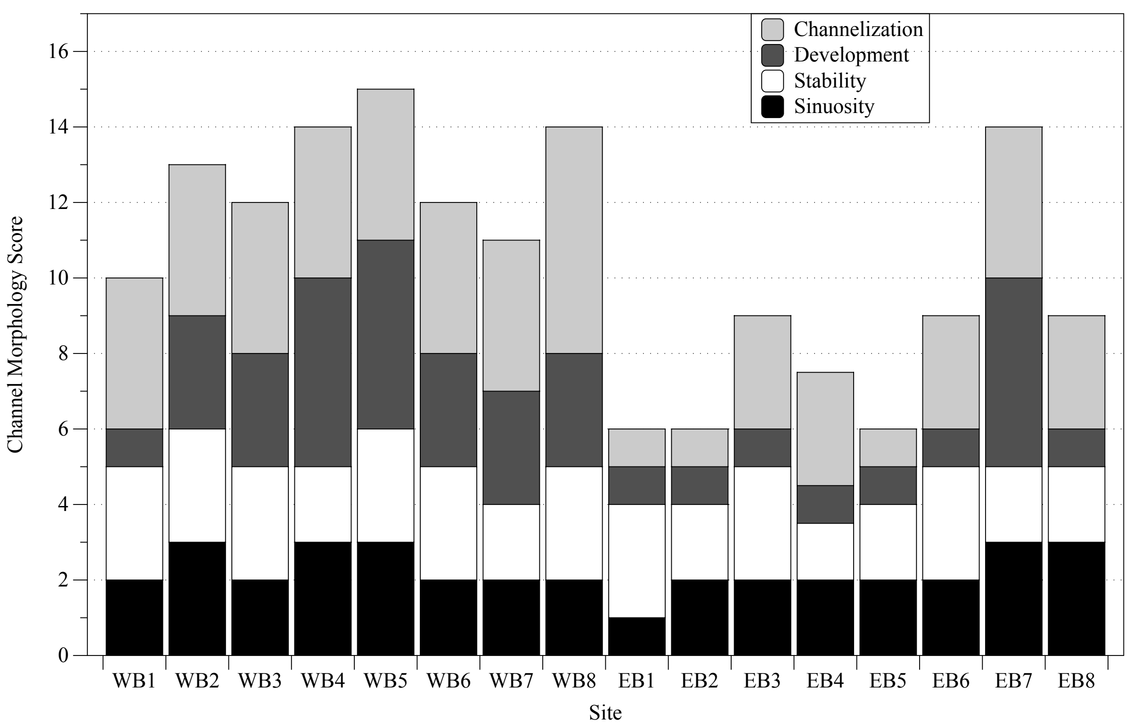


Figure 2.4. Conductivity (mS/cm) for West Branch (gray) and East Branch (black) based on the Julian Date that sampling took place. Regression lines for each branch are in the same colors.

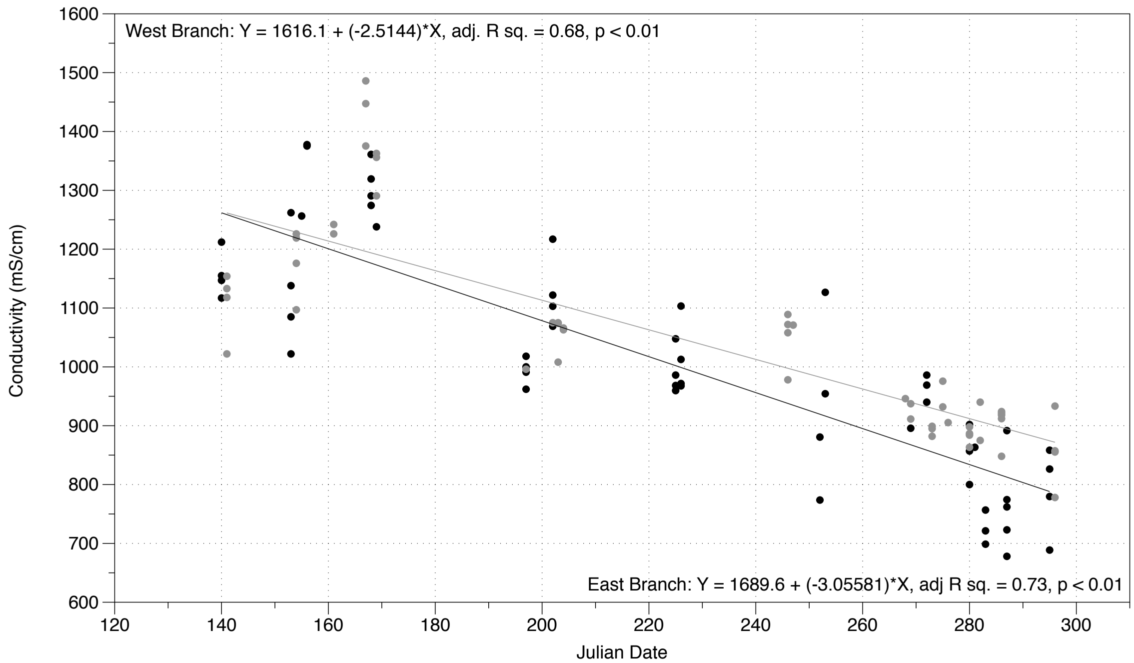


Figure 2.5. Temperature vs. Julian Date for both the West and East Branches. Models were fit using quadratic functions, and the models with their adjusted R square values can be found in the figure.

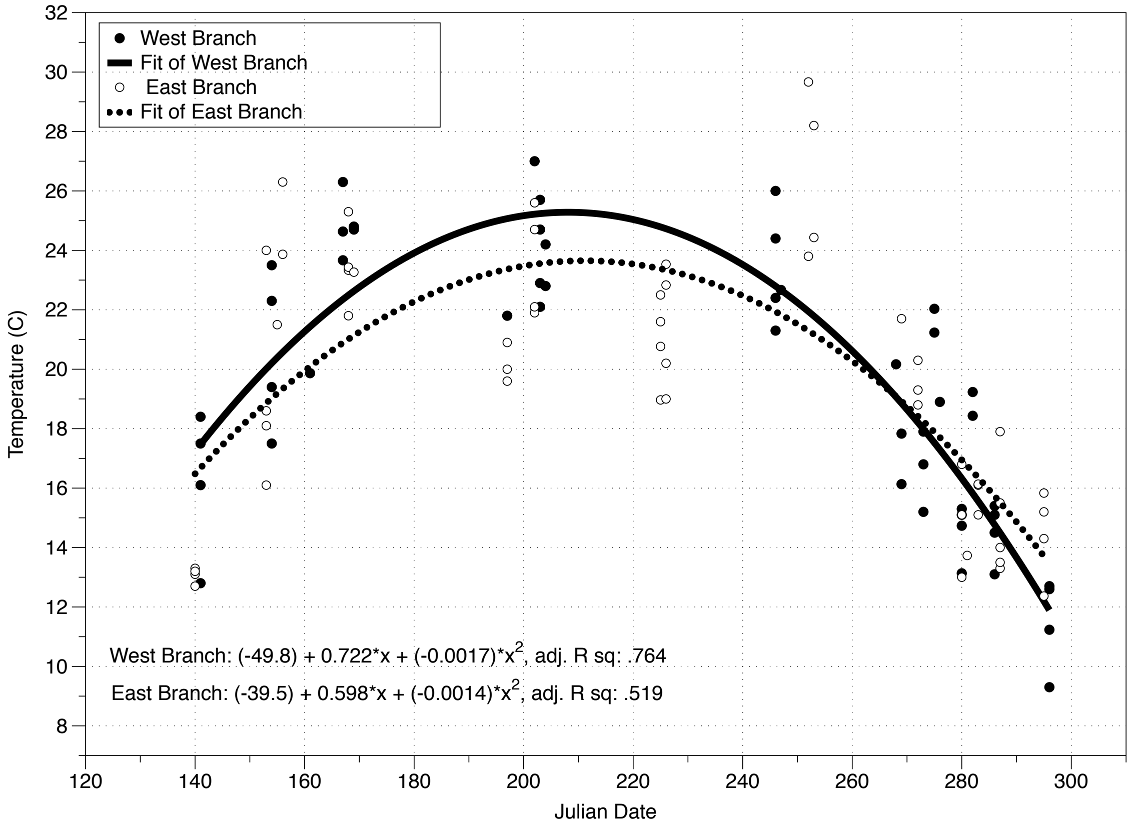
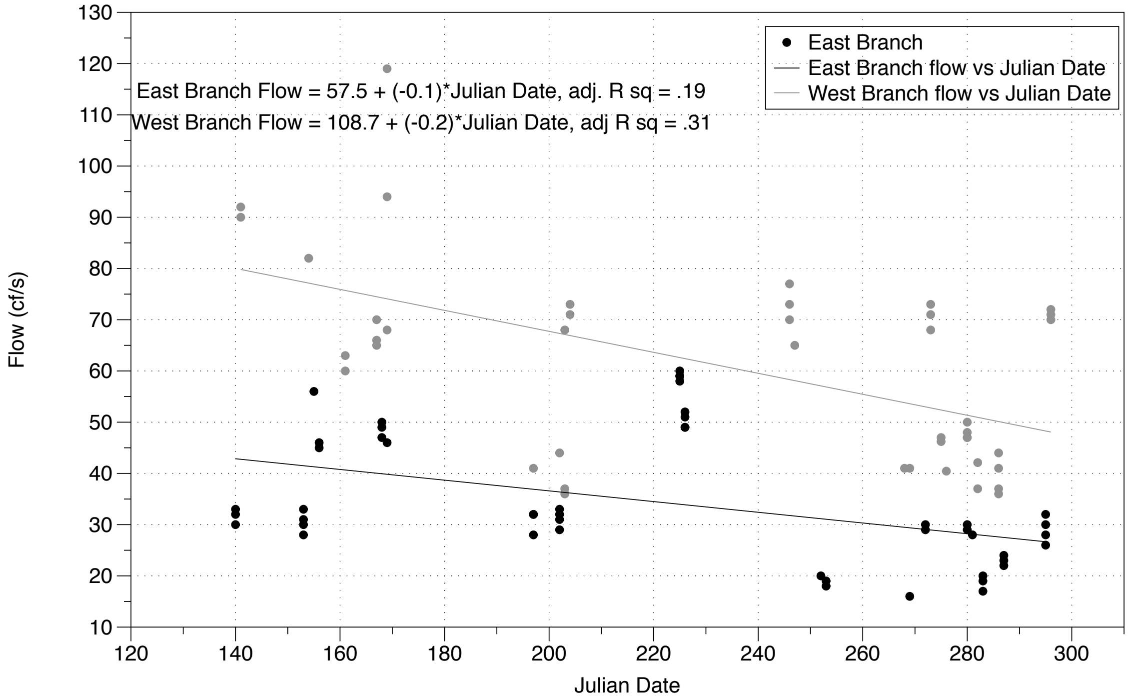


Figure 2.6. Linear functions for flow vs. Julian date for both the West Branch (gray), and East Branch (black) of the DuPage River. Regression formulas and associated adjusted R squares are included in the figure.



Tables

Table 2.1. Metrics and scoring for the Ohio EPA Qualitative Habitat Evaluation Index.

Metric		Score
Substrate		Maximum 20
	Type	0 – 10
	# best types	0 – 2
	Origin	-2 – 1
	Silt quality	-2 – 1
	Silt embeddedness	-2 – 1
Instream Cover		Maximum 20
	Type	0 – 9
	Amount	1 – 11
Channel Morphology		Maximum 20
	Sinuosity	1 – 4
	Development	1 – 7
	Channelization	1 – 6
	Stability	1 – 3
Bank Erosion and Riparian Zone		Maximum 10
	Erosion	1 – 3
	Riparian width	0 – 4
	Flood plain quality	0 – 3
Pool/Glide and Riffle/Run Quality		Maximum 20
	Maximum depth	0 – 6
	Channel width	0 – 2
	Current velocity	-1 – 1
	Total Pool submetric	Maximum 12
	Riffle depth	0 – 2
	Run depth	1 – 2
	Riffle/run substrate	0 – 2
	Riffle/run embeddedness	-1 – 2
	Total Riffle submetric	Maximum 8
Gradient		Maximum 10
	Gradient	2 – 10
Total QHEI		Maximum 100

Table 2.2. All individual metric scores as well as mean metric scores for both West and East Branches with standard error, and p-values for Wilcoxon Signed-Rank Test. Any significant p-value is denoted with *.

Site	Substrate	Instream Cover	Channel Morph.	Bank / Riparian	Pool/Glide, Riffle-Run	Gradient	Total
West 1	16	15	10	10	8	2	61
West 2	16	16	13	9.5	14	2	70.5
West 3	18	13	12	10	11	2	64
West 4	18	15	14	9.25	17	2	75.25
West 5	14	15	15	7.5	18	4	73.5
West 6	17	16	12	9	17	2	73
West 7	15	16	11	10	18	4	74
West 8	16	17	14	6.5	18.5	4	76
Mean:	16 (0.4)	14.8 (0.9)	12.6 (0.6)	9 (0.5)	15.2 (1.4)	2.8 (0.4)	70.3 (2)
East 1	13	16	6	9	7	2	53
East 2	7	10	6	9	9	2	43
East 3	13.5	16	9	8.5	9	2	58
East 4	7	8	7.5	10	3	2	37.5
East 5	8	8	6	8.5	8.5	2	41
East 6	16	15	9	10	4	2	56
East 7	14	16	14	8	18	2	72
East 8	14	17	9	9	12	2	63
Mean:	11.7 (1.4)	13.3 (1.4)	8.3 (1)	9 (0.3)	8.8 (1.7)	2 (0)	53.1 (4.4)
P-value:	0.013*	0.83	0.017*	0.59	0.03*	0.07	0.006

Table 2.3. Summary statistics for conductivities (mS/cm), temperature (degrees Celsius), and dissolved oxygen content (mg/L) for both the West and East Branches of the DuPage for spring, summer, and fall sampling periods. N = 24 for West and East fall samples, N=16 for East spring and summer samples, and for West Spring sample, and N = 13 for West Branch summer sample.

