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2013

Aquatic Macroinvertebrate Response to Invasion and Subsequent Removal of Typha × glauca From Great Lakes Coastal Marshes

Kimberly Anne Greene
Loyola University Chicago

Recommended Citation

LOYOLA UNIVERSITY CHICAGO

AQUATIC MACROINVERTEBRATE RESPONSE

TO INVASION AND SUBSEQUENT REMOVAL

OF *TYPHA × GLAUCA*

FROM GREAT LAKES COASTAL MARSHES

A THESIS SUBMITTED TO

THE FACULTY OF THE GRADUATE SCHOOL

IN CANDADICY FOR THE DEGREE OF

MASTER OF SCIENCE

PROGRAM IN BIOLOGY

BY

KIMBERLY A. GREENE

CHICAGO, IL

MAY 2013

ACKNOWLEDGMENTS

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ABSTRACT

Aquatic macroinvertebrates are important in wetland ecosystems; many fish and wildlife species depend upon them for food resources and they aid in nutrient cycling. Wetland macroinvertebrate communities are influenced by plant community composition; as such, this study examined two implications of invasive hybrid cattail, *Typha × glauca* (*Typha*), on aquatic macroinvertebrate communities in northern Great Lakes coastal wetlands: 1) how the presence of nearly monotypic stands of *Typha* alters wetland aquatic macroinvertebrate communities and 2) how manual removal of *Typha*, via mowing and manual tilling, impacts these invertebrate communities along with wetland plant communities. I found that in emergent marsh habitats the presence of *Typha* decreased aquatic macroinvertebrate density and altered biomass production throughout the summer. I also established that the methods of *Typha* management implemented in this study did not significantly impact the standing stock of wetland macroinvertebrates but did alter the plant community composition within the study wetlands. This research showed that the presence of invasive *Typha* negatively impacts aquatic macroinvertebrate communities within Great Lakes coastal wetlands and that removal of *Typha* from these wetlands improves the quality of the plant community without further disturbing these invertebrate communities one year post removal. Early intervention of invasive plants like *Typha* could help to ensure Great Lakes coastal wetlands retain their unique ecological communities and functions.

CHAPTER ONE:

LITERATURE REVIEW AND THESIS MOTIVATION

Literature Review

Invasive Species

Invasive species are widely recognized as a significant economic burden and major threat to native ecosystems and global biodiversity. According to Pimentel et al. (2000) there are approximately 50,000 non-native species in the United States. These introduced species are partially responsible for roughly 42% of species now listed on the Threatened or Endangered Species List in the U.S., where costs attributed to invasive species reach nearly $120 billion dollars per year (Pimentel et al. 2005).

Certain parameters govern how susceptible an ecosystem may be to invasion, including: ecosystem properties, propagule pressure and properties of potential exotic species (Lonsdale 1999). Wetlands appear to be particularly vulnerable to invasion; they make up a small proportion of Earth's landmass, but harbor a disproportionate number of the most detrimental invasive plant species (Zedler & Kercher 2004). This may be due to their location in the landscape where excess water, sediments, seeds and nutrients accumulate (Zedler & Kercher 2004). Furthermore, wetland plants tend to be particularly successful invaders; many share traits that likely increase their invasion effectiveness, such as high productivity, efficient growth rates and the ability to reproduce vegetatively (i.e. rhizomatous growth) (Boers & Zedler 2008, Schultz & Dibble 2012).

Invasive wetland plant species can greatly impact the diversity of other wetland

organismal groups. Multiple studies in wetlands have established that invasive plants decrease native plant diversity, alter nutrient and water cycles, and modify food webs (Ehrenfeld 2003, Zedler & Kercher 2004, Tuchman et al. 2009, Mitchell et al. 2011). Schultz & Dibble (2012) established that invasive macrophyte species in wetlands frequently lead to decreased abundance of native fish and macroinvertebrate species. Phillips (2008) found significant differences in invertebrate density and richness between wetland areas containing native *Potamogeton richardsonii* versus invasive *Myriophyllum spicatum* or *Potamogeton crispus*. Likewise, Blossey et al. (2001) concluded that invasion of the emergent macrophyte *Lythrum salicaria* into northern North American wetlands drastically decreased the ability of wetland specialist birds including Black Terns, Least Bitterns, Pied-Billed Grebes and Marsh Wrens to utilize wetland habitats.

Typha × glauca

     Typha × glauca (hereafter *Typha*) is an invasive hybrid cattail that is found in North America where native *Typha latifolia* and non-native *Typha angustifolia* regionally overlap and cross pollinate (Smith 1987). Once established, *Typha* typically forms dense monotypic clonal stands with the potential to spread via rhizomatous growth up to 4 m in diameter per year (Boers & Zedler 2008). *Typha* is associated with increased plant biomass, litter biomass and soil organic matter, as well as decreased plant species richness relative to native ecosystems (Tuchman et al. 2009, Lishawa et al. 2010). *Typha* is a highly productive species with much greater biomass than the native species it replaces (Woo and Zedler 2002), resulting in deposition of abundant leaf litter that further promotes the dominance of *Typha* over other wetland plant species (Farrer and Goldberg 2009, Tuchman et al. 2009, Vaccaro et al. 2009, Larkin et al. 2011). Many negative

impacts associated with *Typha* invasion are attributed to litter production (Freyman 2008,

Tuchman et al. 2009, Farrer & Goldberg 2009, Larkin et al. 2011). Larkin et al. (2011)

found that increased litter from *Typha* decreased light transmission to the soil, soil

temperature and native plant species biomass. In *Typha* stands aged more than 35 years,

the accumulation of litter and soil organic matter could potentially shift hydrologic

regimes significantly altering the habitat (Mitchell et al. 2011).

 Wetland Macroinvertebrates

Wetlands contain an abundance of invertebrate prey which are a critical food

resource to numerous Great Lakes wildlife species. Many fish (Jude and Pappas 1992,

Cvetkovic et al. 2010), marsh breeding birds (Krull 1979, de Szalay & Resh 1997,

Conway 2009) and herpetofauna (Anderson 1999, Wieten et al. 2012) are dependent

upon wetland food resources at some point during their life cycle. In addition to local

fauna, migratory birds are exceedingly dependent on wetland macroinvertebrate food

resources along the Great Lakes during spring and fall migrations (Kostecke et al. 2005).

Although aquatic macroinvertebrates are abundant in Great Lakes wetlands, harsh

hydrologic conditions, such as fluctuating hydroperiods and anoxia, select for hardy

species. These conditions lead to lower invertebrate diversity in wetlands as compared to

other aquatic habitats such as, lakes or rivers (Batzer et al. 1999). Variable hydroperiods

or complete desiccation can force organisms to either migrate out of dry areas or to enter

diapause during some life stage (Tronstad et al. 2005). Due to stagnant water, wetlands

typically have anoxic benthic environments, and many wetland invertebrates exhibit

features to cope with low dissolved oxygen. These features include the presence of

invertebrate hemoglobin, elongated respiratory siphons, or the ability to collect and carry

atmospheric air bubbles underwater. A few macroinvertebrate groups including chironomid larvae can survive using only cutaneous respiration (Batzer et al. 1999, Merritt, Cummins & Berg 2008).

Along with variable hydrologic regimes, water quality and sediment composition create stressful conditions for aquatic wetland invertebrates (Batzer et al. 1999). Cardinale (1996) described a water quality gradient in a Lake Huron coastal wetland where less pelagic-littoral mixing took place further away from open water. Along this gradient he found decreased turbidity, pH, and dissolved oxygen with increased sampling distance from open water. These changes in water quality indirectly affected the aquatic macroinvertebrate community by decreasing epiphytic algal production and the feeding efficiency of collector-filterer organisms. Many benthic invertebrates are likewise dependent on the composition and quality of wetland sediments. Cole & Weigman (1983) found that wetland zoobenthos occurred in higher abundances in silty sediments as opposed to sandy substrates, and that the occurrence of amphipods was closely related to the presence and amount of deposited organic matter.

Wetland macroinvertebrates are an important group of invertebrates that are often overlooked in ecological and restoration studies. Among upper Great Lakes coastal wetlands, basic descriptive studies of macroinvertebrates are still lacking: as of 1981 there were no site-specific reports of wetland invertebrates from Lake Superior, Lake Michigan or Lake Huron. Through 1992 there were only 2 studies conducted in coastal wetlands of the upper Great Lakes, one on the St. Marys River and a second in Green Bay. Although aquatic macroinvertebrate communities were extensively studied in the *Schoenoplectus* spp. zones in Saginaw Bay, Lake Huron between 1992 and 1997, more

recent macroinvertebrate community studies are lacking (Batzer et al. 1999). Because coastal wetlands are spatially diverse with high microhabitat and taxonomic diversity, descriptive studies prove problematic. Burton & Uzarski (2009) estimated that macroinvertebrate species richness likely exceeds 300 species in Lake Huron protected coastal wetlands. However, because of the extreme habitat heterogeneity, it is difficult to extrapolate these data into accurate estimates of macroinvertebrate species diversity in other types of Great Lakes coastal wetland habitats.