		West Branch		East Branch	
		Range	Median	Range	Median
Conduct	Spring	1022-1486	1226	1022-1377.7	1247.2
	Summer	978-1089	1071	959.7-1217	1006.3
	Fall	778-975	902.2	678-1126	841.7
	All	778-1486	1008	678-1377.7	986
Temp	Spring	12.8-26.3	21.1	12.7-26.3	21.7
	Summer	21.3-27	22.9	18.9-25.6	21.3
	Fall	9.3-22	15.4	12.4-29.7	16
	All	9.3-27	19.4	12.4-29.7	19.8
DO	Spring	2.6-15.7	6.6	2.2-13.1	5.7
	Summer	2.4-8.3	2.9	2.1-10.6	3
	Fall	4-11.2	7.7	3.9-18	6.8
	All	2.4-15.7	5.8	2.1-18	5.9

Table 2.4. Kruskal-Wallis H test scores for differences between sites on the West Branch, East Branch, and both. Degrees of freedom are denoted with ‘df’, and any significant p-values at alpha=0.05 are denoted with ‘**’

	Stream	Chi-squared	df	p-value
Conductivity	West Branch	1.69	7	0.975
	East Branch	4.97	7	0.664
	Both branches	7.4	15	0.946
Temperature	West Branch	1.38	7	0.986
	East Branch	3.44	7	0.841
	Both branches	4.89	15	0.993
Dissolved Oxygen	West Branch	1.48	7	0.983
	East Branch	2.89	7	0.895
	Both branches	4.72	15	0.994

CHAPTER 3: MACROINVERTEBRATES

Introduction

Population growth and expanding development of urban centers from 2000-2010 has been substantial (US Census Bureau 2010), resulting in the degradation of natural areas, including streams (Paul and Meyer 2001). Stream impairment resulting from urbanization affects the hydrological, physical, and chemical characteristics of flowing water, leading to measurable biological changes to the stream ecosystem. Increased impervious surfaces and storm water drainage lead to rapid fluctuations in flow, more flashiness, and higher and more frequent flooding (Walsh et al. 2005). More nutrients and pollutants are delivered into the system by unfiltered runoff flowing over impermeable surfaces, as well as by more point sources of nutrients and pollution (Paul and Meyer 2001). Drastic changes in flow, increased flooding, landscape changes such as increased impervious surfaces, and degradation of riparian zone due to development along shorelines lead to the scouring of banks and streambeds, reduction of instream habitat, and channelization of urban streams (Walsh et al. 2005).

The abiotic impacts of urbanization in turn negatively affect the biota of the stream, in particular the macroinvertebrate community. Urbanization reduces the number of sensitive macroinvertebrate taxa through reduction of physical habitat as well as reduction of water quality (Booth and Jackson 1997). Because macroinvertebrates play key roles in the stream ecosystem from influencing nutrient cycles, serving as prey items, to transporting organic material downstream (Wallace and Webster 1996), they are of particular interest when examining degraded streams. Additionally, macroinvertebrates are among the most common bioindicators of stream health (Lenat 1988; Klemm et al.

2003), making them an excellent focal point an ecological study into the effects of restoration. Macroinvertebrate community assessments are among the most common and effective methods of assessing stream quality, and can be an effective tool to assess the effects of stream restoration activities (Pander and Geist 2013).

In this study, I conducted an evaluation of the effects of a stream restoration conducted on the West Branch of the DuPage River on the macroinvertebrate community. I used the unrestored East Branch of the DuPage River as a reference stream, and utilized physical habitat and water quality data, along with macroinvertebrate metrics, to assess the differences between the two streams as an indication of the effectiveness of the restoration. I hypothesized that the benefits of the physical restoration of the West Branch would be reflected in the assessment of the macroinvertebrate community through higher numbers of sensitive and intolerant taxa, which would indicate improved stream health and ecosystem function.

Methods

Study Sites

Macroinvertebrate samples were collected at the same 16 study sites on the East Branch and West Branch of the DuPage River described in Chapter 2. Those sites contained a variety of substrates, including cobble, boulder, gravel, sand, mud, and silt. Additionally, instream habitat features such as pools, riffles, runs, aquatic vegetation, root wads, and other microhabitats were found to varying degrees throughout the study sites, as indicated by habitat data provided in Chapter 2. Sites on the restored West Branch generally had more habitat heterogeneity, and all available habitat and substrate types were sampled at each site.

Macroinvertebrate Sampling

Benthic macroinvertebrates were sampled at each site during the fall of 2013, and the spring, summer, and fall of 2014 and 2015, when the rivers were within 25% of base flow (USGS 2015). Macroinvertebrates were sampled with a D-frame dip net using the Illinois EPA standard 20-jab per site visit multi-habitat method (IEPA 2011). Samples were preserved in 70% ethanol in the field and transported to the lab for cleaning and analysis. In the lab, samples were cleaned of mud, detritus, and gravel, and all macroinvertebrates in each sample were identified to family using dichotomous keys (Burch 1989, Voshell 2002) with the exception of Oligocheata, Hydrachnidiea, and Hirudinea, which were identified to class, or lowest possible taxonomic level (Hilsenhoff 1988). After all macroinvertebrates were identified and enumerated, total abundance, taxa richness, Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness, percent of EPT individuals, percent of EPT families, Hilsenhoff Biotic Index (HBI), and the Average Score Per Taxon (ASPT), a version of the Biological Monitoring Working Party (BMWP) score was calculated for all samples.

HBI, a family level biotic index for the assessment of impaired waterways, is calculated using tolerance values (TVs) for a range of macroinvertebrates (Hilsenhoff 1988), with additional TVs being supplied by Hauer and Lamberti (2007). Tolerance values for taxa range from 0-10, with lower values assigned to taxa found only in streams with higher water quality, and higher values assigned to more tolerant taxa. HBI is calculated by multiplying the tolerance values for each family by the number of specimens collected in that family, summing all of those products, and then dividing by the total number of specimens in the sample.

ASPT is a macroinvertebrate biotic index used as an indicator of organic pollution in streams. Similar to HBI, it ranks families or higher taxonomic groups from 1-10, but unlike HBI, a higher score indicates a more sensitive group and therefore lower organic pollution. ASPT scores are derived from BMWP scores (Chesters 1980), which are calculated by summing the tolerance scores of all families present in the sample. ASPT scores are then calculated by dividing the BMWP score by the number of families. Groups for which there was no developed BMWP score were left out of the calculation of the BMWP, as well as the subsequent calculation of the ASPT.

Statistical Methods

Differences in taxa richness, EPT taxa richness, %EPT abundance, %EPT richness, HBI, and ASPT between streams and among seasons were each determined using a generalized linear model with stream as a fixed effect and site as a random effect. Correlations between response variables and the water quality and physical habitat variables discussed in Chapter 2 were calculated using Spearman's rank correlation, including a Bonferonni correction for multiple comparisons, due to the ordinal nature of many of the abiotic variables measured in Chapter 2. Nonmetric multidimensional scaling (NMDS) using Bray-Curtis distance matrices was used to elucidate patterns in the abundance of macroinvertebrate taxa across sites. Abundance data for each site was transformed (log[x+1]) prior to ordination (Jackson 1993; Pond et al. 2008). A Bray-Curtis dissimilarity matrix for samples was tested for differences between streams and seasons using analysis of similarity (ANOSIM). The species with the most influence on the differences in NMDS community composition were determined using the procedure

SIMPER (similarity percentage). All statistical analyses were conducted using R, and all p-values were considered significant at $\alpha = 0.05$.

Results

A total of 10,792 macroinvertebrates were identified during this study, spanning 62 families and three higher taxa (Oligocheata, Hydrachnidiea, and Hirudinea) where identification down to family was not feasible (Table 3.1). Total abundance for each sample ranged from 2 to 558, with a mean of 103 (SE = 10), and total taxa at a site ranged from 2 to 20, with a mean of 10.6 (SE = 0.4). The most common family found on the West Branch was Hyalellidae (21% of total abundance), followed by Simuliidae (18% of total abundance), and Chironomidae (15% of total abundance). On the East Branch, the most common family was Simuliidae (28% of total abundance), followed by Chironomidae (21% of total abundance), and Coenagrionidae, (19% of total abundance). All of the most abundant families found in both streams are common stream macroinvertebrates, thus the comparison of abundance alone provides little insight into possible differences in community composition between branches.

Total taxa richness ($t = 1.67$; $p = 0.12$; Table 3.2, Figure 3.2), as well as non-EPT taxa richness ($t = -0.5$; $p = 0.63$; Table 3.2, Figure 3.2) did not significantly differ between streams. Conversely, EPT taxa richness was significantly higher on the West Branch ($t = 4.7$; $p < 0.001$), indicating higher numbers of more sensitive EPT taxa, but no difference in overall taxonomic richness between the streams. The West Branch had a higher percentage EPT taxa richness (Figure 3.2) than the East Branch ($t = 4.3$; $p < 0.001$), and EPT individuals also comprised a larger percentage of the macroinvertebrate community on the West Branch than the East Branch ($t = 4.4$; $p < 0.001$), indicating the presence of more sensitive EPT taxa in the macroinvertebrate community on the restored West Branch (Table 3.2).

Taxonomic richness did not correlate significantly with any of the physical habitat variables, but did correlate with all water quality variables. However, EPT taxonomic richness did positively correlate significantly with substrate quality (rho = 0.55, $p < 0.001$) channel morphology (rho = 0.20, $p < 0.001$), pool/glide and riffle-run quality (rho = 0.15, $p < 0.001$), and total QHEI score (rho = 0.24, $p < 0.001$), all of which scored significantly better on the West Branch. Additionally, EPT taxonomic richness positively correlated with flow (rho = 0.45, $p < 0.001$). These patterns in both abundance and taxonomic richness show that generally the more sensitive EPT were found in areas with higher quality physical habitat than the general macroinvertebrate community, but interestingly, water quality variables were not highly correlated with EPT taxa or abundances.

Similar to EPT taxonomic richness, both percent EPT abundance and EPT percent of taxa were also positively correlated to the majority of physical habitat variables, indicating more sensitive macroinvertebrates in areas of higher quality stream habitat. Percentage of both significantly correlated to the substrate metric (rho = 0.53, $p < 0.001$ and rho = 0.50, $p < 0.001$, respectively; Table 3.3), channelization (rho = 0.53, $p < 0.001$ and rho = 0.51, $p < 0.001$, respectively), pool-glide/riffle-run quality (rho = 0.44, $p < 0.001$ and rho = 0.46, $p < 0.001$, respectively), and overall QHEI score (rho = 0.57, $p < 0.001$ and rho = 0.58, $p < 0.001$, respectively). These correlations to habitat metrics indicate that sites with higher quality habitat have communities with more sensitive EPT taxa. Additionally, both percent abundance (rho = 0.35, $p < 0.001$) and percent of taxa (rho = 0.32, $p < 0.001$) significantly correlated to flow, but none of the other water quality variables.

HBI scores on the West Branch were significantly lower than scores on the East Branch ($F = 16.5$; df = 1, 99; $p < 0.001$; Table 3.2), indicating the presence of more sensitive and pollution intolerant families on the West Branch than the East. Additionally, there was seasonality to HBI scores ($F = 8.9$; df = 2, 99; $p < 0.001$), with summer HBI scores being significantly lower than both spring ($p = 0.01$) and fall ($p < 0.001$), while spring and fall scores did not significantly differ from each other ($p = 0.85$). The interaction between stream and season was not significant; indicating that seasonality of HBI scores did not vary between the West and East Branches. Quantitative HBI scores have corresponding qualitative water quality rankings ranging from excellent (range of HBI scores), good (numbers), fair (numbers), poor (numbers). The West Branch had five samples with HBI scores that indicate good water quality, and seven that indicated fair water quality, while the East Branch had no samples that indicated good water quality, and seven that indicated fair water quality. Twenty-six of the 55 samples on the East Branch were rated as very poor, while only 12 of the West Branch samples were rated as very poor.

ASPT scores also showed that richness among pollution sensitive families was significantly higher on the West Branch when compared to the East Branch ($F = 10.1$; df = 1, 99; $p < 0.01$; Table 3.2). Additionally, ASPT scores varied across seasons ($F = 4.7$; df = 2, 99; $p = 0.01$), where ASPT scores were higher in the Fall compared to Summer ($p = 0.01$), but not significantly different from spring scores ($p = 0.14$), and summer scores were not from different from spring ($p = 0.64$). The stream by season interaction term was not significant for ASPT scores, indicating that the macroinvertebrate community in both streams changed similarly with season.

HBI was negatively correlated with substrate quality (rho = -0.42, $p < 0.001$), channel morphology (rho = -0.31, $p < 0.001$), pool-glide/riffle-run quality (rho = -0.33, $p < 0.001$) and overall QHEI score (rho = -0.43, $p < 0.001$), all of which were significantly higher on the restored West Branch. This indicates more sensitive macroinvertebrate communities at sites with higher quality substrate, better channel morphology characteristics, and higher overall habitat quality. HBI scores also negatively correlated with flow (rho = -0.31, $p < 0.01$). ASPT did not correlate to any abiotic variable.

The two-dimensional NMDS plot of log (x + 1) transformed total abundance for each site indicated a difference between the restored West Branch and unrestored East Branch (ANOSIM R = 0.21, P = 0.001; Figure 3.3). ANOSIM also revealed significant differences in macroinvertebrate abundance between seasons (R = 0.17, P = 0.001). The macroinvertebrate families that contributed the most to difference between the West and East Branches were Hyalellidae (contribution of 9.4% of the dissimilarity between branches), Simuliidae (8.6%), Leptohyphidae (8.3%), and Hydropsychidae (8.0%). Interestingly, both Leptohyphidae (Order Ephemeroptera) and Hydropsychidae (Order Tricoptera) are sensitive taxa, and were much more common on the West Branch, indicating that the differences in community composition may be indicative of improved stream quality. In addition, Chironomidae (7.8% contribution) and Oligocheata (3.8%), taxa typically found in large numbers in polluted streams, were more abundant on the unrestored East Branch.

Discussion

There is little debate that urbanization has drastic and broad reaching negative effects on the stream macroinvertebrate community (Paul and Meyer 2001; Walsh et al. 2005; Morrissey et al 2013; Docile et al. 2016). These effects include the reduction of sensitive species, dominance of more generalist species (Jones and Leather 2012) as seen in the reduction of EPT, and less taxonomic richness and abundance among more sensitive groups in general (Smith and Lamp 2008). There is limited evidence, however, that stream restoration practices in urban and non-urban environments are particularly effective at improving macroinvertebrate community composition (Palmer et al. 2010; Louhi et al. 2011; Stranko et al. 2012), likely because restoration practices have relied heavily the use of sediment traps and the addition of riprap (Alexander and Allan 2006). These restoration activities, however, may not have direct and positive impacts on macroinvertebrates, and in fact may have negative consequences in some cases. Sudduth and Meyer (2006) found that bank stabilization restorations had reduced intolerant taxa, total richness and diversity when compared to an unrestored reference site. Failures of restoration activities to provide demonstrable evidence of direct improvements to macroinvertebrate communities may be due to the fact that restoration projects tend to be small scale, very focused on a specific impact of urbanization (i.e. flooding, bank stabilization, etc.), and do not mitigate the multiple effects of urbanization (Violin et al. 2011). Additionally, stated restoration goals often do not take macroinvertebrates into account as a possible barrier to the ecological effectiveness of restorations (Laasonen et al 1998). The few studies that have noted the positive effects of restoration on the macroinvertebrate community often found only small gains in diversity, sensitive species, and abundance (Suren and McMurtrie 2006; Selvakumar et al. 2010). Additionally, Leps et al. (2016) found that time since restoration had little or no effect on the vast majority of macroinvertebrate community metrics, suggesting that reestablishment may not be a matter of time, but rather a matter of proper restoration.