Association of Wetland Plant Communities with Aquatic Macroinvertebrates

Wetland ecosystems are typically litter-driven systems in which macroinvertebrates play a critical role in the flow of energy and cycling of nutrients. In these systems, invertebrates are a necessary link between autotrophic organisms and secondary consumers such as fish and birds. Macroinvertebrates are commonly the focus of studies addressing the importance and feasibility of wetlands to act as suitable habitats for wildlife. As Batzer et al. (1999) discusses, the source of autotrophic material in different wetland systems can alter the composition and functioning of the entire macroinvertebrate community therein (Fig. 1).
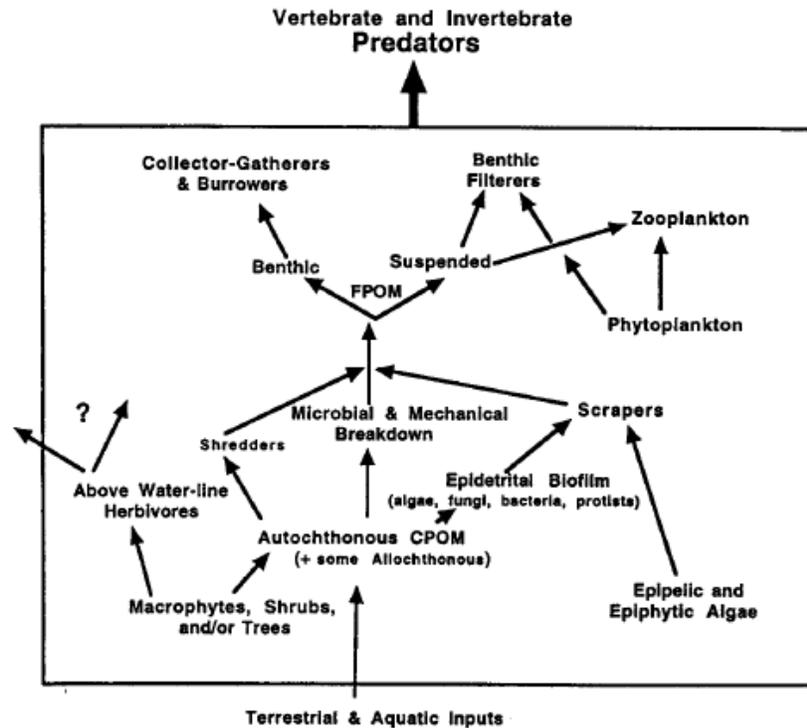
Fig. 1. Flow chart of invertebrate functional feeding group pathways for the conversion of food energy from autotrophic organisms through primary and secondary consumers (from Bazter et al. 1999).

Changes in plant community composition can alter macroinvertebrate abundance and diversity (Voigts 1970, Krull 1979, Olson 1995, Gardner et al. 2001, Burton et al. 2002, Gathman & Burton 2011) by altering food resources (Batzer et al. 1999) and changing the structural heterogeneity of the wetland habitat (Krecker 1939, Dvořaki and Bestz 1982). Alterations in plant community structure and composition can also result in a "trophic shift," as illustrated by Levin et al. (2006) where a mud flat in San Francisco Bay invaded by a hybrid emergent macrophyte, *Spartina*, resulted in a shift from an algal- to a detritus-based system.

Although it has been shown that invasive plant species are associated with drastic alterations in wetland ecosystem functions (Zedler & Kercher 2004, Tuchman et al. 2009,

Larkin et al. 2011) few studies have examined the direct or indirect effects of these invasive plants on wetland macroinvertebrate communities. These studies have found inconsistent and contradictory results. For instance, Warren et al. (2001) examined the effects of *Phragmites australis* invasion and subsequent management on macroinvertebrate communities on the lower Connecticut River. *P. australis* invasion and associated management had little effect on the macroinvertebrate community except that macroinvertebrate densities were positively correlated with *P. australis* litter production. Gardner et al. (2001) determined that aquatic macroinvertebrate abundances did not differ between habitats dominated by invasive *Lythrum salicaria* versus native *T. latifolia* or *Scirpus acutus* in a Washington state wetland. However, some taxa collected from *L. salicaria* stands were smaller than their counterparts collected from the native vegetation zones. Studies conducted in ordinarily unvegetated mudflats invaded by *Spartina anglica* or hybrid *Spartina* report a decrease in invertebrate abundance and composition post-invasion (Frid et al. 1999, Levin et al. 2006). Frid et al. (1999) concluded that the management of *S. anglica* with physical disturbance (i.e. off road vehicles) successfully decreased the presence of the invasive plant but had no apparent effect on the benthic invertebrate community.

Schultz & Dibble (2012) summarized studies focused on the effects of invasive macrophytes on freshwater fish and macroinvertebrate communities in wetlands. They concluded that invasive macrophytes can alter habitat complexity, increase hypoxia and allelopathic chemicals, facilitate invasion by other species and decrease food quality. Additionally, they identified increased growth rates, allelopathic chemical production, and phenotypic plasticity as three unique traits of the invaders likely responsible for

negative effects of macrophytes on fish and macroinvertebrate communities.

Effects of Invasive Species Management on Macroinvertebrate Communities

Many invasive wetland plants form dense, nearly monotypic stands resulting in decreased plant diversity, increased litter production, increased soil nutrients, and decreased soil surface light availability and temperatures (Angeloni et al. 2006, Farrar & Goldberg 2009, Tuchman et al. 2009, Lishawa et al. 2010, Larkin et al. 2012, Mitchell et al. 2011). A variety of management techniques are used to control invasive wetland plants, including physical removal, herbicide application, prescribed fire, shading and water level modification (diking) (Apfelbaum 1985). Ecological management is costly and labor intensive, often requiring repeated treatments; as such there is no single best practice for invasive plant management that applies to all wetlands. When implemented, management strategies are often limited in their efficacy in eradicating and managing the return of plant invaders. Furthermore, many of the management strategies used to control invasive plants prove environmentally unsustainable. Herbicide application, for instance, introduces foreign chemicals into the target ecosystem; and can negatively impact non-target organisms. Prescribed fires, another common management strategy, release substantial particulate matter and greenhouse gases into the atmosphere.

Invaded wetlands are often intensively managed to promote their use by waterfowl and other game species (Gray et al. 1999, Davis & Bidwell 2008). As such, there have been a substantial number of studies conducted on the effects of various wetland management techniques on macroinvertebrates important to waterfowl and migratory bird species (See Huener & Kadlec 1992, Brown et al. 1997, de Szalay & Resh 1997, Gray et al. 1999, Balcombe et al. 2005, Kostecke et al. 2005, Davis & Bidwell

2008). Macroinvertebrate communities may be affected by the presence of invasive wetland plants as well as the ecological management of these plant species. Because of the great variation in wetland ecosystems studied and the array of management techniques employed, there is no single conclusion regarding the overall effect of management on macroinvertebrate communities. De Szalay & Resh (1997) observed the effects of prescribed burning and mowing on macroinvertebrates in a California brackish marsh and found that burning increased macroinvertebrate densities, whereas the macroinvertebrate communities within mowed areas were not significantly different than control plots. In contrast, Gray et al. (1999) found that in an emergent marsh in Mississippi, mowed and control treatments resulted in greater aquatic invertebrate mass and diversity over disked and tilled treatments. Other studies indicate that macroinvertebrate communities do not respond differently to burning, disking and cattle grazing (Kostecke et al. 2005) or burning, grazing, mowing and disking (Davis & Bidwell 2008).

Thesis Motivation

The effects of invasive plants on wetland macroinvertebrate communities are poorly understood and not consistent across study systems and organisms (Frid et al. 1999, Levin et al. 2006, Gardner et al. 2001, Schultz & Dibble 2012). Few studies have examined the effects of invasive *Typha* on wetland macroinvertebrate communities (Gardner et al. 2001, Burton et al. 2002, Christensen & Crumpton 2010), and of these, only Burton et al. (2002) was conducted within the Great Lakes basin. Likewise, descriptive studies of macroinvertebrate communities within Great Lakes coastal wetlands are sparse.

With this study, I aim to further describe aquatic macroinvertebrate communities within upper Great Lakes coastal wetlands and investigate the effect of invasive *Typha* on these communities, which may be an important indicator of ecosystem health (Burton et al. 1999, Kashian & Burton 2000, Niemi et al. 2009). I also examine the early-stage impacts of *Typha* removal on aquatic macroinvertebrate communities.

In Chapter 2, I describe the aquatic macroinvertebrate community within three northern Lake Huron coastal wetlands that have been largely invaded by *Typha*. Using aquatic macroinvertebrate survey techniques, I characterize the aquatic macroinvertebrate communities within native emergent marsh habitats, and compare them with macroinvertebrate communities occurring in the same habitat invaded by *Typha*.

In Chapter 3, I test the effect of three mechanical management techniques of *Typha* on aquatic macroinvertebrate communities within two northern Lake Huron coastal wetlands. I describe aquatic macroinvertebrate communities where *Typha* was removed by mowing or modified tillage during July and August 2011. I compare the macroinvertebrate communities within these managed plots to control areas of the marsh, which are dominated by invasive *Typha*.

CHAPTER TWO:

EFFECTS OF *TYPHA × GLAUCA* ON AQUATIC MACROINVERTEBRATE

COMMUNITIES IN NORTHERN LAKE HURON COASTAL WETLANDS

Introduction

Wetland macroinvertebrates play a crucial role in wetland ecosystems, linking primary producers and secondary consumers. Many wildlife species, including fish (Jude & Pappas 1992, Cvetkovic et al. 2010), marsh breeding birds (Krull 1979, de Szalay & Resh 1997, Conway 2009) and herpetofauna (Anderson 1999, Wieten et al. 2012) depend upon wetland systems for habitat and food resources during some portion of their life cycle. Wetland invertebrates are also an indispensable food resource for many migratory bird species during spring and fall migrations (Anderson & Smith 2000, Kostecke 2005).

Wetland invertebrate populations are closely associated with plant communities, and changes in plant community structure impact invertebrate assemblages (Krecker 1939, Voigts 1970, Olson et al. 1995, Frid et al. 1999, Warren et al. 2001, Burton et al. 2002, Levin et al. 2006, Gathman & Burton 2011, Holomuzki & Klarer 2010, Schultz & Dibble 2012). Invasive plant species pose a serious threat to native plant community structure because they alter the vegetative community within wetlands. Invaded wetland habitats are associated with decreases in habitat heterogeneity and plant diversity (Zedler & Kercher 2004), often leading to changes in aquatic macroinvertebrate community structure (Stiers et al. 2011, Schultz & Dibble 2012).

The effects of invasive macrophytes on wetland macroinvertebrate communities are complex and unclear. No distinct pattern has been established regarding the various impacts on wetland macroinvertebrate communities attributed to invasive macrophytes. Often, invasive macrophytes alter physicochemical parameters important to survival and development of aquatic macroinvertebrates. For example, Rose & Crumpton (1996) showed that dense stands of emergent plants are associated with lower light availability, dissolved oxygen concentration and water temperature than less vegetated or non-vegetated areas. Similarly, Schultz & Dibble (2012) found that invasive macrophytes are associated with altered habitat complexity and dissolved oxygen availability, which leads to variable effects on aquatic macroinvertebrate communities. Stiers et al. (2011) suggests that uninvaded and invaded areas harbor different aquatic invertebrate taxa, and that aquatic invertebrate abundance is negatively associated with percent cover of invasive macrophytes. Finally, Levin et al. (2006) found decreased density and biomass of benthic macroinvertebrates in tidal mud flats after invasion by hybrid *Spartina.*

Typha × glauca (hereafter *Typha*) is an invasive hybrid cattail species that occurs where native *Typha latifolia* and non-native *Typha angustifolia* populations overlap and cross pollinate (Smith 1987). *Typha* can expand vegetatively at rates of up to 4-m in clone diameter per year (Boers and Zedler 2008), often resulting in large monotypic stands. The persistent spread of *Typha* throughout the Great Lakes region (Galatowitsch et al. 1999, Tulbure et al. 2007) has altered coastal wetland ecosystems by increasing plant biomass and litter production, decreasing native plant species diversity (Farrer & Goldberg 2009, Tuchman et al. 2009, Mitchell et al. 2011), altering nutrient cycling regimes (Farrer & Goldberg 2009, Tuchman et al. 2009) and decreasing soil light and

temperature regimes (Larkin et al. 2012). The preceding wetland ecosystem alterations have been attributed to *Typha* invasion; however, few studies have examined the effects of this invasive plant on wetland fauna.

Invertebrates are regularly overlooked in research on wetland ecosystems despite the critical role they play in food web configuration and nutrient cycling. Since *Typha* invasion drastically alters plant community composition and abiotic factors, it may also affect wetland invertebrate communities. Within the upper Great Lakes, no study has directly examined the impact of invasive *Typha* on wetland invertebrate communities and only a few studies have examined this interaction in other ecosystems (Gardner et al. 2001, Burton et al. 2002, Christensen & Crumpton 2010). Christensen & Crumpton (2010) found a negative correlation between the quantity of *Typha* litter and invertebrate abundance in a prairie pothole marsh. Burton et al. (2002) showed that plant communities, including areas dominated by *Typha*, coupled with wave action had an impact on the distribution and occurrence of macroinvertebrate communities at two sites in Saginaw Bay, Lake Huron.

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┌─────────────────────────────────────┐
│        Typha × glauca Invasion       │
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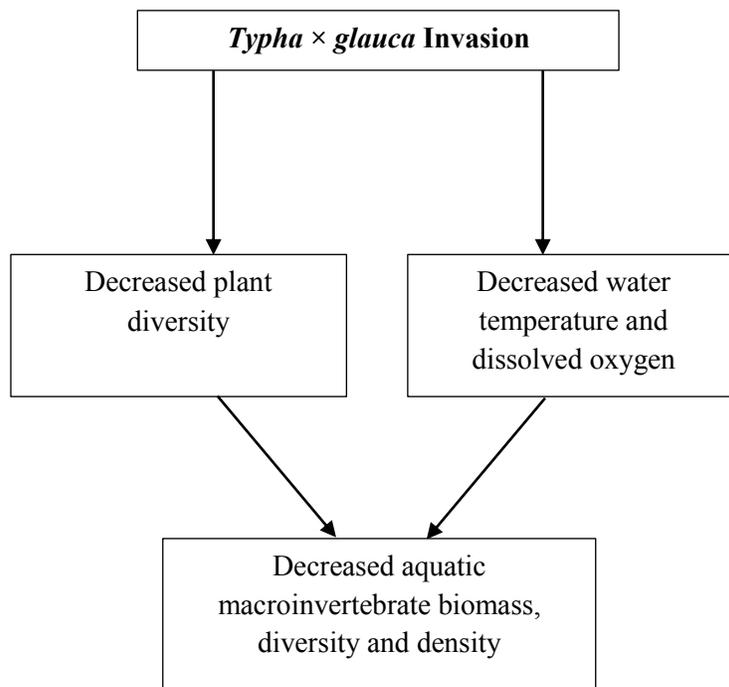

Fig. 2. Flow chart of mechanisms through which invasive *Typha* could decrease aquatic macroinvertebrate diversity and density in an invaded coastal marsh.

Great Lakes coastal wetlands contain a diversity of habitats, including sedge meadows, emergent marshes and swamps. Emergent marsh habitats contain narrow- and wide-leaved emergent plant species, in water typically 5- to 100-cm deep. *Typha* can potentially invade all coastal wetland habitats. *Typha* could theoretically alter wetland macroinvertebrate communities in emergent marshes through two main pathways: 1) by altering physicochemical characteristics such as dissolved oxygen or water temperature, and/or 2) by altering the plant community by decreasing plant diversity and plant species richness (Fig. 2.1).

My objective for this study was to determine how *Typha* invasion affects aquatic macroinvertebrate communities within emergent marshes. Because different aquatic

macroinvertebrate communities are associated with various plant community types and

Typha is associated with a decrease in plant species richness and diversity, I hypothesized

that areas invaded by *Typha* would also support less aquatic macroinvertebrate biomass,

density and diversity than native zones. Due to dense litter layers, macrophytes such as

Typha, are also associated with decreases in water temperature and dissolved oxygen

(Rose & Crumpton 1996, Larkin et al. 2012) which are important for aquatic

macroinvertebrate development and survival. Therefore, I predicted that decreases in

these physical parameters would also decrease aquatic macroinvertebrate biomass,

density, and diversity. To investigate this question, I conducted a wide-spread survey

comparing the aquatic macroinvertebrate communities present in native marsh areas to

comparable marsh zones dominated by *Typha* in three northern Lake Huron coastal

wetlands.

Methods

Survey Sites

To examine the effects of *Typha* on aquatic macroinvertebrate communities in

northern Great Lakes coastal wetlands I chose Cheboygan and Cedarville Marshes which

are open embayment freshwater coastal wetlands located on the coast of Lake Huron in

northern Michigan. St. Ignace Marsh is an open embayment freshwater coastal wetland,

located on the Straits of Mackinac (Fig. 2.2). Due to regional proximity, all three

wetlands have similar climates, plant assemblages, and soils (Albert et al. 2005). All

three sites also have a significant area of their emergent marsh habitats dominated by

nearly monotypic stands of *Typha* (Dennis Albert, personal communication).

Each of the study sites contained *Typha*-invaded emergent marsh and native

emergent marsh. Using GIS, aerial photography and ground truthing, I visually confirmed

Typha-invaded areas were dominated by *Typha* stems whereas areas with native plant

communities had few to no *Typha* stems and were comprised mainly of native wetland

plant species. To reduce the confounding effects of water depth, I targeted sites where

each *Typha*-invaded area and corresponding native zone had comparable water depths. In

June 2011, I randomly chose three plot locations in each of the vegetation zones,

resulting in 18 total plots, across the three sites.
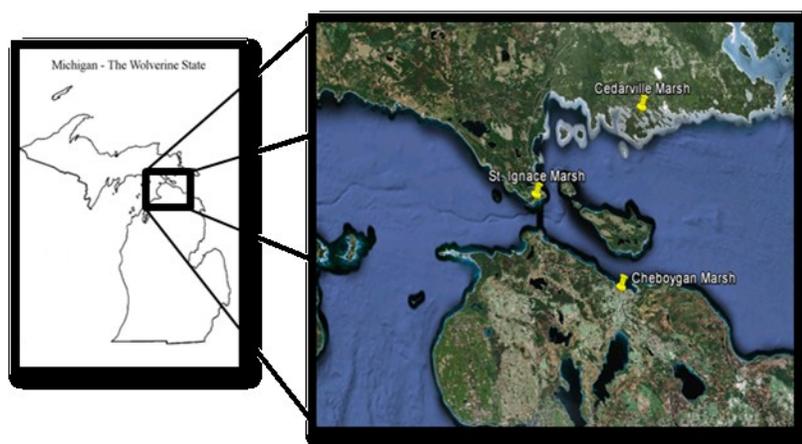
Fig. 3. Map of the Mackinac Straits where Lake Michigan and Lake Huron join, with yellow marks
indicating the location of the three study sites where samples were collected in 2011.

Vegetation Zone Characterization

I counted stems of all plant species present within a 1-m^2 plot where each

invertebrate sample was taken. I also recorded percent cover of any submergent or

floating-leaved plant species. I measured soil redox potential with a voltmeter as the

difference in electron potential between a calibrated platinum-tipped redox probe and an

Accumet calomel reference electrode (Vespraskas & Faulkner 2001). I also measured

water depth to the nearest 0.5-cm with a meter stick and water temperature and dissolved

oxygen (DO) using a Hach dissolved oxygen probe (Hach co. Loveland, CO, Model HQ30d).