In this study, I aimed to discover whether the macroinvertebrate population of the restored West Branch of the DuPage River differed from that of the East Branch of the DuPage River, an unrestored and degraded reference stream. Restoration of the West Branch involved the removal and replacement of poor quality substrate with higher quality substrates, naturalization of channel morphology, clearing of debris and bank stabilization, and installation of riffles and pool-glide complexes to increase habitat heterogeneity and improve water quality (Burke 2006). Macroinvertebrates can be an excellent indicator of water quality, and therefore can also be used to estimate the health of a stream and provide an indication of the success of a comprehensive restoration (Geist and Pander 2013). I hypothesized that the restoration of the West Branch would improve water quality and instream habitat, leading to a macroinvertebrate community with higher abundance and increased taxonomic richness that would shift in composition towards more sensitive taxa such as EPT when compared to the East Branch. Ordination of the West and East Branch communities showed that the macroinvertebrate assemblages were distinct between the streams largely due to consistently higher EPT taxonomic richness and EPT percent abundance on the West Branch, taxa richness of non-EPT macroinvertebrates were similar between both streams, indicating that increases in EPT on the West Branch reflected fewer of the more tolerant non-EPT taxa as well. The increase of EPT richness may be a reflection of improving water quality considering EPT taxa are sensitive to pollution, and are therefore often used as indicators of water quality

(Lenat 1988; Klemm et al. 2003). Lower HBI scores and higher ASPT scores on the West Branch also point to improved water quality and lower organic pollution levels. Organic pollution and heavy metals were not directly measured in this study, although they often affect macroinvertebrate communities (Paul and Meyer 2001; Merritt et al. 2008; Kartikasari et al. 2013). A follow up study that focuses on a wider range of water quality variables could elucidate a broader array of positive impacts of instream restoration activities relative to the macroinvertebrate community.

Despite the lack of data on organic pollutants and heavy metals, the use of macroinvertebrates and biotic indices to assess water quality is well documented (Resh and Jackson 1993; Carter et al. 2006; Kartikasari et al. 2013). Multiple studies have demonstrated that physical restoration of streams leads to positive impacts on the macroinvertebrate community (Spanhoff and Arle 2007; Pander and Geist 2013). However, as Stranko et al. (2012) found, these positive impacts of physical restoration are often more pronounced in rural streams, with urban restored and unrestored streams both having lower EPT richness and sensitive species than rural restored and unrestored streams. Additionally, McDermond-Spies et al. (2014) noted improvements in EPT and biotic indices in rural stream restorations similar in size and climate to the West Branch, while Smith and Lamp (2008) again noted the dichotomy between urban and rural macroinvertebrate communities. In this study, significant positive correlations between measures of sensitive or intolerant taxa (i.e., EPT richness and HBI) with total QHEI as well as substrate quality, channel morphology, and pool-glide/riffle-run quality component scores indicate a positive effect of the restoration on the macroinvertebrate community. Flow was positively correlated with all macroinvertebrate community metrics except for ASPT, which is consistent with previous work showing that the low base flow of urban streams can often be a limiting factor for macroinvertebrate communities (Merritt et al. 2008; Palmer et at. 2010). Though restoration activities on the West Branch have positively impacted the macroinvertebrate community, Purcell et al. (2009) noted that successful restorations in urban streams, while effective at having positive impacts on the biota, are limited in their success by the urban environment. More holistic approaches to restoration that include both instream restoration and riparian zone restoration (such as the restoration undertaken on the West Branch) may be necessary to restore the macroinvertebrate community (Spanhoff and Arle 2007; Stranko et al. 2012).

Aside from serving as indicators of water quality, macroinvertebrates play a vital role in the food web (Kartikasari et al. 2013) and are key to the flow of energy through stream ecosystems (Wallace and Webster 1996). The multiple functional groups of macroinvertebrates, including grazers, shredders, and filter feeders, reduce algae, shred coarse organic material to allow downstream flow, and provide other functions to the stream ecosystem (Wallace and Webster 1996). Functional groups were not investigated further in this study due to the cost of achieving the taxonomic resolution necessary to assess macroinvertebrate functional groups. Additional studies that include pre-restoration data and lower taxonomic levels of identification would allow for a closer look at the effects of the restoration on the macroinvertebrate community.

In stark contrast to the majority of studies looking at urban restorations and the subsequent macroinvertebrate communities, I found that the restored West Branch had a more diverse, more intolerant and pollution sensitive community when compared to the unrestored East Branch. One possible reason for this is the extensive, nearly comprehensive restoration that took place on the West Branch. For example, a restoration of two urban streams in Maryland where less than 1 km of channel was reconstructed and a single wetland was created on one, and 5 km of channel was reconstructed and 150 m of concrete was removed from the stream did not create changes in the macroinvertebrate community when compared to a reference urban stream (Stranko et al. 2012). Additionally, the removal of sediment and repair of a bank over 125 m in a small urbanizing stream in Connecticut did not result in higher macroinvertebrate community metrics (Schiff et al. 2011). These post-project studies were conducted following either a single or few restoration activities conducted over small spatial scales. The restoration of the West Branch included a large number of activities, including the addition of high quality substrate, the alteration of channels and banks, addition of instream habitat and pool-riffle-run complexes, and the addition of wetland areas to control flooding and nutrient inputs to the stream spread over 13 km of river. With all of the macroinvertebrate community metrics other than abundance being higher on the West Branch, and the correlation of these metrics to habitat variable associated with the restoration of the West Branch, I contend that this more comprehensive restoration aided in the recovery of the macroinvertebrate community, which could in turn impact the recovery of stream biota at higher trophic levels, such as fish.

Figures

Figure 3.1. Box plots showing the nine major macroinvertebrate metrics assessed in this study by stream. West Branch is red and the East Branch is blue.

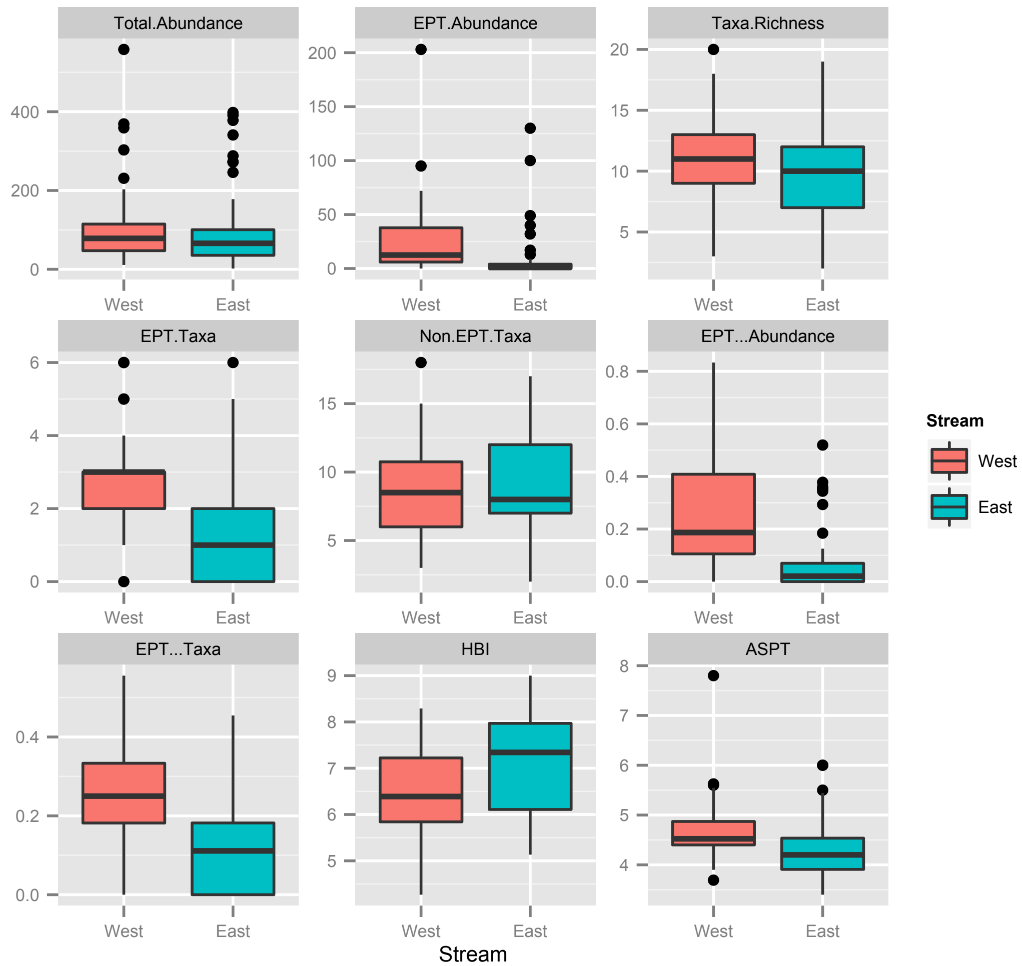


Figure 3.2. Taxonomic richness of benthic macroinvertebrates on the West and East Branches of the DuPage river, with groups from the families Ephemeroptera, Plecoptera, and Tricoptera being separated from the rest of the groups.

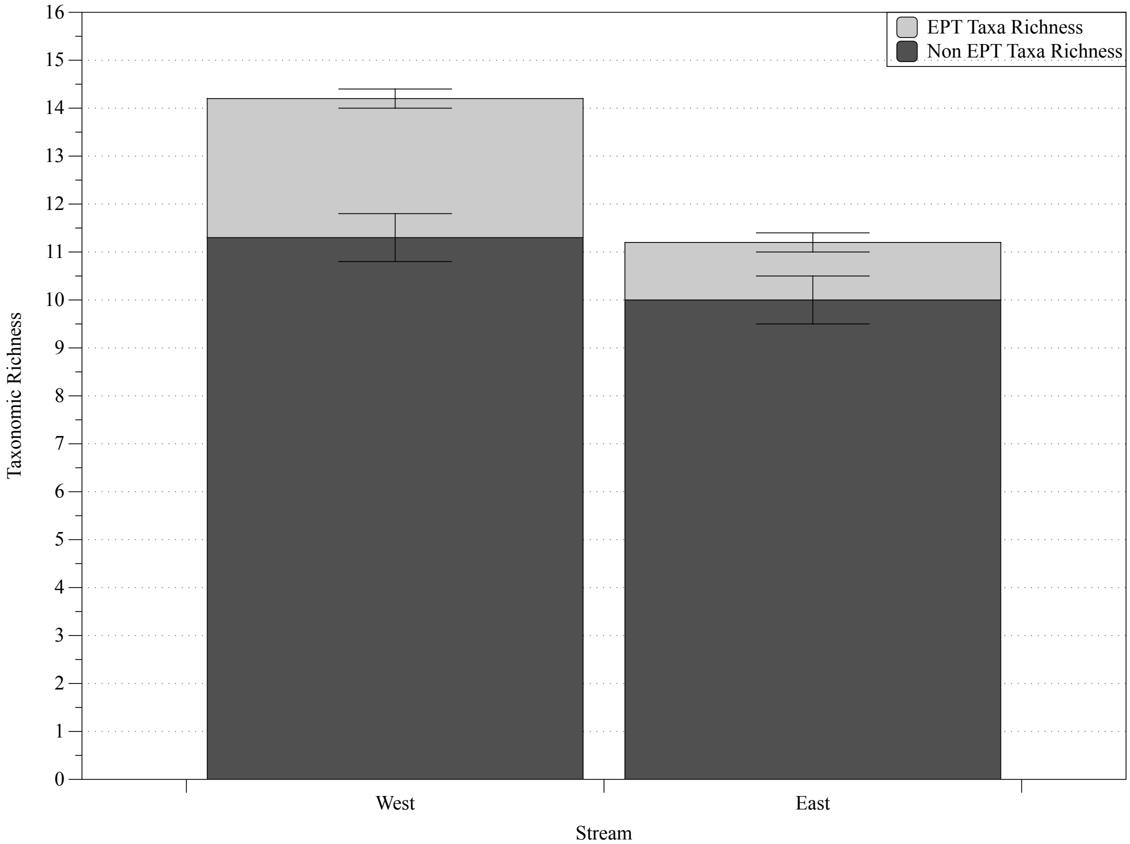
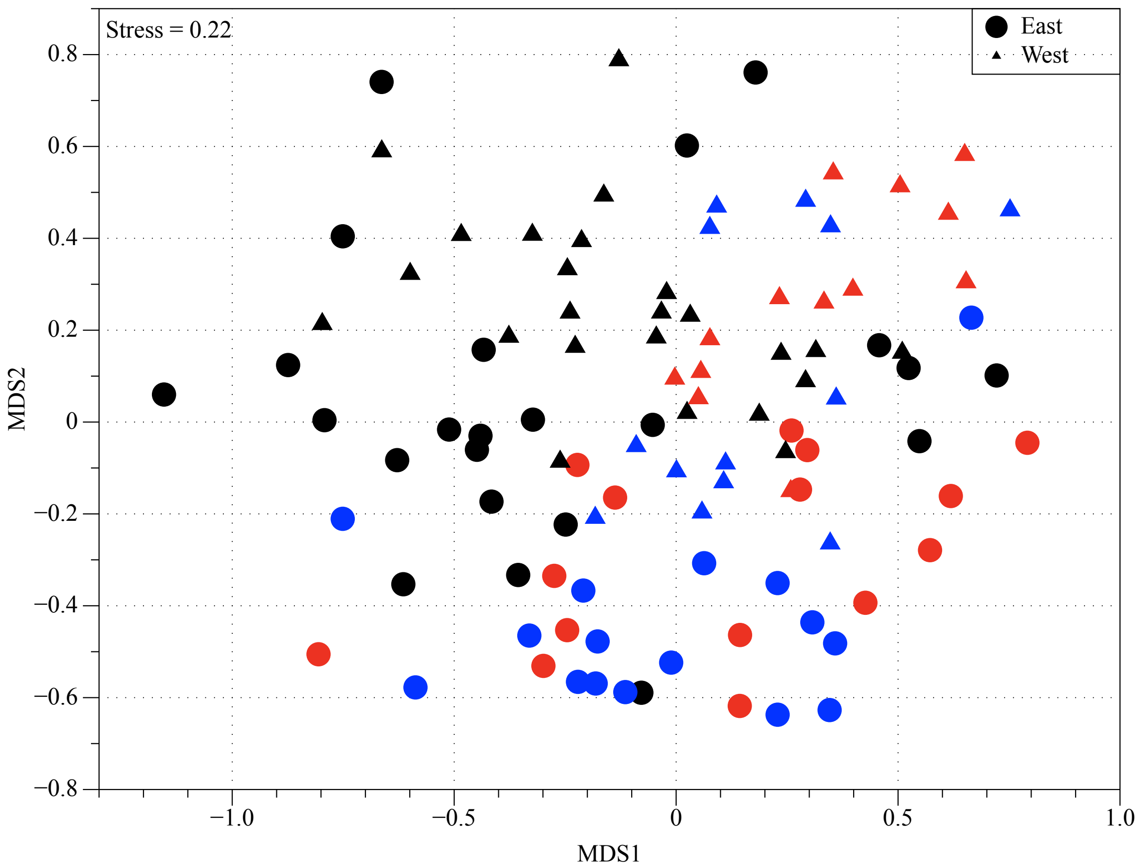


Figure 3.3. Two-dimensional NMDS plot for all macroinvertebrate samples on both the West (square) and East (circular) Branches of the DuPage River. Spring samples are colored blue, summer samples are colored red, and fall samples are colored black.



Tables

Table 3.1. Comparison of total abundance by family and order between the West Branch and East Branch. Where taxonomic identification to order was not possible, the lowest possible classification above order was used.