Aquatic Macroinvertebrate Sample Collection

During the months of June, July and August 2011, I sampled each of the 18 plots using the stovepipe collection method (Gathman & Burton 2011). At each plot I randomly chose a location where the sample would be taken within 3-m of plot center. I then placed a 1-m^2 quadrat and collected an invertebrate sample by forcing a 14.5-cm diameter aluminum stovepipe through the water column and approximately 5-cm into the sediments. Once the stovepipe was in place, I scooped out any loose vegetation and placed it into a 300-μm sieve. I then agitated the water within the stovepipe to loosen the top 2- to 3-cm of sediment and used a syphon-pump to extract the water and invertebrates out of the pipe and into the sieve.

To standardize biomass and species richness calculations, each stovepipe was pumped 100 times, regardless of water depth; I established that 100 total pumps was sufficient to extract the majority of invertebrates within the stovepipe. If fewer than 100 pumps emptied the stovepipe, I allowed water to seep back into the stovepipe before continuing. After the invertebrates were pumped into the sieve, I transferred them to collection jars and transported them back to the laboratory in a cooler.

I processed the stovepipe samples within 24 hours of collection by rinsing them through a 300-μm sieve to decrease the amount of fine sediment and particulate matter in the sample. Once rinsed, I placed each sample into a white enamel pan and picked out all visible macroinvertebrates. I considered sampling complete after a five minute interval

without finding any additional organisms. All invertebrates were stored in 95% EtOH.

I identified each invertebrate to the lowest taxonomic unit possible using Merritt, Cummins & Berg (2008) to identify insect specimens and Thorp & Covich (1991) for non-insect specimens. I identified chironomid larvae to sub-family or tribe. To use length-mass regressions to calculate biomass, I measured the body length of each individual from the anterior tip of the head to the posterior tip of the abdomen. Length-mass regressions are ideal for calculating biomass as they are faster and more precise than direct weighing or biovolume calculations, and because most organisms lose dry mass when preserved (Benke et al. 1999). I used established length-mass regressions at the lowest possible taxonomic unit available (see Benke et al. 1999, Baumgartner & Rothhaaupt 2003, Miserendino 2001, Hall et al. 2006, Stead et al. 2003, Stoffels et al. 2003, Edwards et al. 2009, Gruner 2003). For the two taxa that had no published regressions, *Parapoynx* sp. and Enchytraeidae, I used a published regression for a species with a comparable morphology (Reese & Batzer 2007).

Statistical Analysis

Because I sampled the same plots in June, July and August, I utilized repeated measures ANOVA to compare water depth, *Typha* stem density, water temperature and dissolved oxygen among plots. I used two-sample t-tests to compare means between the vegetation zones within each site during each month sampled. I also compared macroinvertebrate density, biomass and Shannon-Wiener diversity index among vegetation zones, sites and months using repeated measures ANOVA in R (Version 2.15.1, R Development Core Team 2012). I used repeated measures ANOVA because

monthly samples from the same plot were not independent from one another. For each analysis I used site, vegetation zone and site × vegetation zone as between-sample variables; while month, month × site, month × vegetation zone and month × site × vegetation zone were within-sample variables. To meet assumptions of normality, I transformed all biomass data using log (x + 1) transformation prior to statistical analysis (McCune & Grace 2002). To test for linear relationships between plant species diversity and macroinvertebrate diversity and biomass, I used ordinary least squares regression in SYSTAT 13 (SYSTAT software Inc. 2009). To compare aquatic macroinvertebrate communities between vegetation zones and relate them to multiple environmental variables, I conducted Nonmetric Multidimensional Scaling (NMDS). I ordinated sample units from all three months in species space using R (Version 2.15.1, R Development Core Team 2012).

Results

Vegetation Zone Characterization

The vegetation communities were distinctly different between *Typha* and native emergent zones. The native emergent areas were typified by native emergent and submergent plant species including *Schoenoplectus* spp., *Eleocharis* spp., *Nymphaea odorata* and *Utricularia* spp. The emergent plants in these areas are generally small, with a basal circumference of less than 2-cm and heights typically less than 1.5-m. In contrast, *Typha*-invaded areas were mainly dominated by large *Typha* plants, many with a basal circumference over 5-cm and heights over 2-m. Other wetland species were present within the *Typha*-invaded areas; however, they were generally sparse and were small

statured species such as *Galium* spp. and *Lysimachia thyrsiflora*.

 Typha density was consistently greater (P < 0.05) in *Typha*-invaded than native emergent zones with all interactions also significant (P < 0.05) except month × site. On average, *Typha*-invaded zones contained 30 times the number of *Typha* stems than native zones. There were few differences in environmental measurements between the two vegetation zones (Table 1).

Table 1. Mean environmental parameters measured in native and *Typha* areas sampled in June, July and August 2011 (± SE; n = 3). Comparisons within each vegetation zone during each month at each site are represented by letters; means with the same letter are not significantly different (P > 0.05).

		Cheboygan		Cedarville		St. Ignace	
		Native	*Typha*	Native	*Typha*	Native	*Typha*
June	Water Temp (˚C)	19.67 [a] (0.13)	19.50 [a] (0.15)	17.73 [a] (0.03)	17.49 [a] (0.13)	21.13 [a] (0.29)	20.03 [b] (0.19)
	Water Depth (cm)	19.67 [a] (1.83)	20.17 [a] (1.42)	62.50 [a] (2.59)	33.94 [b] (1.09)	35.67 [a] (2.40)	18.50 [b] (1.04)
	DO%	48.8 [a] (4.56)	47.13 [a] (4.09)	64.23 [a] (16.02)	88.80 [a] (12.42)	56.17 [a] (3.23)	52.93 [a] (6.41)
	Typha stems (no/m^2)	1.67 [a] (1.67)	32.00 [a] (8.14)	0.00 (0.00)	27.11 (4.56)	0.00 (0.00)	41.67 (6.17)
July	Water Temp (˚C)	23.87 [a] (0.35)	23.17 [a] (0.13)	24.20 [a] (0.26)	22.17 [a] (1.82)	25.37 [a] (0.09)	24.03 [a] (0.44)
	Water Depth (cm)	16.33 [a] (0.88)	20.17 [a] (1.42)	58.33 [a] (4.41)	27.00 [b] (3.51)	39.33 [a] (1.45)	19.67 [b] (0.33)
	DO%	52.55 [a] (6.05)	66.03 [a] (5.22)	79.80 [a] (4.58)	70.17 [a] (31.35)	68.8 [a] (9.25)	45.30 [a] (3.96)
	Typha stems (no/m^2)	5.00 [a] (4.00)	45.00 [b] (5.22)	0.00 (0.00)	23.67 (4.06)	0.00 (0.00)	46.67 (4.70)
August	Water Temp (˚C)	23.03 [a] (0.52)	23.10 [a] (0.15)	23.00 [a] (0.20)	23.03 [a] (0.78)	19.33 [a] (0.38)	18.60 [a] (0.55)
	Water Depth (cm)	14.50 [a] (0.50)	14.33 [a] (3.17)	45.50 [a] (2.50)	22.17 [b] (1.87)	30.00 [a] (1.53)	15.00 [b] (2.52)
	DO%	55.50 [a] (12.76)	63.47 [a] (5.27)	80.53 [a] (4.89)	120.73 [a] (44.79)	67.73 [a] (4.81)	35.90 [b] (0.90)
	Typha stems (no/m^2)	4.00 [a] (4.00)	19.33 [a] (5.69)	0.00 (0.00)	31.33 (3.53)	0.00 (0.00)	55.33 (7.31)

Aquatic Macroinvertebrate Communities

Between June and August 2011, I collected 83 aquatic macroinvertebrate taxa representing 18 Orders. The most common non-insect taxa were isopods (*Caecidotae* sp.), amphipods (*Crangonyx* sp.) and oligochaete worms (*Stylaria lacustris* and *Stylodrilus heringianus*). Dipterans (*Bezzia* sp. and Chironomidae) and mayflies (*Caenis* sp.) were the most numerically dominant insect taxa. Diptera, Odonata and Lumbriculidae collectively represented 44% of the total biomass.

Within each vegetation zone, macroinvertebrate total biomass, density and diversity were not significantly different among the three sites sampled ($P > 0.05$); indicating that the sites were suitable replicates. There was a visible trend of higher biomass and slightly lower Shannon-Wiener Index (H') in the native zone as compared with the *Typha*-invaded zone (Fig. 4 A-C).
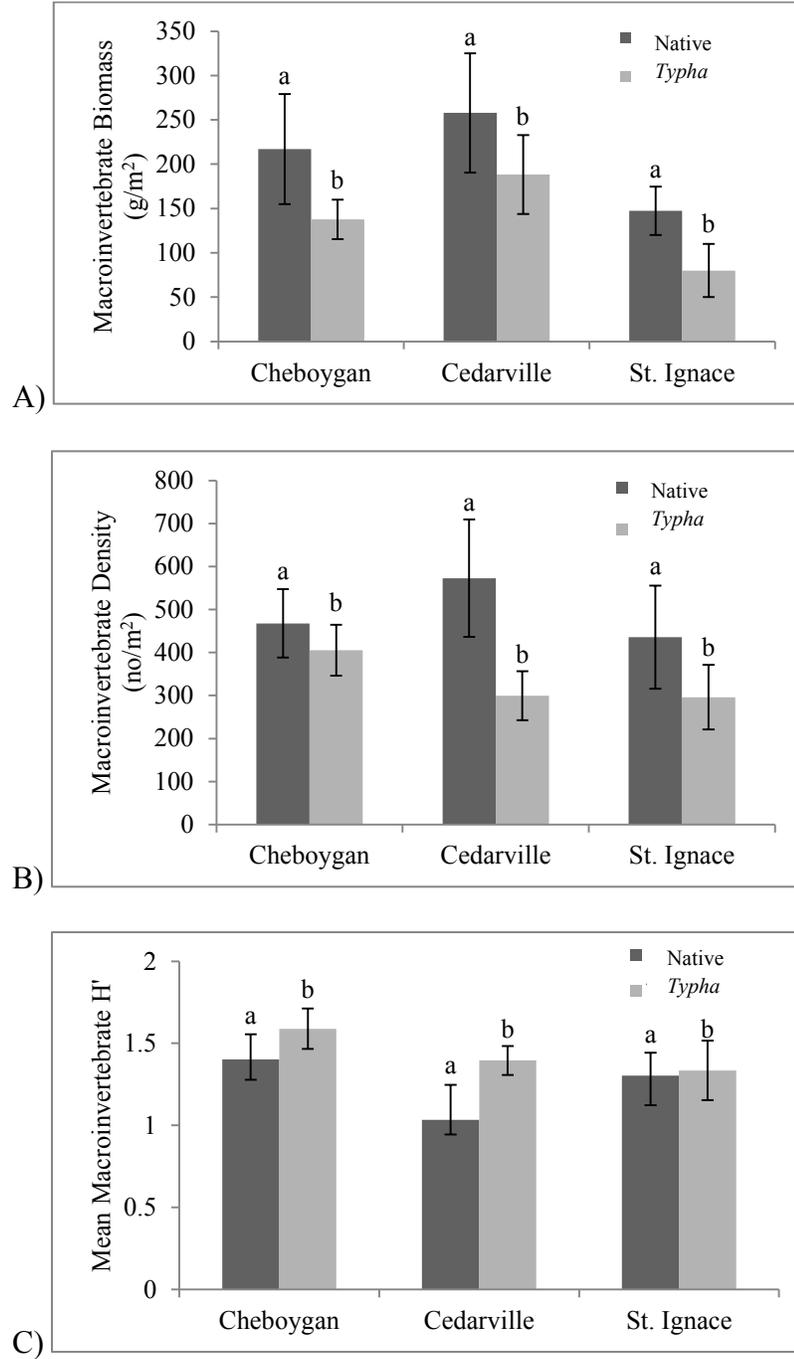
Fig. 4. Mean total A) biomass, B) density and C) Shannon-Wiener diversity (H') among the three sites sampled, error bars represent ±1 SE. Both vegetation zones are shown for each study site. Comparisons are within each vegetation zone, means with the same letter are not significantly different (P > 0.05).

Repeated measures ANOVA on vegetation zones or sampled sites revealed that aquatic macroinvertebrate density was significantly greater (P = 0.024, Fig. 5 A) in the native emergent zone in comparison with *Typha*-invaded zone over the three month sampling period; the only significant interaction was month × habitat × site (P = 0.04). Neither biomass nor diversity were significantly different between the two vegetation zones (P > 0.05, Fig. 5 B, C). In the native emergent marsh zone, mean aquatic macroinvertebrate biomass steadily accumulated throughout the months of June (159.68 gm/m^2), July (182.18 gm/m^2) and August (280.33 gm/m^2). In contrast, mean aquatic macroinvertebrate biomass in *Typha*-invaded emergent zones decreased from June (174.01 gm/m^2) to July (108.15 gm/m^2), and increased marginally from July to August (123.9 gm/m^2) (Fig. 5 B).
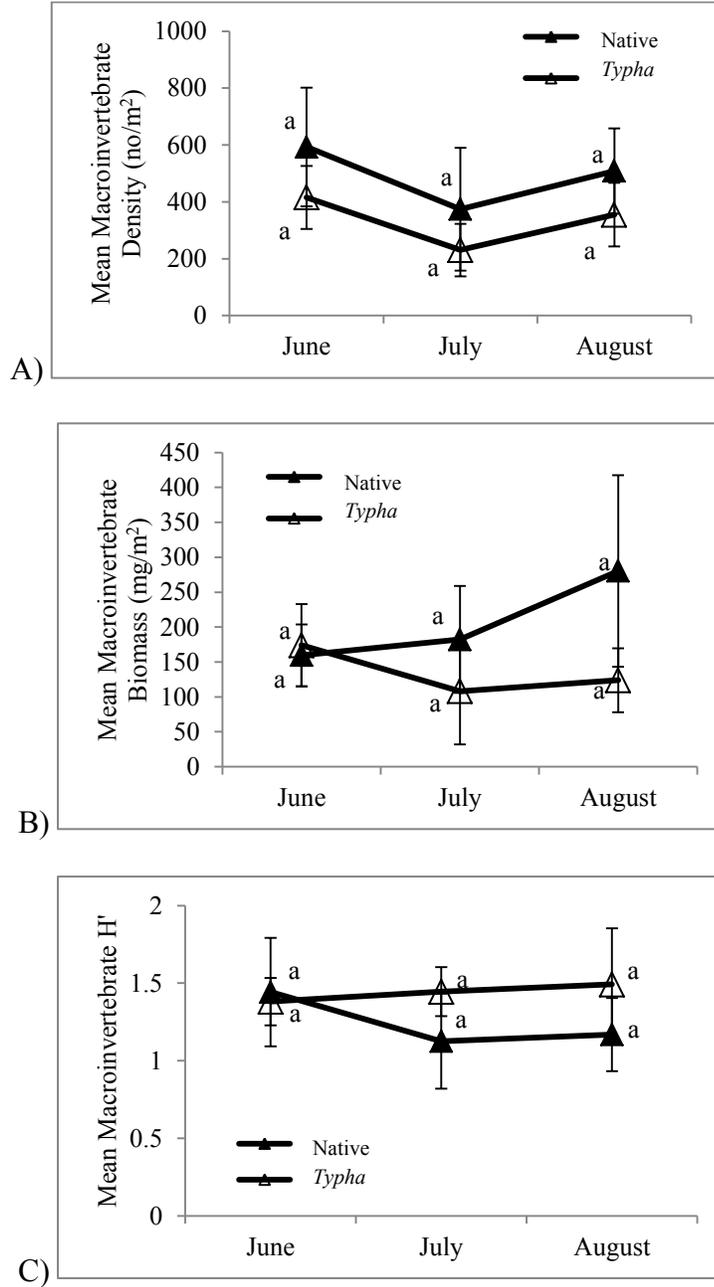
Fig. 5. Differences in aquatic macroinvertebrate A) density, B) biomass and C) Shannon-Wiener diversity. (± 1SE) between native emergent and *Typha*-invaded emergent zones sampled during June, July and August 2011. Comparisons are within each month, means with the same letter are not significantly different (P > 0.05).

Aquatic macroinvertebrate biomass showed a positive relationship with plant species richness (n = 54, R^2 = 0.21, P = 0.020, Fig. 6 A). However, there was no relationship between plant species richness and aquatic macroinvertebrate density (n = 54, R^2 = 0.0004, P = 0.744, Fig. 6 B). Insect biomass was also positively associated with plant species richness in both native and *Typha*-invaded emergent marsh zones (n = 54, R^2 = 0.16, P = 0.007, Fig. 7 A). In contrast, the relationship between non-insect macroinvertebrates and plant species richness was not significant (n = 54, R^2 = 0.04, P = 0.110, Fig. 7 B).

A)

B)
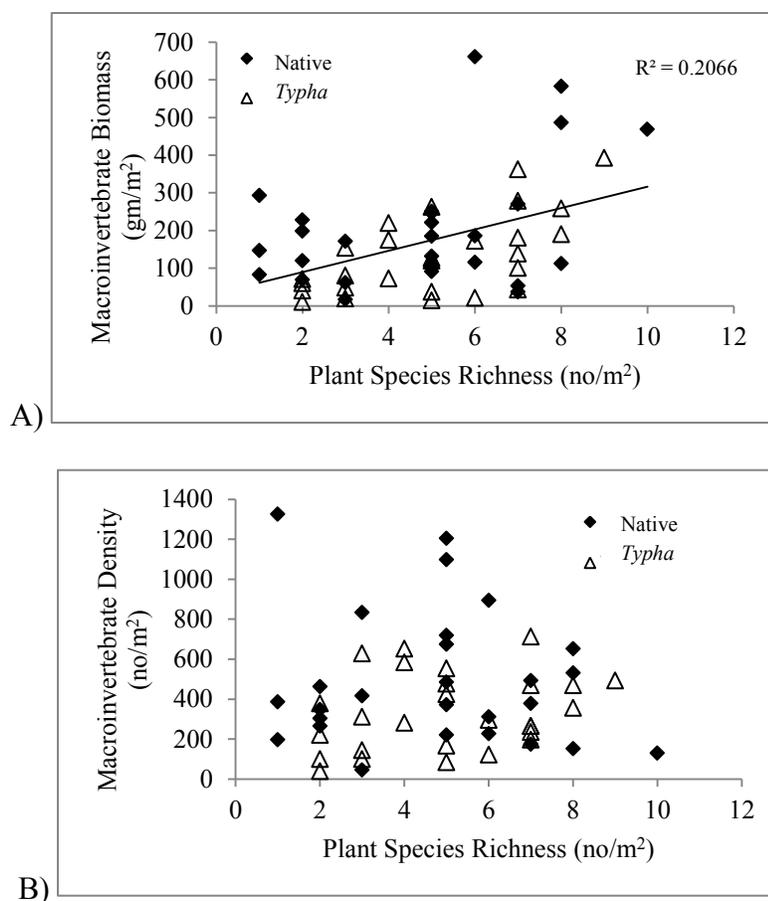
Fig. 6. Least squares regression for mean A) aquatic macroinvertebrate biomass (P = 0.02) and B) aquatic macroinvertebrate density (P = 0.744) with plant species richness as the independent variable.
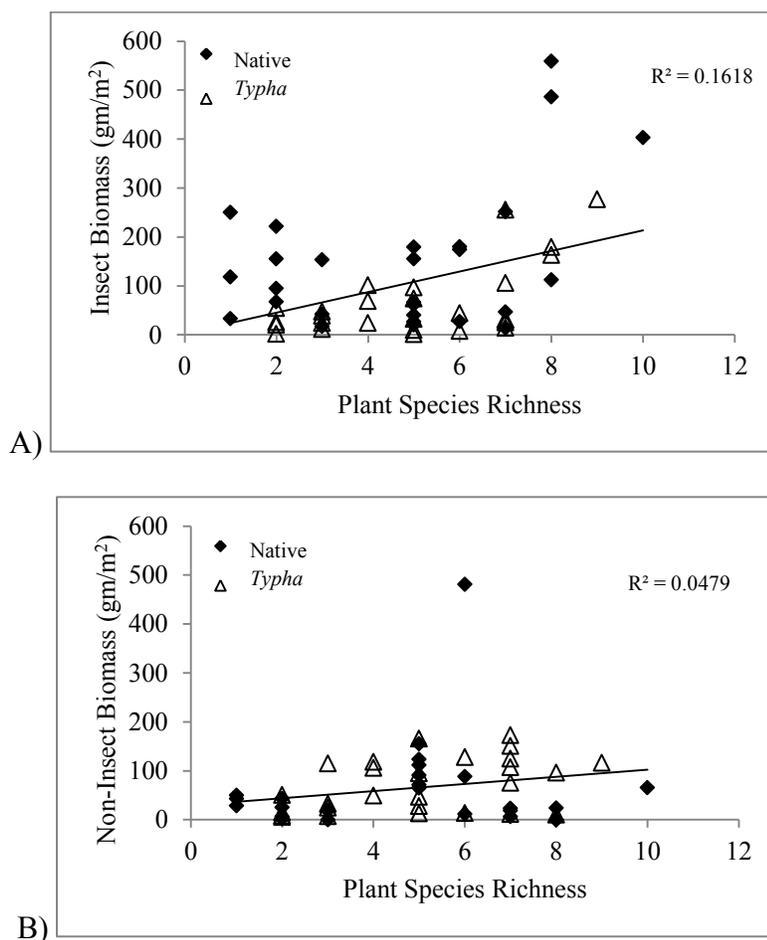
Fig. 7. Least squares regression for A) insect biomass (P = 0.007) and B) non-insect biomass (P = 0.110) with plant species richness as the independent variable.