		West Branch		East Branch	
Order (or lowest)	Family	n	%	n	%
Hydrachnidia		2	<0.1	3	<0.1
Hirudinea		19	0.3	34	0.6
Oligocheata		65	1.2	123	2.2
Tricladida	Planariidae	25	0.5	116	2.1
Isopoda	Asellidae	9	0.2	59	1.1
Decapoda	Cambaridae	8	0.2	20	0.4
Amphipoda	Hyalellidae	1116	20.8	154	2.8
Odonata		737	13.8	1090	20.0
	Aeshnidae	2	<0.1	0	0
	Corduliidae	2	<0.1	0	0
	Libellulidae	3	<0.1	6	0.1
	Calopterygidae	80	1.5	58	1.1
	Coenagrionidae	644	12.0	1026	18.8
	Lestidae	6	0.1	0	0
Ephemeroptera		554	10.3	247	4.5
	Baetidae	48	0.9	34	0.6
	Heptageniidae	7	0.1	16	0.3
	Caenidae	28	0.5	2	<0.1
	Leptohyphidae	450	8.4	195	3.6
	Leptophlebiidae	2	<0.1	0	0
	Potamanthidae	19	0.4	0	0
Plecoptera	Pteronarcidae	1	<0.1	0	0
Tricoptera		691	12.9	210	3.9
	Hydropsychidae	684	12.7	185	3.4
	Hydroptilidae	5	0.1	16	0.3
	Leptoceridae	0	0	9	0.2

Table 3.1 continued.

Order (or lowest)	Family	West Branch n	West Branch %	East Branch n	East Branch %
	Philopomatidae	1	<0.1	0	0
	Limnephilidae	1	0.1	0	0
Coleoptera		84	1.6	127	2.3
	Elmidae	48	0.9	66	1.2
	Curculionidae	0	0	2	<0.1
	Dytiscidae	1	<0.1	22	0.4
	Gyrinidae	4	0.1	1	<0.1
	Haliplidae	28	0.5	34	0.6
	Lampyridae	1	<0.1	2	<0.1
	Tenebrionidae	2	<0.1	0	0
Megaloptera	Corydalidae	0	0	1	<0.1
Diptera		1765	32.9	2696	49.6
	Tipulidae	2	<0.1	1	<0.1
	Ceratopagonidae	1	<0.1	2	<0.1
	Chironomidae	781	14.6	1156	21.3
	Simuliidae	957	17.9	1509	27.8
	Culicidae	19	0.4	17	0.3
	Syrphidae	2	<0.1	0	0
	Stratiomyidae	0	0	7	0.1
	Dixidae	0	0	1	<0.1
	Empididae	3	<0.1	1	<0.1
	Ptychopteridae	0	0	1	<0.1
	Muscidae	0	0	1	<0.1
Hemiptera		105	2.0	210	3.9
	Belostomatidae	5	0.1	4	<0.1
	Corixidae	48	0.9	177	3.3
	Gerridae	6	0.1	6	0.1
	Hebridae	22	0.4	1	<0.1
	Mesoveliidae	10	0.2	3	<0.1
	Naucoridae	0	0	1	<0.1
	Nepidae	3	<0.1	0	0
	Notonectidae	2	<0.1	3	<0.1
	Pleidae	1	<0.1	0	0
	Saldidae	3	<0.1	1	<0.1
	Veliidae	5	0.1	14	0.3

Table 3.1 continued.

Order (or lowest)	Family	West Branch n	West Branch %	East Branch n	East Branch %
Collembola	Poduridae	0	0	7	0.1
Pulmonata		27	0.5	131	2.4
	Physidae	19	0.3	116	2.1
	Planorbidae	8	0.2	15	0.3
Architaenioglossa	Viviparidae	45	0.8	80	1.5
Neotaenioglossa		12	0.2	6	0.1
	Pleuroceridae	12	0.2	5	0.1
	Hydrobiidae	0	0	1	<0.1
Heterostropha	Valvatidae	6	0.1	2	<0.1
Basommatophora	Lymnaeidae	4	<0.1	0	0
Veneroida		63	1.2	118	2.2
	Cyrenidae	40	0.8	52	1.0
	Sphaeridae	23	0.4	66	1.2

Table 3.2. Mean (with standard error) for eight benthic macroinvertebrate variables on the West and East Branches of the DuPage River. T statistic and p-value are also included. Significant p-values for difference between streams are denoted with '*'.

	W. Br. $\bar{x}$ (SE)	E. Br. $\bar{x}$ (SE)	T	P value
Taxonomic richness	11.3 (0.5)	10.0 (0.5)	5.9	< 0.02*
EPT taxa richness	2.9 (0.2)	1.2 (0.2)	50.3	< 0.001*
EPT abundance %	27.2 (3.2)	7.4 (1.7)	40.9	< 0.001*
EPT taxa %	26.3 (1.5)	11.5 (1.6)	45.4	< 0.001*
HBI	6.4 (0.1)	7.1 (0.1)	8.9	< 0.001*
ASPT	4.7 (0.1)	4.3 (0.8)	10.1	< 0.01*

Table 3.3. Spearman rank correlation Rho values with Bonferroni correction (p values in parenthesis) for correlations between macroinvertebrate community metrics and water quality and habitat metrics. Significant correlations are denoted with bold lettering.

	Taxa Richness	EPT Taxa Rich.	EPT % Abund.	EPT % Taxa	HBI	ASPT
Substrate	0.18 (0.06)	**0.54 (<0.001)**	**0.53 (<0.001)**	**0.50 (<0.001)**	**-0.42 (<0.001)**	0.23 (0.24)
Instream Cover	-0.04 (0.69)	0.22 (0.21)	0.30 (0.35)	0.29 (0.30)	-0.31 (0.29)	0.02 (0.82)
Channel Morph	0.11 (0.25)	**0.51 (<0.001)**	**0.53 (<0.001)**	**0.51 (<0.001)**	**-0.42 (<0.001)**	0.19 (0.06)
Bank/Riparian	0.10 (0.28)	0.06 (0.55)	0.01 (0.93)	0.03 (0.74)	-0.05 (0.61)	0.02 (0.87)
Pool/Riffle	0.03 (0.74)	**0.43 (<0.001)**	**0.44 (<0.001)**	**0.46 (<0.001)**	**-0.33 (<0.01)**	0.16 (0.11)
Total QHEI	0.06 (0.52)	**0.55 (<0.001)**	**0.57 (<0.001)**	**0.58 (<0.001)**	**-0.43 (<0.001)**	0.23 (0.24)
Conductivity	**0.40 (<0.001)**	0.03 (0.71)	- 0.05 (0.61)	-0.09 (0.34)	-0.15 (0.12)	-0.23 (0.24)
DO	**-0.35 (<0.001)**	-0.07 (0.49)	-0.05 (0.62)	0.06 (0.53)	0.25 (0.12)	0.18 (0.06)
Temperature	**0.28 (<0.01)**	0.04 (0.67)	0.02 (0.81)	-0.05 (0.59)	-0.12 (0.19)	-0.16 (0.11)
Flow	**0.47 (<0.001)**	**0.45 (<0.001)**	**0.35 (<0.001)**	**0.32 (<0.001)**	**-0.31 (<0.01)**	0.07 (0.50)

CHAPTER 4: FISH

Introduction

Rapid urbanization over the past several decades has degraded the natural resources in the urban environment, especially urban streams (Paul and Meyer 2001). The impacts to urban streams are diverse and far-reaching, and have profound effects on the fish community (Schwartz and Herricks 2007). Many of the impacts to fish communities are indirect, and stem from other impacts of urbanization, such as increased flooding and reduced summer baseflows, chemical inputs from drainage of the urban landscape, reduced instream habitat heterogeneity, and the destruction of natural riparian zones and stream banks (Paul and Meyer 2001).

Abiotic changes to the stream environment induced by urbanization lead to changes to the instream biota, including changes in macrophyte, macroinvertebrate, and fish communities. Fish are dependent on different, often unique habitats during various life stages, and the reduced availability of diverse habitat types can impact fish assemblages. Additionally, changes to water chemistry can reduce the abundance and richness of more sensitive species, and allow tolerant, generalist species to dominate an impacted urban stream. Abiotic changes to urban streams indirectly change fish communities through changes to the macroinvertebrate community, on which many fish species are dependent as a prey source (Stranko et al. 2012). The effects of urbanization on abiotic conditions and the biotic community at lower trophic levels can cause predictable shifts in the urban stream fish community, including reduced biodiversity and shifts towards a more tolerant assemblage of macroinvertebrates and fish (Walsh et al. 2005). Additionally, non-native fish often colonize disturbed environments, leading to further reductions in species richness in urban streams (Paul and Meyer 2001). Changes to fish communities also have negative effects on ecosystem services such as recreation and fishing opportunities, negatively impacting humans living around the urban stream environment (Walsh et al. 2005).

In response to the degradation of the urban streams and impacted fish communities, restoration of the stream environment has become commonplace (Berndardt et al. 2005). Urban stream restorations vary in scale and goals, but many attempt to positively impact the fish community using methods such as the addition of instream habitat, the creation of pool-riffle-run complexes, and the reintroduction of native fish species. Despite the widespread use of restorations to combat the effects of urbanization on streams, there are limited examples of pre- and post-restoration assessment of the effectiveness of the restoration (Small 2012; Stranko et al. 2012). The few existing assessments have yielded mixed results (Bernhardt et al. 2005; Jahnig et al 2011), whether due to unclear statements of the ecological aspects of restoration goals, or to not achieving those goals. Few studies have assessed the effectiveness of urban stream restoration with respect to the recovery of fish communities (Paul and Meyer 2001; Palmer et al 2010). Fish are one of the most commonly studied biota in streams, and because the monitoring of fish can give important information about ecosystem services, as well as a broad range of trophic and ecological niches (Karr 1981), it is important that if goals for restoration include bolstering fish diversity or increasing the numbers of specific species, examination of fish communities post restoration should be more commonplace.

This study assessed the impacts of an extensive, 15 km restoration of the West Branch of the DuPage River, a degraded urban stream near Chicago, IL, using the neighboring East Branch as an unrestored reference stream. I assessed multiple fish community metrics, including species richness, fish abundance, biomass, and Shannon-Weaver Diversity (H'). I also examined two economically and recreationally important sport fish, the Smallmouth bass (*Micropterus dolomieu*) and Largemouth bass (*Micropterus salmoides*) to better understand the impact of urban stream restoration on an ecosystem service important to recreational anglers. I hypothesized that the extensive physical restoration of the West Branch of the DuPage would provide higher quality habitat and more diverse and abundant macroinvertebrate communities to support greater fish species richness, abundance, biomass, and diversity, and specifically support life history stages of sport fish when compared to the unrestored East Branch.

Methods

Study Sites

Fish community sampling was conducted at the same 16 sites on the East and West Branches of the DuPage River described in Chapter 2. The sites had a variety of substrates and habitat types utilized by fish, and all habitat types available within a site were sampled during this study.

Fish Community Sampling

The fish communities at all 16 sites were sampled during the fall of 2013, and the spring, summer, and fall of 2014 and 2015, when stream flow was within 25% of base flow (USGS 2015), except during the summer of 2014 when high water levels and rainfall did not allow for the sampling of sites 5, 6, or 7 on the West Branch. At each site, a 150-foot stretch of the stream was blocked at the upstream and downstream ends with 5mm bar mesh block nets to prevent fish passage into or out of the sampling site. Sampling consisted of two backpack electroshocker operators and at least two more dip netters standing side-by-side beginning at the downstream end of the sample site, and conducting one full circuit from the downstream block net to the upstream block net and back. Each electroshocker was set to 100 volts and 60 hertz, and sampling time for each complete circuit was recorded. All fish captured during sampling, as well as any fish captured in the downstream block net were identified to species, total length was recorded to the nearest mm, and individual weight was recorded to the nearest gram. Once three fish of a particular species were recorded in a 10 mm size class, all others in that size class were enumerated, but not weighed. After lengths and weights were recorded, fish were returned to the stream.

For Smallmouth Bass greater than 200g, a t-bar floy tag with a unique identifying number was inserted into the dorsal musculature, the number recorded and scales were collected for age determination as a part of a separate study. Lengths and weights were recorded for all young-of-the-year and juvenile Smallmouth.

Statistical Analysis

For the following analyses, data collected at all sampling sites on each stream was pooled within each season across all years. Differences between streams and among seasons in relative abundance (fish/hour), species richness, relative biomass (g/hour), Shannon-Weiner diversity, Smallmouth bass relative abundance, and Largemouth bass relative abundance were determined using general linear models with site as a random effect. Linear regression was utilized to determine patterns between the fish community metrics and conductivity, as well as in-stream habitat and macroinvertebrate community measures. Due to the effect of conductivity on capture efficiency, regression was used to test for differences between the streams in fish community metrics using only fall samples.

Nonmetric multidimensional scaling (NMDS) using Bray-Curtis distance matrices was used to elucidate patterns in fish relative abundance across sites and species. Relative abundance data for each site was transformed (log[x+1]) prior to ordination (Jackson 1993; Pond et al. 2008). The NMDS ordination was then tested for site and species groups with significant differences between streams using analysis of similarity (ANOSIM), and similarity percentage (SIMPER) analysis was used to identify fish species making significant contributions to dissimilarities between the two streams. All analyses were performed using the R statistical platform.

Fish Guild Structure

Captured fish species were identified as belonging to one of four reproductive guilds: complex spawners with parental care, complex spawners without parental care, simple spawners that utilize a rocky or gravel substrate, and simple spawners that do not need a specific substrate (Balon 1975). Captured fish species were also placed into four feeding guilds: omnivores, insectivores, benthic insectivores, and insectivore/piscivores (Berkman and Rabeni 1987) based on life history information from Cross and Collins (1995). Berkman and Rabeni (1987) also identified an herbivore guild, but with only one strictly herbivorous species captured during this study, the herbivore guild was omitted from fish feeding guild analyses. ANOSIM was used to elucidate patterns and to determine differences between the streams with respect to reproductive and feeding guild structure.

Results

Throughout the seven sampling seasons in this study, a total of 14,661 fish representing 39 species and 10 families were captured (Table 4.1). Centrarchidae dominated the overall catch on both the West (mean = 69.3%, SE = 3.1%) and East Branches (mean = 48.1%, SE = 3.0%). Cyprinidae was the second most abundant family on both branches, making up 15.8% of the catch on the West Branch (SE = 2.2%) and 35.6% of the catch on the East Branch (SE = 2.6%). Catostomidae and Ictaluridae also made up approximately 5-10% each of the catch on both branches, with species from the families Fundulidae, Percidae, Clupidae, Poeciliidae, Gobiidae, and Umbdridae also being captured during this study (Figure 4.1).

Green sunfish (*Lepomis* cyanellus) was the most common fish species on both streams, making up 55.9% of the catch on the West Branch, but only 25.8% of the catch on the East Branch. White sucker (*Catostomus commersonii*) was the second most common fish on the West Branch (7.4%), followed by Bluegill (*Lepomis macrochirus*, 6.2%), Bluntnose minnow (*Pimephales notatus*, 5.3%), and Yellow bullhead (*Ameiurus natalis*, 4.1%). Bluntnose minnow was the second most common species on the East Branch (12.8%), followed by Bluegill (10.8%), Sand shiner (*Notropis stramineus*, 9.5%), and white sucker (6.3%). During the course of our study, we captured 11 invasive Round gobies (*Neogobius melanostomus*) at two sites on the East Branch, the first record of the invasive species in the DuPage River system.

Overall species richness on the East Branch (mean = 12.0, SE = 0.4) did not differ from the West Branch (mean = 9.9, SE = 0.4; $t = 1.7$; $p = 0.10$; Figure 4.2). However, species richness differed significantly by season, with fall samples having significantly higher species richness than spring and summer samples ($p < 0.001$), and summer samples having significantly lower species richness when compared to spring samples ($p = 0.04$).

Relative abundance (fish caught per hour of sampling) of all species combined did not differ significantly ($t = 0.3$, $p = 0.80$) between the West Branch (mean = 296.4, SE = 22.4) and East Branch (mean = 310.5, 34.9). Seasonality did affect relative abundance, however, with spring and summer samples both having significantly lower relative abundance than fall samples ($p < 0.001$; Figure 4.3).

Relative biomass (g/hr) on the East Branch (mean = 14334.7, SE = 2296.6) did not differ significantly ($t = 1.2$; $p = 0.33$) when compared to the West Branch (mean = 6313.8, SE = 1079.1) likely due to high variability by site. Seasonality was also not a significant factor in variation in relative biomass (Figure 4.4). In many of the samples, several large common carp (*Cyprinus carpio*) were captured, which heavily influenced the relative biomass data. Mean proportion of relative biomass of Common carp was 29.6% (SE = 3.6%) across all samples, but comprised only 2.7% (SE = 0.1%) of the relative abundance of the samples. Relative biomass excluding common carp (West Branch mean = 3226.2, SE = 303.0; East Branch mean = 3582.7, SE = 307.0) did not differ significantly between streams ($t = 0.7$; $p = 0.48$), but differed among seasons, with summer and spring not differing significantly ($p = 0.7$), but fall having significantly higher relative biomass than either ($p < 0.001$; Figure 4.5).

Shannon-Weaver Diversity Index (H') was higher on the East Branch (mean H' = 1.9; SE = 0.1) compared to the West Branch (mean H' = 1.4; SE = 0.1) indicating the fish community was significantly more diverse on the East Branch than the West Branch ($t = 2.8$; $p = 0.02$; Figure 4.6). Seasonality did not significantly affect the Shannon-Weaver diversity scores.

Conductivity and Capture Efficiency

The fact that all metrics but Shannon-Weaver diversity differed significantly by season, may indicate lower gear capture efficiency in the spring and summer due to substantially higher conductivity. Conductivity ranged in this study ranged between 678 – 1486 mS/cm (mean = 1022, SE = 17.3), which are considered high conductivities for lotic water bodies (Reynolds 1996). Conductivities in fall samples (mean = 868.4, SE = 12.7) when compared to spring (mean = 1236.3, SE = 21.0) and summer (mean = 1041.3, SE = 11.1) were significantly lower ($F = 154.0$; df = 2, 106; $p < 0.001$). Further, fall conductivity was within/much closer to/ ranges considered optimal for electofishing (Reynolds 1996).

Based on a linear regression of species richness against conductivity, high conductivity resulted in significantly lower species richness on the West Branch ($F = 6.7$; $p = 0.01$; adj. $R^2 = 0.10$), but had no impact on species richness on the East Branch ($F = 2.2$; $p = 0.15$). Shannon Weaver diversity index was not significantly influenced by conductivity on either West Branch ($F = 0.01$; $p = 0.92$) or East Branch ($F = 0.1$; $p = 0.71$). Regression analysis revealed that high conductivity also resulted in significantly lower relative abundance (fish/hour of electrofishing time) on both the West Branch ($F = 20.7$; $p < 0.001$; $R^2 = 0.27$), and East Branch ($F = 16.6$; $p = 0.001$; $R^2 = 0.22$; Figure 4.7).

Additionally, biomass (g/hr) was also significantly negatively influenced by conductivity on both the West ($F = 5.7$, $p = 0.02$, adj. $R^2 = 0.08$) and East Branches ($F = 17.0$; $p < 0.001$; adj. $R^2 = 0.23$; Figure 4.8). The negative relationship between conductivity, relative abundance and relative biomass without corresponding negative relationships with diversity indicates reduced gear efficiency in the spring and summer samples rather than substantive seasonal changes to the fish community. Subsequent analyses of fish community diversity and abundance, as well as evaluations of sport fish species are based on fall samples only.

Fall Fish Community Assessment

Data from fall samples collected at all eight sites between 2013-2015 was used in a series of general linear models to determine differences in fish diversity and abundance between streams, with sites as a random effect. There were no significant differences between the East Branch and the West Branch with respect to fall species richness ($t = 1.3$; $p = 0.20$), Shannon-Weaver diversity index ($t = 1.2$; $p = 0.24$), relative abundance ($t = 0.5$; $p = 0.61$) or relative biomass ($t = 0.9$; $p = 0.38$; Table 4.2). Differences between streams with respect to the fish community structure were further examined using non-metric multidimensional scaling (NMDS), revealing separation between the communities of the West Branch and East Branch in multidimensional space based on relative abundance of individual species (Figure 4.10). Analysis of similarity (ANOSIM) revealed a significant difference in fish community structure between the two streams (ANOSIM $R = 0.38$, $p < 0.001$), and similarity percentage (SIMPER) analysis showed that Smallmouth Bass relative abundance contributed the most to dissimilarity between the West and East Branches, explaining 7.7% of the difference, with Bluntnose Minnow

(*Pimephales notatus*) explaining 7.3% and Bluegill (*Lepomis macrochirus*) explaining 6.9%. Smallmouth Bass were more abundant on the West Branch and both Bluntnose minnows and Bluegill were more abundant on the East Branch (Table 4.1). Relative abundance of Smallmouth bass was over 20 times greater on the West Branch ($t = -4.3$; $p < 0.001$) than on the East Branch.

Sportfish

The two most common sport fish caught during this study were Smallmouth bass (*Micropterus dolomeiu*), which comprised 2.9% of total catch across and 4.2% of relative biomass across both streams, and Largemouth bass (*Micropterus salmonoides*), which comprised 4.0% of total catch and 4.8% of relative biomass. Largemouth were less abundant on the West Branch, comprising 3.3% of the catch compared 4.7% on the East. In contrast, Smallmouth comprised 5.5% of the catch on the West Branch and only 0.1% on the East Branch. Largemouth comprised 4.4% of the relative biomass on the West Branch and 5.2% of the relative biomass on the East Branch, whereas Smallmouth comprised 7.2% of the relative biomass on the West Branch while only making up 1.2% of the relative biomass on the East Branch.

Smallmouth bass caught in this study ranged from 41 to 209 mm on the West Branch ($n = 207$; mean = 95.7; SE = 1.5), and from 102 to 254 mm on the East Branch ($n = 10$; mean = 139; SE = 17.5). Largemouth bass on the West ranged from 62 to 165 mm ($n = 102$; mean = 90.9; SE = 2.0), and from 60 to 260 mm ($n = 112$; mean = 102.1; SE = 3.5) on the East Branch. Black Bass in both streams were predominantly younger fish with few adult sized individuals represented (Figure 4.11) based on known size at age for both Smallmouth Bass (Baylis et al 1993, Phelps et al. 2008) and Largemouth bass

(Reiser et al 2004). Smallmouth relative abundance ($t = -4.3$; $p < 0.001$) and relative biomass ($t = -2.9$; $p = 0.01$) were both significantly higher on the West Branch than the East Branch. Conversely, Largemouth relative abundance did not differ significantly between the West and East Branches ($t = 0.3$; $p = 0.75$), and neither did Largemouth relative biomass ($t = 0.8$; $p = 0.41$).

Regression was used to assess the effect of habitat and macroinvertebrate community metrics on Smallmouth relative abundance. Due to the very small number of Smallmouth caught on the East Branch (n = 10, CPUE = 0.9 fish/hr), only West Branch Smallmouth data was used in this analysis. Among the six habitat metrics that comprise the QHEI score, none significantly correlated to Smallmouth Bass biomass. Additionally, relative abundance did not have a significant relationship to any of the macroinvertebrate community metrics assessed in Chapter 3 (Table 4.3). Analysis of reproductive and feeding guilds did not elucidate any patterns in fish assemblages that were not apparent from fish community ordination. ANOSIM revealed no significant differences between streams with respect to reproductive guild structure ($R = 0$, $p = 0.58$) or feeding guild structure ($R = 0.11$, $p = 0.11$) of the fish community. Additionally, the analysis of both feeding and reproductive guilds combined did not detect significant differences between the branches ($R = 0.03$, $p = 0.28$).

Discussion

Though stream restorations are undertaken for a multitude of reasons, the vast majority are either explicitly or implicitly designed with the improvement of instream habitat and stream health for fish communities in mind (Bernhardt and Palmer 2011).

Diverse fish communities inhabit a multitude of ecological and physical niches requiring habitat heterogeneity (Balon 1975; Berkman and Rabeni 1987), which is often reduced by the hydrology of urban streams making habitat heterogeneity a focus of urban stream restorations (Bernhardt et al 2005; Bernhardt and Palmer 2011). Restored streams are expected to have higher habitat heterogeneity as well as more resilience to physical and hydrological stress, which in turn is expected to benefit stream biota, including fish and macroinvertebrates. In theory, higher habitat quality and heterogeneity will increase availability of prey items and provide refuge for fish, supporting a more diverse, abundant fish community within restored reaches. I predicted restoration of the West Branch of the DuPage that increased quality and quantity of heterogeneous habitats would result in higher densities of fish, would increase available niches resulting in for higher fish species diversity, and increase availability of a macroinvertebrate and fish forage base that could support greater sport fish biomass. I specifically compared the relative abundance, relative biomass, species richness, and species diversity of the fish communities within the West Branch and East Branch of the DuPage River, and compared the relative abundance and relative biomass of Smallmouth Bass and Largemouth Bass on the two streams.

In this study, I found that neither the relative abundance nor the relative biomass of fish differed between the West and East Branches. These findings could indicate that while the physical habitat on the West Branch is of higher quality than on the East Branch, there are other factors limiting fish abundance, possibly including prey abundance, water quality, or ability to move throughout the stream. This is in contrast to Selego et al. (2012) and Hockendorff et al. (2017) who both found an increase in fish abundance after restoration of riparian zone and instream habitat including riffle-run complexes, stream channel, and substrate. Multiple other studies, however, have shown that habitat restoration of instream habitats had insignificant or no effect on abundance or biomass of fish (Jahnig et al. 2011; Nilsson et al. 2015). There are several possible reasons why the abundance and biomass of the fish community has not responded to restoration. Unlike Hockendorff et al., who sampled for 15 years post-restoration, and saw increases in fish metrics throughout, I sampled shortly after the finish of the restoration, possibly not allowing time for full recolonization. Additionally, the highly urbanized landscape around the West Branch makes habitat restoration only one piece of the solution to degradation of the stream (Bernhardt and Palmer 2011), with other, watershed scale impairments potentially causing problems.

Black bass are the most commonly targeted sport fish in recreational freshwater fisheries in the US (USFWS 2011). Increasing density of human populations in large cities has created the need to provide improved fishing opportunities in urban areas (Balsman and Shoup 2008), which was an explicitly stated goal of the restoration of the West Branch (Burke 2006). Although I found no differences between streams in the relative abundance and relative biomass of Largemouth Bass, Smallmouth Bass were significantly more abundant and comprised greater fish biomass on the West Branch. Largemouth Bass better tolerate poor water quality and low quality substrate when compared to Smallmouth (Grabarkiewicz and Davis 2008), which likely explains the similarity in the populations of Largemouth Bass on both streams. The vast majority of Smallmouth Bass captured in this study were young of the year or juveniles, indicating that adults may utilize restored instream habitat on the West Branch for reproduction and as a nursery for juveniles. Young of the year Smallmouth Bass commonly eat benthic macroinvertebrates, most notably Chironomids, Isopods, and Ephemeropterans (Brown et al. 2009), of which the last two were significantly more abundant on the West Branch compared to the East Branch (Chapter 3). As YOY Smallmouth Bass grow, they undergo an ontogenetic diet shift (Easton and Orth 1992), moving towards larger invertebrates and fish, which were abundant on the West Branch as well. Restoration activities in the West Branch included instream habitat features adult Smallmouth are known to prefer for spawning, including more coarse (> 3 cm) substrates and areas with cover such as large boulders (Brown et al 2009; Brown and Bozek 2010). Two habitat variables in particular where the lack of correlation between West Branch sites and the habitat variable was surprising were substrate quality and chnnel morphology, since both are considered key factors in Smallmouth habitat selection (Sowa and Rabeni 1995; Fayram et al. 2014). Because a majority of the Smallmouth Bass collected in this study were YOY or juveniles, our results are consistent with Brewer and Rabeni (2011) who found that instream metrics were not of particular importance to YOY Smallmouth habitat occupancy, and instead found that land use was a driver of Smallmouth abundance and biomass. Sowa and Rabeni (1995) additionally found that while physical habitat is considered important for the creation of a healthy Smallmouth population, there are a multitude of other factors, including land use, sediment, flow, and others which also impact Smallmouth Bass populations. Differences between Smallmouth Bass populations on the East Branch and West Branch are, therefore, likely the result of factors other than instream habitat quality.

Biodiversity was not higher on the West Branch in comparison to the East Branch, with neither species richness nor Shannon-Weaver Diversity indicating differences between the streams. While the fish community metrics assessed did not indicate any distinction between the restored West Branch and the reference East Branch, there was a strong shift in fish community assemblage between the branches, with the West Branch shifting from Cyprinids to Centrarchids as the dominant family in the stream. Stranko et al. (2012) found that a wide array of fish biodiversity metrics in urban restored streams did not differ from unrestored, reference urban streams, including species richness. Additionally, in a review of post-restoration evaluations on 24 rivers, Haase et al (2013) found that less than half of the restorations increased fish species richness, Shannon-Weaver Diversity, and community assemblage similarity when compared to reference streams. Arango et al., (2013) found that urban stream restoration created a short-term change in biodiversity, but within a year the studied community had shifted back towards the pre-restoration state. Interestingly, the stream studied by Arango et al., (2013) was similar to the West Branch as it had similar barriers to recolonization including a dam directly downstream of the restoration.

The relative abundance of fish species was higher on the East Branch, and differences in relative abundance of Smallmouth Bass (higher on West Branch), Bluntnose Minnow, and Bluegill (both higher on the East Branch) were the strongest drivers of overall differences in relative abundance between the streams. Other studies have utilized ordination to demonstrate that urbanization and associated impairments play a role in fish assemblages (Helms et al 2005; Kennen et al 2005). While these studies did not include all of the same species, several similar species of Cyprinids and Centrarchids were noted as significantly changed by urbanization. While the restoration has provided improvements to instream habitat on the West Branch relative to the East Branch, the changes in relative abundance in the West Branch have been limited, though an increase in Green Sunfish, and decreases in Bluegill and Bluntnose Minnow were noted. The differing sampling methods between pre- and post-restoration assessments limit assessment of size class differences. Though there have been some positive results of the restoration, it is important to note that physical restoration does not improve landscape wide effects of urbanization, which may also need to be addressed.