NMDS ordination of aquatic macroinvertebrate biomass highlights some differences in aquatic macroinvertebrate taxa composition between native and *Typha*-invaded zones (Fig. 8); native plots tend to cluster on the left side of NMDS1, whereas *Typha* emergent plots occupy the right side of the ordination space. Overlaid vectors indicate that plant species richness (R^2=0.209), water temperature (R^2=0.143) and water depth (R^2=0.133) were all significant factors (P < 0.05) impacting the aquatic Macroinvertebrate communities within the two vegetation zones.
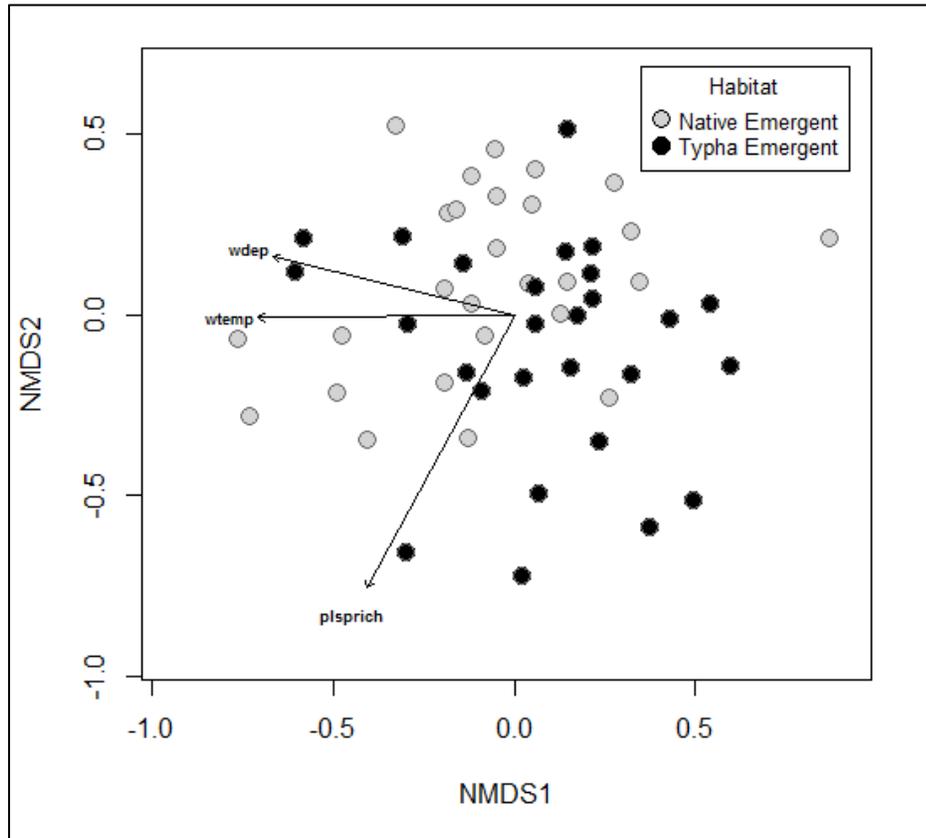
Fig. 8. A 2-dimensional NMDS ordination of 2011 macroinvertebrate species composition data. using Bray-Curtis (Polar) distances. Stress (0.2) indicates weak relationships. Vector overlays are significant and the strength of the relationship is signified proportionally by the length of the vector. Vectors represented are water depth (wdep), water temperature (wtemp) and plant species richness (plsprich).

Discussion

Impacts of Typha Invasion

This study explored whether habitat alterations associated with the presence of invasive *Typha*, including decreased plant diversity and alterations in temperature regimes (see Angeloni et al. 2006, Farrer & Goldberg 2009, Tuchman et a. 2009, Larkin et al. 2011, Mitchell et al. 2011), affect aquatic macroinvertebrate communities within northern Lake Huron coastal wetlands. The presence of *Typha* was associated with shifts in the aquatic macroinvertebrate communities of these wetlands. There were no

consistent differences in environmental variables measured between the native and *Typha*-invaded zones sampled in this study. However, *Typha* stem density was, on average, 30 times greater in *Typha*-invaded areas, leading to vastly different appearances between the two vegetation zones.

Aquatic macroinvertebrate density was consistently lower in *Typha*-invaded emergent marsh zones over all three summer months in which I sampled. Although macroinvertebrate biomass was not significantly different between native and *Typha*-invaded emergent zones, there were noticeably different trends in macroinvertebrate biomass accumulation between the two vegetation zones throughout the summer. In July and August, the mean total macroinvertebrate biomass was lower in *Typha*-invaded areas than in native emergent zones, suggesting that areas invaded by *Typha* may produce fewer food resources throughout the summer growing season for fish and wildlife, and therefore may be less suitable as habitat for these taxa. Two potential reasons for the difference in macroinvertebrate communities between the vegetation zones sampled are a lack of food resources or habitat complexity within the *Typha*-invaded areas. Future research might investigate the algal community within *Typha*-invaded areas, to see if algal taxa or densities occurring there are suitable for a healthy macroinvertebrate community. Also, since *Typha* truly alters the structural complexity of wetland habitats by adding dense layers of litter, and reducing the number of native flora species, it would be intriguing to see how these structural changes affect the aquatic macroinvertebrate community.

To decrease variability between vegetation zones in substrate type, water depth and distance from open water, I sampled *Typha*-invaded areas that are likely recently invaded and therefore closer in proximity to native vegetation zones. In marshes with a longer invasion history, it is possible that I would see much greater effects on the aquatic macroinvertebrate community. Previous studies show that old *Typha* stands, some up to 80 years old, result in stronger changes to the wetland community. Loss of plant species richness, diversity and considerable increases in litter biomass (Tuchman et al. 2009, Mitchell et al. 2011) may further impact macroinvertebrate communities by decreasing dissolved oxygen content, limiting light penetration, reducing available habitat, decreasing algal growth and decreasing available food resources for many macroinvertebrate taxa.

In June 2011, I sampled an area in Cheboygan marsh, where *Typha* has been established for over 40 years (Vail 2009, Lishawa et al. 2010) that exhibits many of the adverse impacts associated with long-term *Typha* presence. Due to below average water levels within the Great Lakes (NOAA 2012), this area did not have any standing water during the months of July and August 2011, making the habitat unsuitable for aquatic macroinvertebrates. However, during the month of June water was present; as such, I collected macroinvertebrate samples that had very low aquatic macroinvertebrate species richness and diversity. These samples only contained Amphipoda, Diptera (mostly Chironomidae), Isopoda, Lumbriculidae and a very small number of Sphaeriidae clams. Over time, it is conceivable that *Typha* in Cheboygan marsh could have an impact on the entire aquatic macroinvertebrate community of the marsh, as it has in this secluded area

that has been dominated by *Typha* for decades. Additionally, because *Typha* plants are so large, typically over 2 meters in height, and produce copious amounts of litter, invaded areas accumulate organic matter at higher rates than native vegetation areas (Vail 2009). With the proper environmental factors in place, organic matter could build up over time and lead to a decrease in water depth within these wetlands, resulting in a loss of habitat for aquatic macroinvertebrates.