Limited evidence indicating habitat improvements on the West Branch resulted in large-scale changes to the fish community, including higher relative abundances of Smallmouth, more Centrarchids, and fewer Cyprinids on the West Branch relative to the East Branch, indicate that instream improvements alone may limit the success of an urban stream restoration. Dams are major impediments to upstream and downstream movement, and may not allow repopulation of areas upstream of the dam (Gillette et al. 2005; Alexander and Allan 2006; Hansen and Hayes 2012). In the case of the West Branch, one potential roadblock to fish community recovery post restoration is the presence of Fawell Dam just south of McDowell Grove Forest Preserve. Another major challenge to successful urban stream restoration is that restorations tend to be centered on instream and/or riparian zone improvements, and tend to ignore landscape-scale activity throughout an urban watershed that negatively impact stream health. Expansive impervious surfaces and ineffective storm water management can result in altered hydrology of urban streams, making it difficult for instream improvements to result in benefits to fish communities (Helms et al. 2005; Kennen et al 2005; Bernhardt and

Palmer 2007). These landscape-scale effects would require management far beyond the stream scale, and are very difficult to control in an urban or urbanizing environment.

In the case of the West Branch, continued monitoring of both the fish community, but also the macroinvertebrate community, water quality, and instream habitat is crucial to understanding the long-term effects of this substantial stream restoration. Consistent, assessments over time can provide scientists and managers with insights into the long-term viability of instream improvements and their effects on the fish community over time. I detected some evidence that instream restoration led to changes in the fish community and is supporting a thriving Smallmouth Bass population, but with the persistent effects of continued urbanization and degradation of natural waterways, long-term monitoring these streams is necessary to evaluate the effect of this restoration over longer time scales.

Figures

Figure 4.1. Proportions of fish caught during each sampling season on both the West and East Branches of the DuPage River, grouped by family. Other refers to fish from the families Fundulidae, Percidae, Clupidae, Poeciliidae, Gobiidae, and Umbdridae.

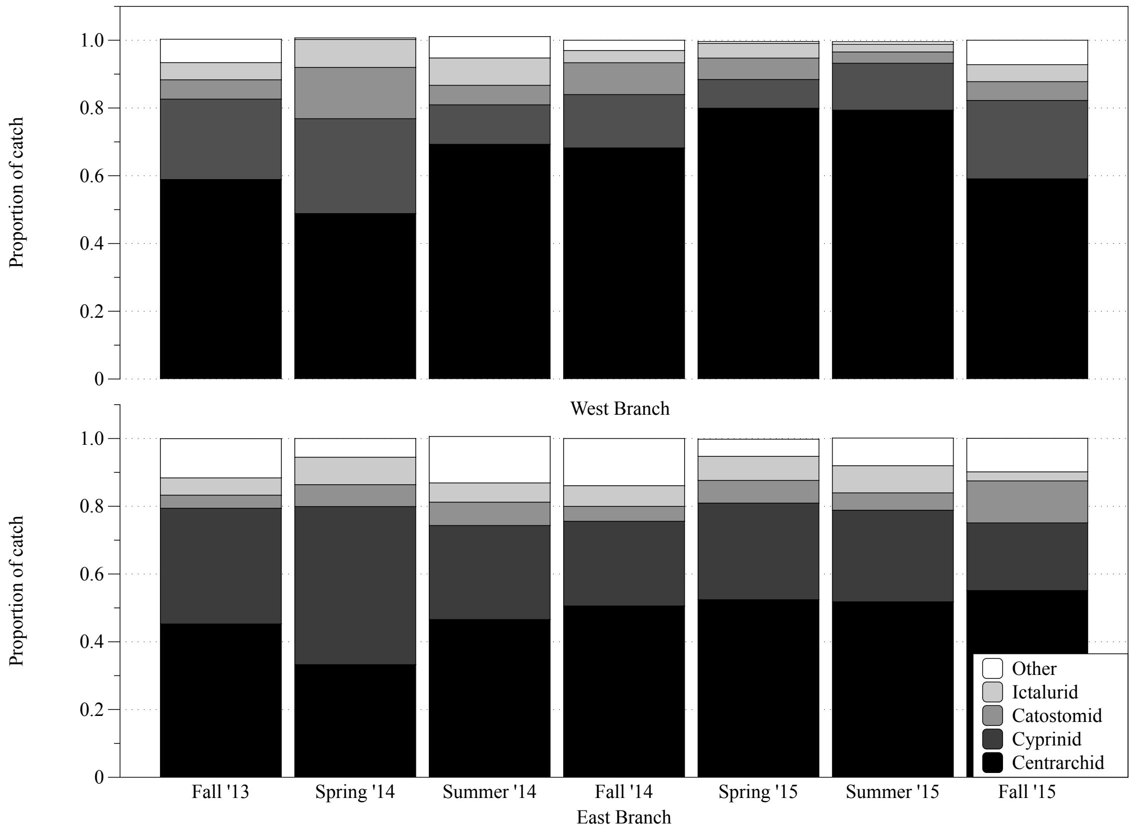


Figure 4.2. Comparison of mean species richness with standard error bars across seasons for West Branch (gray) and East Branch (white) of the DuPage River.

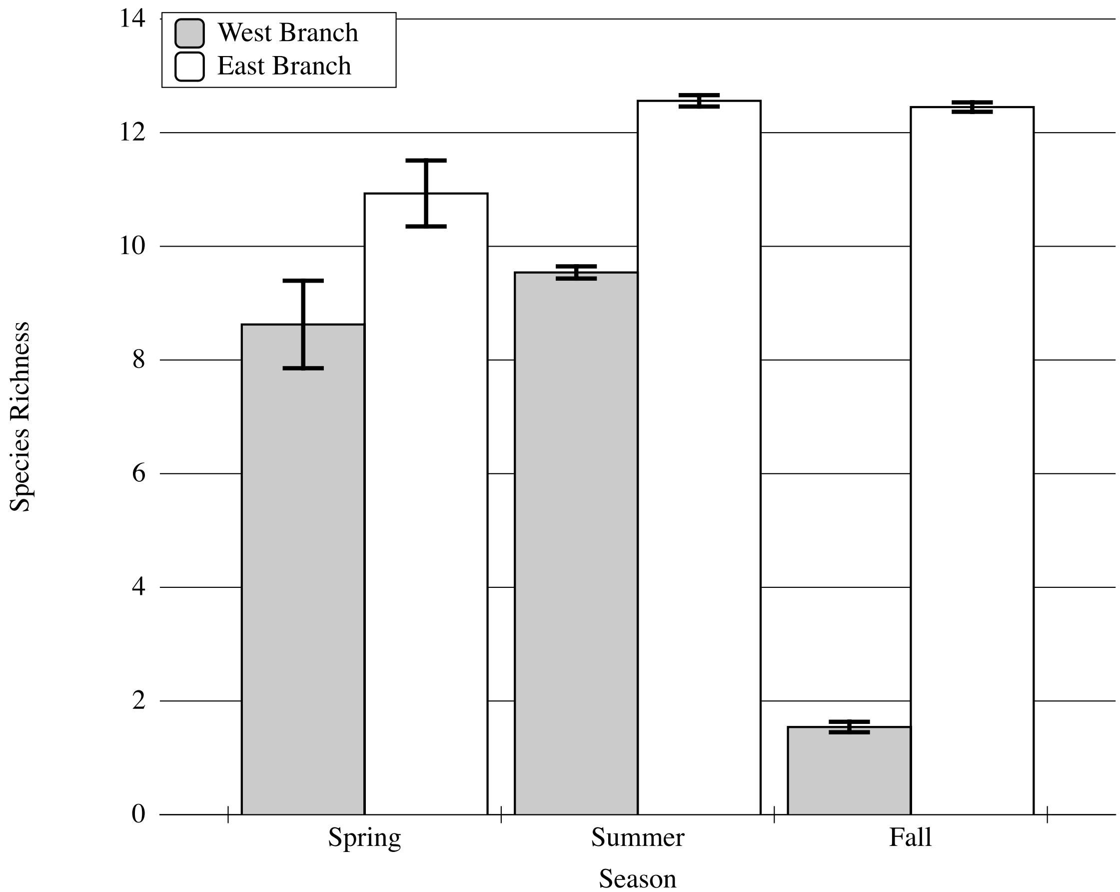


Figure 4.3. Mean relative abundance (fish/hr) with standard error bars for all seasons for West Branch (gray) and East Branch (white) of the DuPage River.

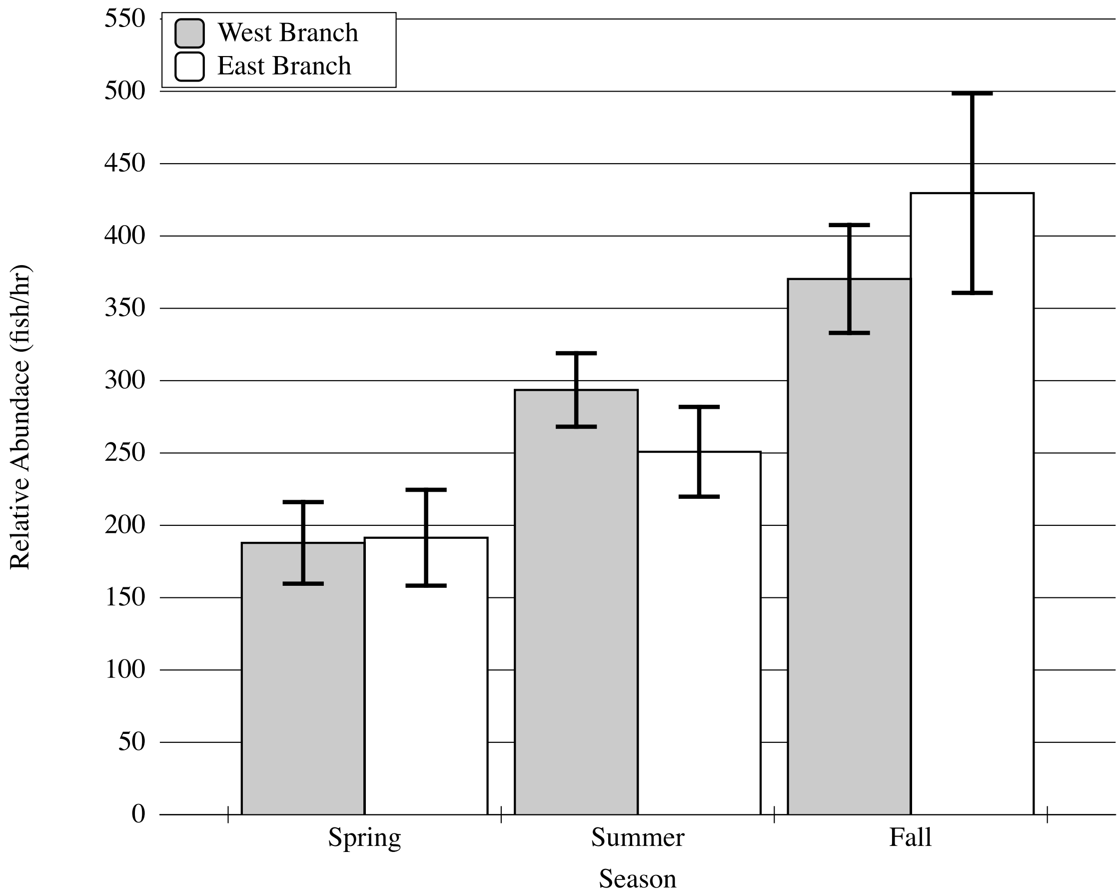


Figure 4.4. Mean relative biomass (g/hr) with standard error bars for all seasons for West Branch (gray) and East Branch (white) of the DuPage River.

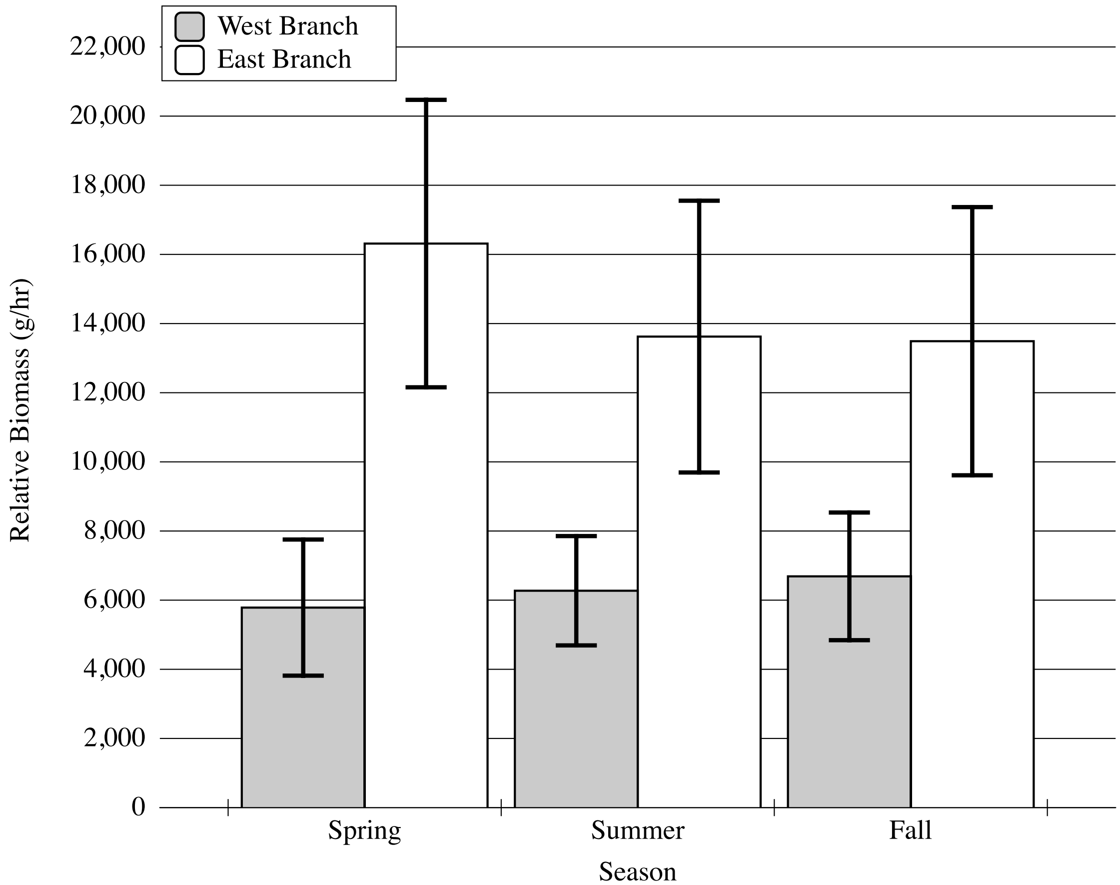


Figure 4.5. Mean relative abundance (fish/hr) with standard error bars for all seasons for West Branch (gray) and East Branch (white) of the DuPage River, excluding Common carp from all samples.