Plant Species Richness and Insect Biomass

A common effect associated with *Typha* is a decrease in plant species richness (Mitchell et al. 2011). Although there was no significant difference in plant species richness between native emergent and *Typha*-invaded zones in this study, plant species richness was positively associated with insect biomass and total aquatic macroinvertebrate biomass. However, non-insect biomass did not show similar correlations. The positive association between plant species richness and total macroinvertebrate biomass stemmed primarily from the presence of large-bodied odonates in plots where plant species richness was high (7 to 10 species).

The effects of *Typha* on insect and non-insect biomass are ecologically important due to the differences in life history traits between the two groups. Insects are often more closely associated with individual plants within the water column. Many insects use plant structures for refuge or as perches for various feeding methods. The majority of wetland insects are aquatic in larval life stages, while their adult forms are typically winged terrestrial forms. Therefore, insects are a much more important aspect of allochthonous prey flux (Nakano & Murakami 2001, Fausch 2002) into adjacent non-aquatic

ecosystems than non-insect macroinvertebrates. Decreased plant species richness

attributed to invasion of *Typha* could lead to decreases in prey subsidies available to

terrestrial and aerial predators by decreasing the production of insect biomass.

Water Quality Parameters

I did not observe any differences in water temperature or dissolved oxygen

concentration between the invaded and native vegetation zones I sampled. All wetland

habitats were low in oxygen due to stagnant water and high rates of respiration; however,

there were no differences in these measures between invaded or native vegetation zones.

This suggests that any differences in invertebrate growth rates or diversity were not a

result of these water quality parameters. It is possible that other water quality parameters

such as turbidity, total suspended solids or conductivity may impact the aquatic

macroinvertebrate community (Cardinale 1996). It would be worthwhile to consider these

parameters in future research.

Ecosystem Management Implications

This study demonstrates how the invasion of *Typha* into northern Great Lakes

coastal wetlands affects the aquatic macroinvertebrate community within these

ecosystems. Once invaded by *Typha*, the numerous habitat types characteristic of Great

Lakes coastal wetlands become homogenized, decrease in habitat quality and are

ultimately represented by a single dominant plant community (Mitchell et al. 2012).

When compared to native zones, areas invaded by *Typha* show decreased

macroinvertebrate density, along with a decrease in macroinvertebrate biomass

production throughout the growing season. Along with decreasing habitat quality through

homogenizing the emergent vegetative structural composition of Great Lakes coastal wetlands, *Typha*-invaded habitats might supply fewer suitable macroinvertebrate food resources for these organisms as well. The results of this study support the case for early intervention for and control of *Typha* invasions in Great Lakes coastal wetlands so as to retain and support native flora and fauna whenever possible, and minimize the many negative impacts associated with invasive *Typha*.

CHAPTER THREE:

IMPACTS OF MANUAL REMOVAL OF *TYPHA × GLAUCA* ON THE PLANT AND AQUATIC MACROINVERTEBRATE COMMUNITIES IN NORTHERN GREAT LAKES COASTAL MARSHES

Introduction

Great Lakes coastal wetlands are vital to both the overall health of the Lakes and the many fish and wildlife species that inhabit the region. Coastal wetlands provide breeding grounds and nurseries for many Great Lakes fish species (Jude & Pappas 1992) and marsh breeding birds (Prince et al. 1992, Riffell et al. 2001). In spite of this importance, many Great Lakes coastal wetlands have been lost due to development and agriculture (Batzer et al. 1999). Those remaining are often severely degraded as a result of urban development, nutrient loading and invasive species (Mitsch & Gosselink 2000). Degraded wetlands are frequently dominated by invasive species, making it difficult for many native Great Lakes plant and animal species to survive in these habitats (GLCWC, 2008).

Invasive plants are a major concern in Great Lakes coastal wetlands; once established they can quickly dominate the entire ecosystem. *Typha × glauca* (*Typha*) is one such dominant plant. *Typha* is an invasive hybrid cattail that occurs throughout the Great Lakes region where the native *T. latifolia* and non-native *T. angustifolia* co-occur and cross pollinate (Smith 1987). Once established, *Typha* typically forms dense monotypic clonal stands with the potential to spread via rhizomatous growth up to 4-m in

diameter per year (Boers & Zedler 2008). *Typha* dominated wetlands are associated with

increased plant biomass, litter biomass and soil organic matter, as well as decreased plant

species richness and diversity (Tuchman et al. 2009, Lishawa et al. 2010). *Typha* is a

highly productive species with much greater biomass than the native species it replaces

(Woo and Zedler 2002), resulting in deposition of abundant leaf litter that further

promotes the dominance of *Typha* over other wetland plant species (Farrer and Goldberg

2009, Vaccaro et al. 2009, Larkin et al. 2011).

Coastal wetlands are often intensively managed to reduce the negative effects of

invasive species like *Typha*, and optimize conditions for native wildlife (Sojda & Solberg

1993). Removal of invasive species, and other measures, (i.e. limiting nutrient input), can

lead to increased biodiversity within recovering wetlands (Keddy 2010). Marsh

management strategies are continually being developed, and thus far no single best

practice has emerged. Many management strategies used to control invasive plants may

have damaging effects on non-target organisms. Applying herbicides, a common

management strategy, kills invasive plant species but introduces easily dispersed foreign

chemicals into the target ecosystem. Relyea (2005) found that application of glyphosate,

the most common herbicide used on *Typha*, resulted in a decrease in tadpole species

richness of up to 70%, and likely caused a trophic cascade affecting the predators in the

system as well. Prescribed fire, another common management practice, releases

substantial particulate matter and greenhouse gases into the atmosphere, and is not

always effective in eliminating *Typha* (Apfelbaum 1985).

Macroinvertebrates are often used to evaluate the health of aquatic ecosystems (Niemi et al. 2009, Burton et al. 1999), especially after management or restoration efforts. For instance, De Szalay & Resh (1997) examined how aquatic macroinvertebrates and plants responded to burning and mowing in stands of *Distichlis spicata* in saltwater marshes in California. They found that burned areas contained higher densities of common taxa such as Chironomidae and Corixidae than control areas. There were few significant differences in the macroinvertebrate communities between mowed and control plots, while plant species richness was higher in burned treatment plots as compared with mowed or control areas. Kostecke et al. (2005) studied macroinvertebrate response to *Typha* management at Cheyenne Bottoms, Kansas by observing macroinvertebrate response to four treatments: burning, disking and grazing by 5 and 20 head of cattle on four hectare plots throughout the wetland. There were few differences in macroinvertebrate diversity, biomass or density between the four treatments and the control areas. A similar study (Davis & Bidwell 2008) found few differences in invertebrate communities between wetlands that underwent burning, grazing, mowing and disking treatments, although more heavily vegetated plots were associated with greater macroinvertebrate diversity and biomass. In sum, responses of wetland macroinvertebrates to various management strategies are variable and warrant more detailed study.

The objective of this study was to examine the impact of a sustainable approach to *Typha* management on plant and macroinvertebrate communities within experimental wetland plots. The treatments include mowing with vegetation removal, mowing and

leaving felled litter and plants and modified tilling (i.e. removing below-ground rhizomes). I aimed to determine 1) how the treatments impacted physicochemical parameters, 2) how the plant community responded to the implemented management techniques and 3) what impacts our treatments had on the aquatic macroinvertebrate community. Due to increased soil light availability, I hypothesized that the removal of *Typha* stems and litter would allow native plants to grow. I also predicted that aquatic macroinvertebrate communities would differ among treatments, as the various techniques implemented would likely create different wetland micro-habitats.

Methods

Study Sites

I tested the effect of *Typha* harvest treatments in two marshes located in northern Michigan. Munuscong Marsh is at the mouth of the Munuscong River which flows into the St. Marys River at the eastern end of the Upper Peninsula of in Michigan. Cedarville Marsh is an open embayment marsh located in Cedarville Bay in the Les Cheneaux Islands, also in the eastern Upper Peninsula. Due to their close proximity, both wetlands experience similar climate and exhibit similar vegetative communities and soils (Albert 2005). Both wetlands have also been invaded by *Typha* and contain large areas dominated by the plant (Dennis Albert, personal communication).

During July and August 2011, our research team implemented three different *Typha* harvest treatments in 24, 4-m^2 plots at both study sites. We carried out treatments in 12 plots within a sedge meadow habitat and 12 plots in an emergent marsh habitat within each site. The three treatments were 1) mowing and removing all cut vegetation

 and litter from within the plot (cut and remove), 2) mowing and leaving felled litter within the plot (cut and leave), and 3) modified tilling by removing both above ground plant biomass and belowground rhizomes from within the plot (below ground). We also had three control plots in each wet meadow and emergent marsh habitat. The cut and leave treatment was intended to mimic a mowing treatment, a common management technique for invasive plants. The below ground treatment is generally not used within the Great Lakes basin, but has been used in other wetland complexes (Osland et al. 2011). I was interested in how the plant and macroinvertebrate communities would respond to these types of management practices in a subset of plots, namely those located in the emergent marsh habitat.

Plot Characterization

I characterized the vegetation by randomly placing a 1-m^2 quadrat in the treatment plot and conducting stem counts for each plant species. Within the quadrat, I also estimated percent cover of litter and moss within each of the 1-m^2 quadrats. I measured water depth to the nearest 0.5-cm, and also measured water temperature and dissolved oxygen concentration using a Hach dissolved oxygen probe (Hach co. Loveland, CO Model HQ30d).