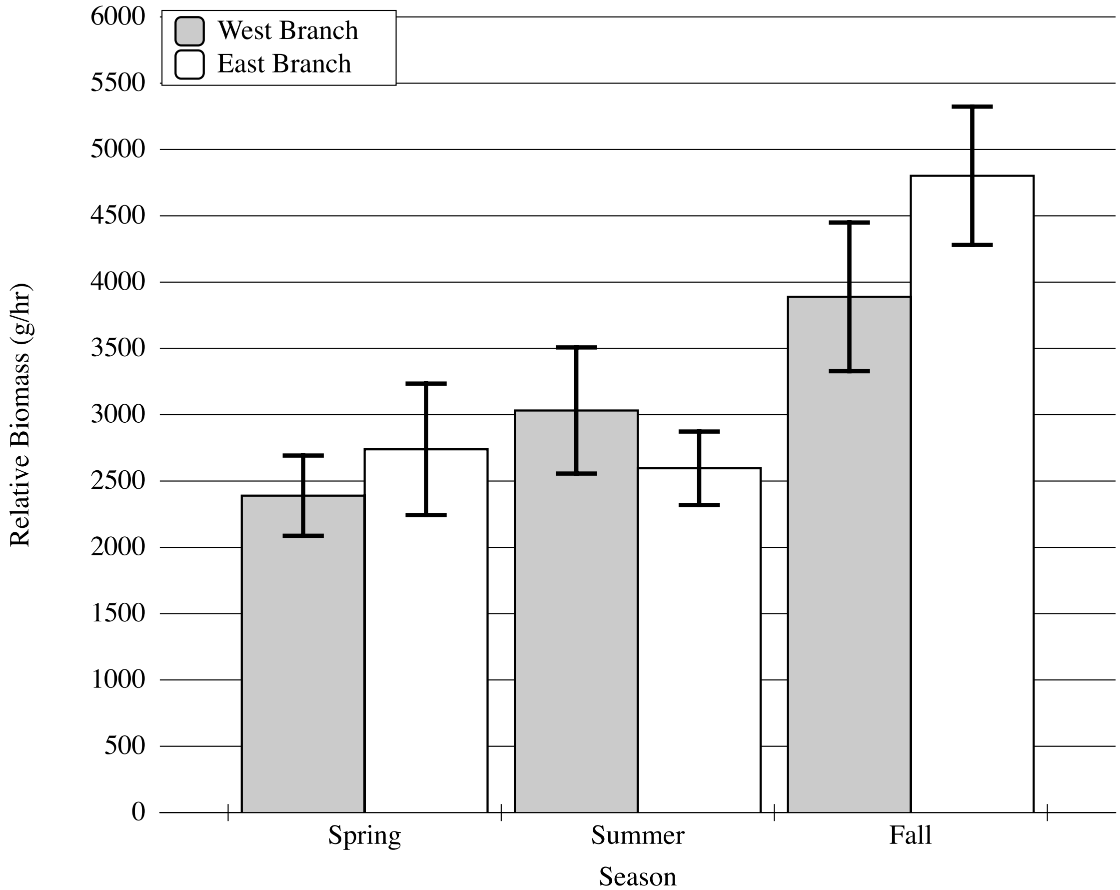


Figure 4.6. Shannon-Weaver Diversity scores for the West and East Branches of the DuPage River.

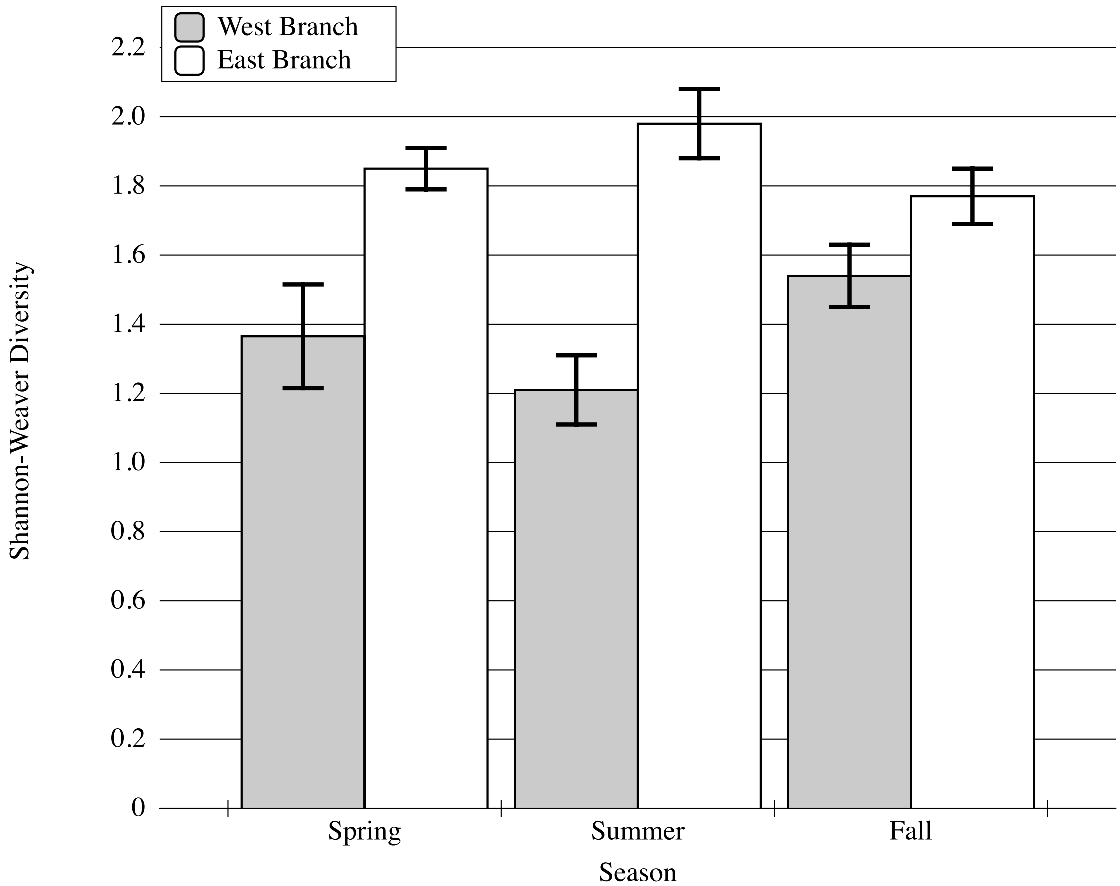


Figure 4.7. Linear relationship between conductivity and relative abundance (fish/hr) on both the West and East Branches of the DuPage River.

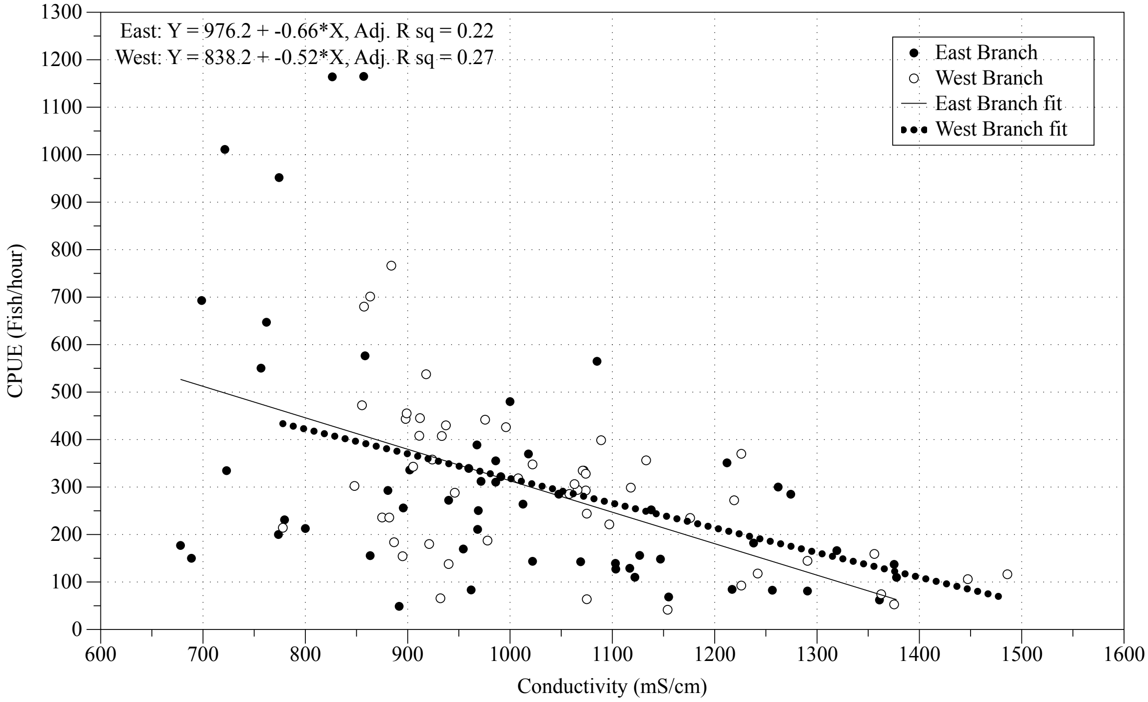


Figure 4.8. Linear relationship between conductivity and relative biomass (g/hr) of fish on both the West and East Branches of the DuPage River.

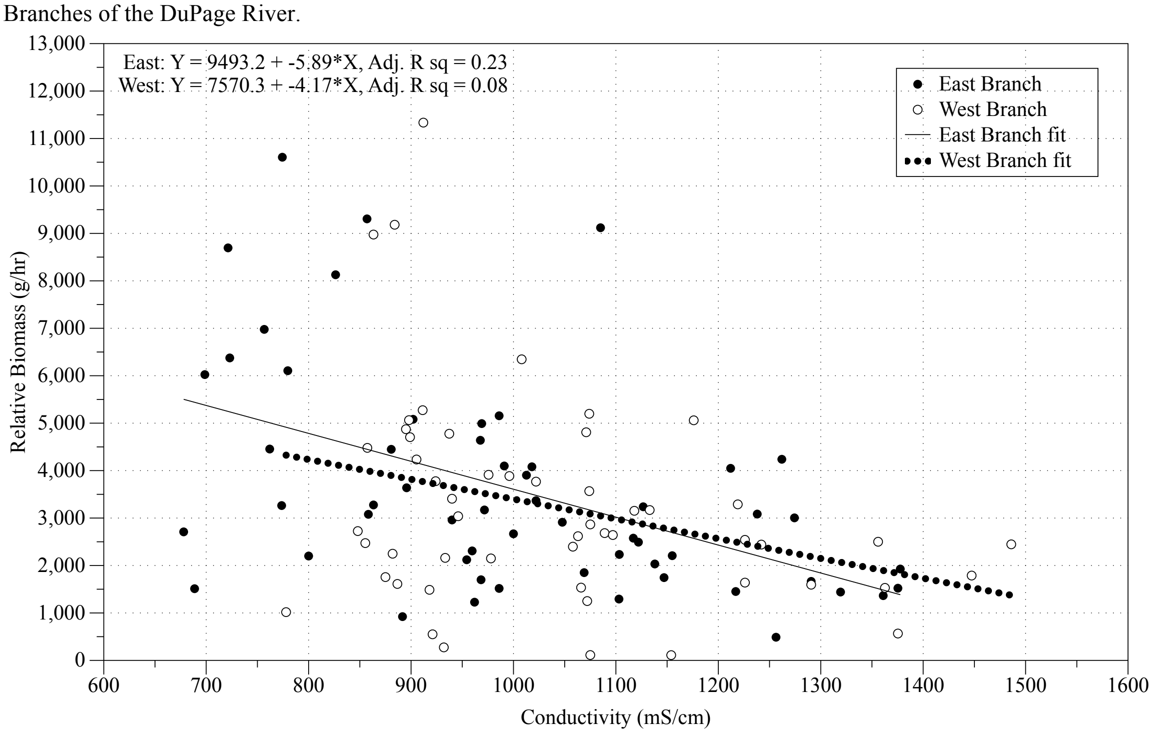


Figure 4.9. Fall fish community breakdown by stream for both the West and East Branches of the DuPage River.

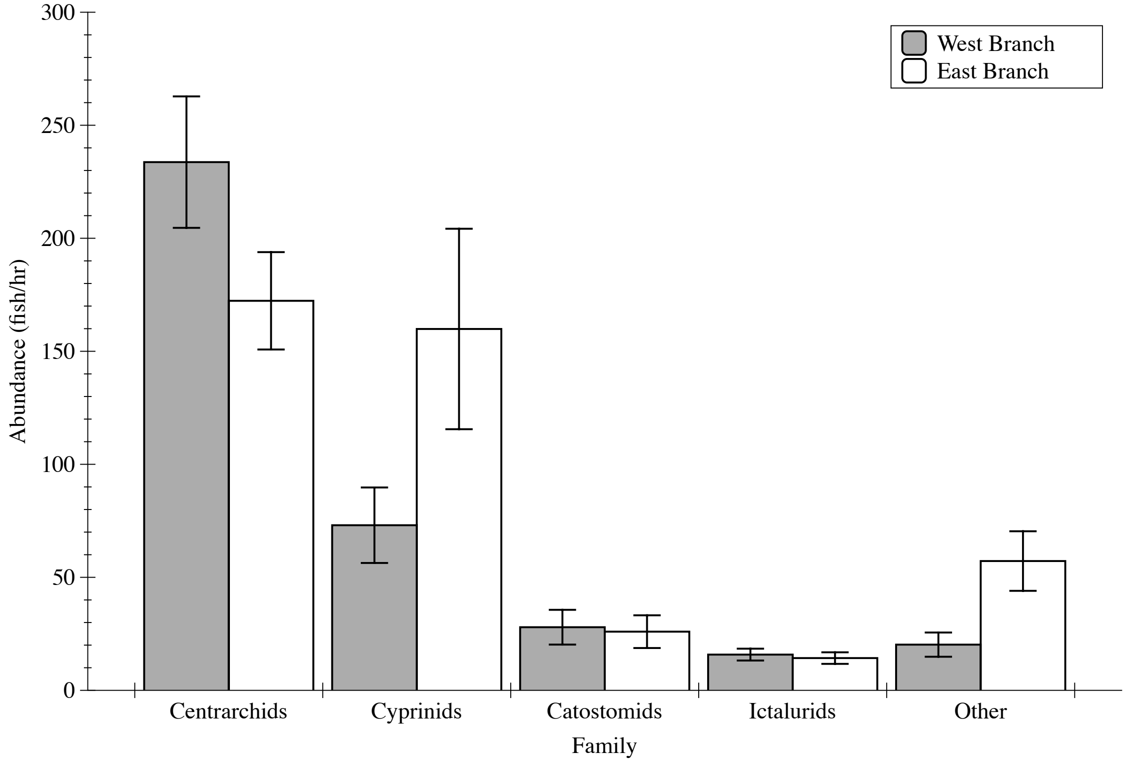


Figure 4.10. Non-metric Multidimensional Scaling (NMDS) plot of relative abundance (CPUE) of fish communities on the West and East Branches of the DuPage River utilizing a Bray-Curtis dissimilarity matrix. Filled in triangles correspond to West Branch sites and open circles correspond to East Branch sites.

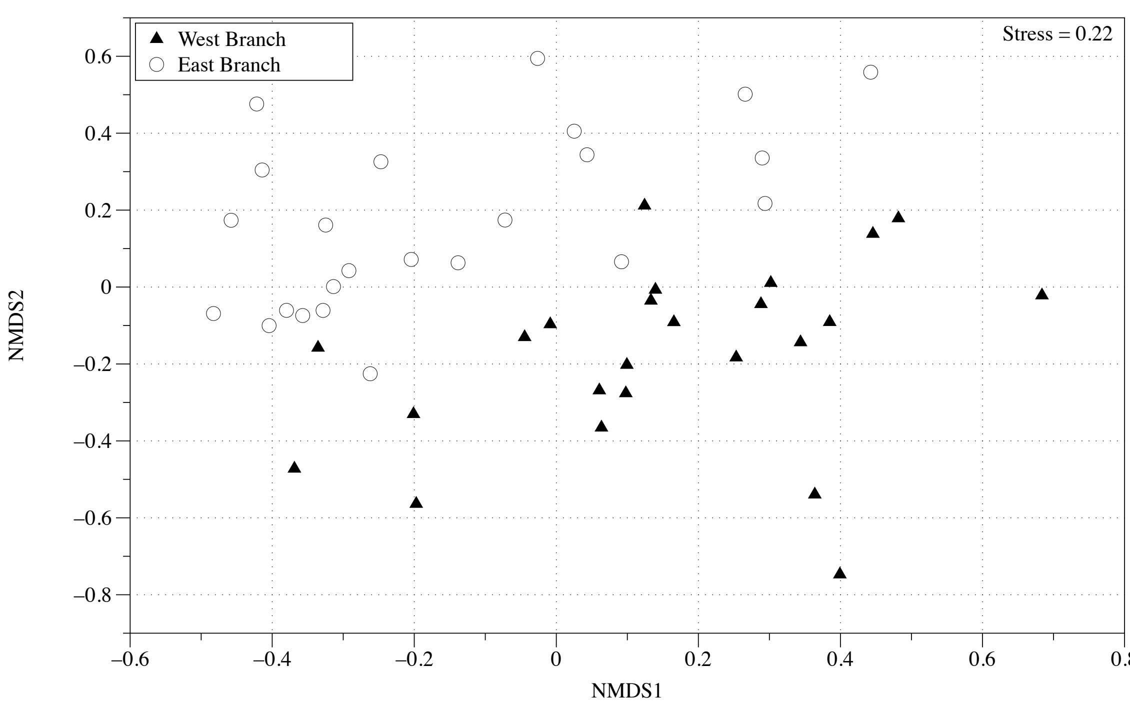
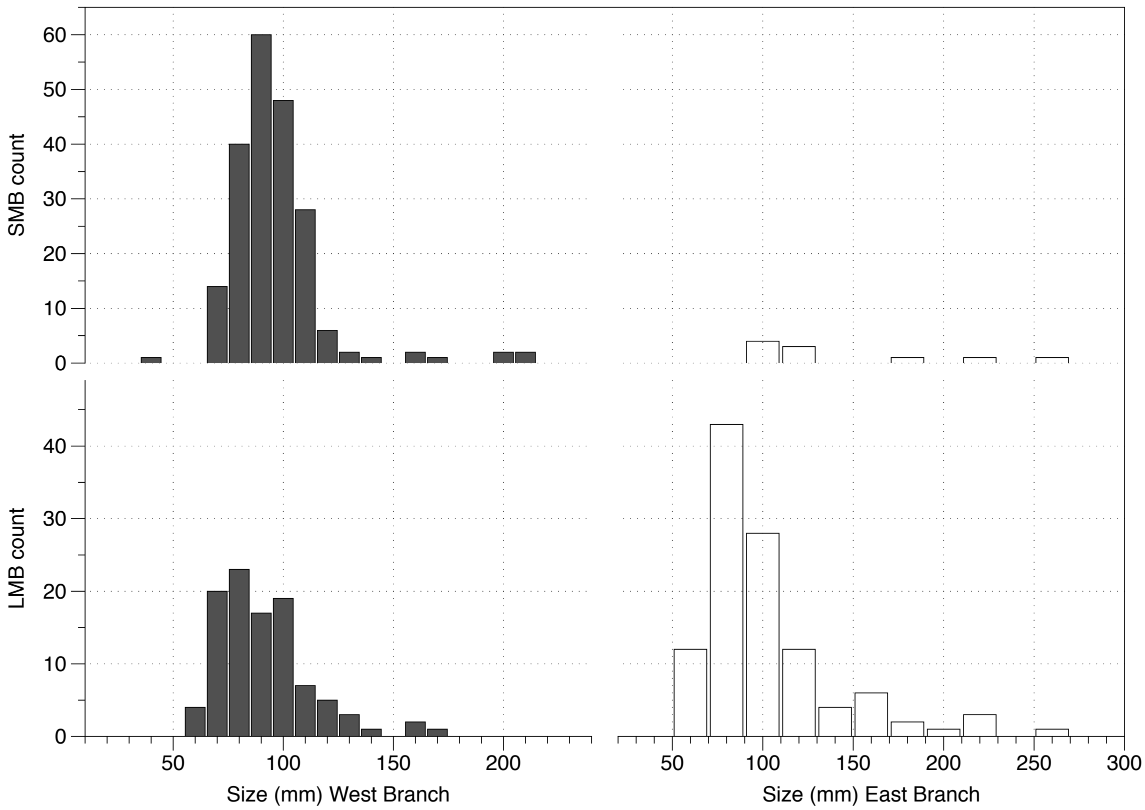


Figure 4.11. Size class data for both Smallmouth bass (top) and Largemouth bass on both the West Branch (dark gray) and East Branch of the DuPage River (white).



Tables

Table 4.1. Proportion of all fish caught on the West and East Branches of the DuPage River by family and species, based on relative abundance.

Family	Species	West Branch	East Branch
Centrarchidae		69.3%	41.8%
	Micropterus dolomieu	3.3%	0.1%
	Micropterus salmonoides	2.5%	3.6%
	Ambloplites rupestris	0.3%	0.3%
	Lepomis spp.	0.6%	0.6%
	Lepomis cyanellus	55.9%	25.8%
	Lepomis macrochirus	6.2%	10.8%
	Lepomis humilis	0.4%	0.5%
	Lepomis microlophus	0%	<0.1%
	Lepomis gibbosus	<0.1%	0%
	Pomoxis nigromaculatus	0.2%	0%
	Pomoxis annularis	<0.1%	0%
Cyprinidae		15.8%	35.6%
	Notropis stramineus	0.8%	9.5%
	Notropis dorsalis	<0.1%	0.1%
	Cyprinella spiloptera	2.6%	2.8%
	Cyprinella whipplei	0%	<0.1%
	Notomigonus crysoleucas	0.4%	0.2%
	Campostoma anomalom	<0.1%	3.0%
	Pimephales notatus	5.3%	12.8%
	Pimephales promelas	0.8%	<0.1%
	Semotilus atromaculatus	3.7%	2.4%
	Carassius auratus	0.2%	1.6%
	Cyprinus carpio	1.8%	2.9%
	Nocomis biguttatus	<0.1%	0.2%
Catostomidae		7.4%	6.4%
	Catostomus commersonii	7.4%	6.3%
	Carpiodes cyprinus	0%	<0.1%
Ictaluridae		5.3%	10.4%
	Ameiurus natalis	4.1%	3.1%
	Ameiurus melas	0.2%	0.5%
	Noturus flavus	0.2%	0.1%
	Noturus gyrinus	<0.1%	0.8%
	Ictalurus punctatus	0.1%	0%
Percidae		0.6%	6%
	Etheostoma nigrum	0.6%	6%
	Etheostoma zonale	<0.1%	<0.1%
Gobiidae			
	Neogobius melanostomus	0%	0.2%
Fundulidae			
	Fundulus notatus	4.6%	0.7%
Poeciliidae			
	Gambusia affinis	1.2%	0%
Umbridae			
	Umbra limi	<0.1%	0%
Clupeidae			
	Dorosoma cepedianum	0.4%	0.7%

Table 4.2. Means (SE) for fish community metrics using fall samples (n=24) on the West and East Branch of the DuPage River. ANOVA tests for significant differences are included. Significant p-values at alpha=0.05 are denoted with *.

Metric	West Br.	East Br.	F	P-value
Species Richness	10.9 (0.6)	12.5 (0.7)	1.3	0.20
Abundance (fish/hr)	370.3 (37.3)	429.6 (69.0)	0.5	0.61
Biomass (g/hr)	3889 (560.6)	4803 (521.5)	0.9	0.38
Shannon-Weaver H’	1.5 (0.1)	1.8 (0.1)	1.2	0.24

Table 4.3. Regression statistics between relative abundance of YOY and juvenile Smallmouth bass and QHEI habitat metrics and macroinvertebrate community metrics on the West Branch of the DuPage River. All relationships significant at alpha=0.05 are denoted with *.

Metric	t-statistic	Adj. R^2	P-value
Substrate	0.1	0.0	0.79
Instream Cover	5.0	0.15	0.18
Channel Morph.	4.2	0.12	0.06
Riparian Zone	1.8	0.03	0.19
Pool-Run	0.6	0.0	0.44
Gradient	0.1	0.0	0.76
Total Mac Abund.	1.3	0.01	0.26
Taxonomic Rich.	0.1	0.0	0.77
EPT Taxa Rich.	0.9	0.0	0.36
EPT Abundance	0.2	0.0	0.66
Percent EPT Taxa	1.6	0.03	0.22
Percent EPT Ab.	2.2	0.05	0.14
HBI	0.1	0.0	0.70
ASPT	0.1	0.0	0.75

CHAPTER 5: SUMMARY AND CONCLUSIONS

Urbanization has impacted streams in the urban environment in many ways, including hydrologically, physically, chemically, and biologically. Impacted streams are often subject to restoration in an attempt to mitigate the negative effects of the urban environment on the ecological function of the stream and the diversity of life within it. Though restorations are common, they often lack assessment after the fact, leaving a knowledge gap as to the effectiveness of restorations in achieving the goals set forth prior to restoration activities. In this thesis, I examined a large-scale restoration of the West Branch of the DuPage River, an urban stream in West Chicago, IL, using the East Branch of the DuPage River, a nearby, unrestored urban stream as a reference to determine the effectiveness of a suite of restoration actions, including riparian zone improvements, stream bank stabilization, channel remeandering, substrate improvement, instream habitat improvement, and sediment remediation, on improving quality of instream habitat and water quality as well as the restoration's impact on the macroinvertebrate and fish communities.

In chapter two, I discussed the restoration of the West Branch and it's impacts on the physical habitat in the stream as well as the water chemistry. This chapter demonstrated that the restoration of the West Branch created more heterogeneous instream habitat, less incised channels, and more developed, higher quality pools and riffles when compared to the East Branch. In terms of water quality, restoration activities on the West Branch had little effect on temperature, dissolved oxygen, and flow, which were within expected ranges for Midwestern temperate streams. Conductivity remained extremely high on both streams, indicating that changes in land cover and land use, rather than in stream habitat improvements, would be necessary to reduce artificially high conductivity in these urban streams. In chapter three, I found that the two streams contained different macroinvertebrate communities based on differences in species richness, biodiversity, and tolerance levels. The West Branch had higher numbers of EPT taxa, higher abundance of EPT individuals, and contained more sensitive and intolerant species as indicated by HBI and ASPT scores. The increased presence of EPT and other environmentally sensitive species indicates improved instream habitat and better water quality on the West Branch, leading me to conclude that instream habitat may play a more important role in determining macroinvertebrate community diversity, although other potentially important water quality parameters (i.e. nitrates, phosphates, and pollutants) were not examined in this study. In chapter four, I assessed the restoration of the West Branch with regards to the fish community. There were few differences between the West and East Branches in terms of fish community metrics examined, but the abundance of Smallmouth Bass was much higher on the West Branch, especially with respect to abundance of young of the year, indicating the improvement of habitat and available forage for this economically and socially important sport fish species. The abundance of young Smallmouth throughout the West Branch indicates that it may provide quality nursery habitat and forage for young bass, but the presence of a dam just downstream of the restoration creates the possibility of isolation of this population. This could lead to problems such as genetic isolation, inbreeding depression, and an unsustainable fishery, all of which could be studied in subsequent projects. Interestingly, the consistent difference between macroinvertebrate communities in concert with the relative lack of a difference in fish communities could indicate that specific restoration methods were better designed for macroinvertebrate community recovery, or that other, landscape scale factors are continuing to impact fish communities in ways that they are not impacting macroinvertebrate communities.

My thesis demonstrated the success of specific goals of the restoration, namely the creation of a diverse macroinvertebrate community as well as forage and cover for Smallmouth Bass, a prized sport fish, but also demonstrated the limitations of restoration in the urban setting. High conductivity due to high rates of road salt use, impervious surfaces, and lack of natural indicates that changes to land use and human activities on the landscape may be needed to further address water quality in urban streams, and the removal of dams or creation of fish passage structures downstream of the restoration may be necessary to restore connectivity and facilitate the upstream movement of fish into restored reaches of the West Branch. My thesis is an example of the benefits of a structured post-project restoration, but also highlights the need for pre-project data sets for more robust comparisons. Without repeatable pre-restoration data, or consistent sampling sites and methodologies, assessments can still be done, but will lack the rigor of consistent methods. Consistency in, and use of pre- and post-restoration assessments is especially important for complex, costly, and ecologically important restorations to be effective at achieving their goals.

The methods used in my thesis were selected to focus on macro scale abiotic and biotic responses to restoration activities. Though I looked at multiple abiotic and biotic factors, there were some limitations to my study which could be used in future research. They include a more fine scale, quantitative evaluation of the physical instream habitat, which would allow for a more detailed examination of the effects of specific habitat features and reveal how restoration activities may have provided specific ecological benefits to the stream ecosystem. Genus and species identification of macroinvertebrates would allow for the utilization of functional groups in the assessment of stream health, but due to the level of expertise required to be confident in identifications at that classification level, as well as the focus of the thesis being a broader view across multiple biotic and abiotic factors, this was not done. Finally, the use of long term data sets and consistent monitoring of water quality, physical habitat, macroinvertebrate community, and the fish community would allow for feedback about the process of restoration, including what methods are effective for attaining goals, and where there may be areas for improvement.

The restoration of the West Branch of the DuPage River was a large-scale effort to mitigate decades of degradation of an urban stream while providing the ecological foundation ecosystem services highly valued by a densely urban human population. With rapid expansion and population growth in urban centers, the need for outdoor recreation, fishing opportunities, and ecologically sound natural areas has become greater. The West Branch restoration has provided these ecosystem services, with aesthetic recreation areas, a sport fishery, and other outdoor opportunities. The West Branch has also provided quality physical instream habitat, which supports a healthier, more diverse macroinvertebrate community than the East Branch. Outside of Smallmouth Bass and several other Centrarchids, however, the restoration has not created a more diverse fish community. This may be due to lag time following restoration, or may never occur due to the landscape scale effects of urbanization that have not been, and are very difficult to mitigate. Whether the effects of the restoration on the fish community will be gradual, or there are other factors limiting change, I contend that there is a need for premeditated, well thought out assessment of stream restoration project to adequately evaluate their effects on ecological function and the impact of improvements on ecosystem services.

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