Aquatic Macroinvertebrate Collection

I collected aquatic macroinvertebrates at the 12 edge plots in both marshes during the months of June, July and August 2012 using a stovepipe collection method following Gathman & Burton (2011). Within the 1-m^2 quadrat used to collect plant data, I collected an invertebrate sample by forcing a 14.5-cm diameter stovepipe through the water

column and down approximately 5-cm into the sediment. Once the stovepipe was in place, I scooped out any loose vegetation and placed it into a 300-µm sieve. I then agitated the water within the stovepipe to loosen the top 2- to 3-cm of sediment and used a syphon-pump to extract the water and invertebrates out of the pipe and into the sieve.

To standardize biomass and species richness calculations, each stovepipe was pumped 100 times, regardless of water depth; I established that 100 total pumps was sufficient to extract a majority of the invertebrates within the stovepipe. If fewer than 100 pumps would empty the stovepipe, I stopped pumping to allow water to seep back into the stovepipe before continuing. After the invertebrates were pumped into the sieve, I transferred them to collection jars and transported them back to the laboratory in a cooler.

I processed the stovepipe samples by rinsing them through a 300-µm sieve to decrease the amount of fine sediment and particulate matter in the sample. Once rinsed, I placed each sample into a white enamel pan and picked out all visible macroinvertebrates. I considered samples complete after a five minute interval without finding any additional organisms. All invertebrates were stored in 95% EtOH.

I identified each invertebrate to the lowest taxonomic unit possible using Merritt, Cummins & Berg (2008) for insect specimens and Thorp & Covich (1991) for non-insect specimens. I identified Chironomidae to sub-family or tribe using Merritt, Cummins & Berg (2008). To utilize length-mass regressions to calculate biomass, I measured the body length of each individual organism from the anterior tip of the head to the posterior tip of the abdomen. Length-mass regressions are ideal for calculating biomass as they are faster and more precise than direct weighing or biovolume calculations, especially

considering most organisms lose dry mass when preserved (Benke et al. 1999). I used

established length-mass regressions at the lowest possible taxonomic unit available (see

Benke et al. 1999, Baumgartner & Rothhaaupt 2003, Miserendino 2001, Hall et al. 2006,

Stead et al. 2003, Stoffels et al. 2003, Edwards et al. 2009, Gruner 2003).

Statistical Analysis

I used repeated measures ANOVA in R (Version 2.15.1, R Development Core

Team 2012) to compare environmental variables (water depth, *Typha* stem density, water

temperature, percent litter cover, and dissolved oxygen) among plots, and

macroinvertebrate variables (density, biomass and Shannon-Wiener diversity index) for

differences among treatments, sites and months. I used repeated measures ANOVA

because monthly samples from the same plot were not independent from one another. For

each analysis I used site, treatment and site × treatment as between-sample variables;

while month, month × site, month × treatment and month × site × treatment were within-

sample variables. To meet assumptions of normality, I transformed all biomass data using

log (x + 1) transformation prior to statistical analysis (McCune & Grace 2002).

To compare plant and aquatic macroinvertebrate communities among treatments

and with multiple environmental variables, I conducted Nonmetric Multidimensional

Scaling (NMDS) ordination. I ordinated sample units from all three months in species

space using R (Version 2.15.1, R Development Core Team 2012).

Results

Plot Characterization

There were no significant differences in water temperature or *Typha* density

between sites (P > 0.05). However, water depth, dissolved oxygen concentration, and percent litter cover were significantly different between sites (P < 0.05). Mean water depth at Munuscong marsh (32-cm) was nearly twice that of the mean depth in Cedarville Marsh (13.2-cm), whereas mean dissolved oxygen concentration was more than twice as high in Cedarville Marsh (60.6 % saturation) as compared with Munuscong Marsh (28.2 % saturation).

Several environmental characteristics differed among treatments (Table 2). Water depth was significantly deeper in the below ground removal plots in comparison with other treatments (P = 0.003). Number of *Typha* stems was also significantly lower in treatment plots in comparison with control treatments (P < 0.001). Over the three months I sampled, water temperature and dissolved oxygen concentration did not vary significantly between the four treatments (P > 0.05). Percent litter cover was significantly less in cut & remove and below treatments when compared with cut & leave and control treatments (P < 0.05).

Table 2. Environmental measurements for the four treatments at Cedarville (Ced) and Munuscong (Mun) marshes. Significant differences between sites were evident in water depth, dissolved oxygen concentration (DO%) and litter cover. Values are the means for that treatment type at each site averaged over the three months in which samples were collected. Standard error is shown in parentheses. Comparisons at each site are represented by letters; means with the same letter are not significantly different (P > 0.05). n = 9.

		Cut & Leave	Cut & Remove	Below	Control
Ced	Water Temp (˚C)	21.9^a(1.27)	23.4^a(1.24)	21.7^a(1.09)	21.8^a(1.27)
	Water Depth (cm)	11.5^{ab}(4.43)	9.1^{ab}(3.61)	23.7^b(3.81)	8.6^a(3.06)
	DO (%)	64.3^a(19.94)	79.75^a(16.97)	50.7^a(5.31)	50.8^a(15.89)
	Typha stems (#)	3.0^b(0.85)	3.8^b(0.87)	0.1^b(0.11)	23.4^a(2.95)
	% Litter Cover	29.8^{ab}(12.43)	0.1^a(0.11)	0.1^a(0.11)	54.4^b(11.79)
Mun	Water Temp (˚C)	23.3^a(0.34)	23.3^a(0.79)	22.5^a(0.48)	21.9^a(0.79)
	Water Depth (cm)	28^a(1.72)	21.7^a(2.13)	29.2^a(2.40)	25.1^a(2.16)
	DO (%)	10.0^a(4.5)	57.7^b(11.13)	30.9^{ab}(7.63)	14.4^a(2.46)
	Typha stems (#)	0.6^a(0.56)	4.6^a(2.99)	0.6^a(0.34)	32.2^b(4.60)
	% Litter Cover	80.7^b(11.36)	5.1^a(2.87)	16.1^a(6.38)	53.8^b(13.87)

Plant Community

There were significant differences in plant species diversity between treatments; cut and remove treatments had higher diversity than below ground treatments, however no treatment differed significantly from the control (P = 0.017, Fig. 9 A). Plant diversity was not significantly different between sites (P > 0.05). The cut and remove treatment yielded the highest plant diversity, followed by control, then cut and leave; below ground removal plots showed the lowest plant diversity. The proportion of native to invasive (i.e. *Typha* and *Hydrocharis morsus-ranae*) was altered differently by each of the treatments

(Fig 9 B). Generally, the treatments at both sites resulted in different plant communities
(Fig. 10). The below ground treatments promoted fewer emergent plants and were
generally colonized by submergent plant species such as *Utricularia* spp. and
Myriophyllum spp. The cut and remove treatments were dominated by small native
emergent plants such as *Juncus* spp., *Eleocharis* spp. and *Schoenoplectus* spp. Similarly,
cut and leave treatments contained many of the same species as cut and remove
treatments, but in much lower densities. Control treatments were largely dominated by
Typha.
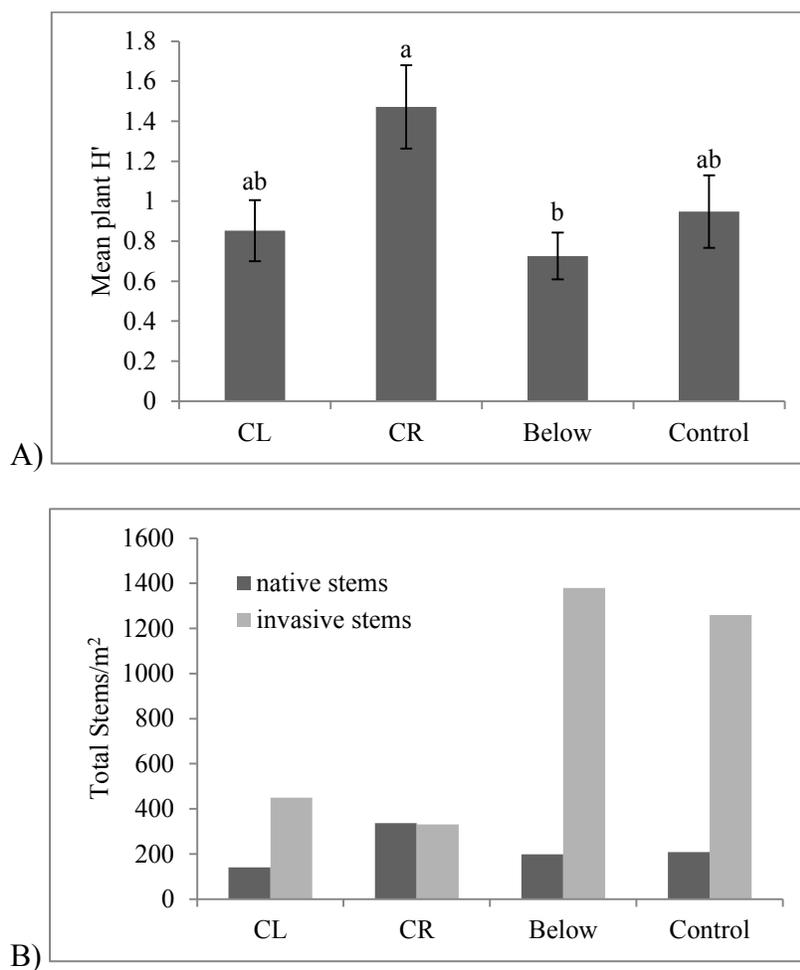
Fig. 9. A) Mean plant diversity (Shannon-Wiener index, H') for the month of August 2012 in cut & leave (CL), cut & remove (CR), below ground removal (Below) and control plots. Error bars represent ± 1 SE. Only data for the month of August are shown, as plant diversity peaked in August. Comparisons are represented by letters; means with the same letters are not significantly different (P > 0.05). B) Total stems represented by native and invasive species in the month of August 2012 for each of the treatments.
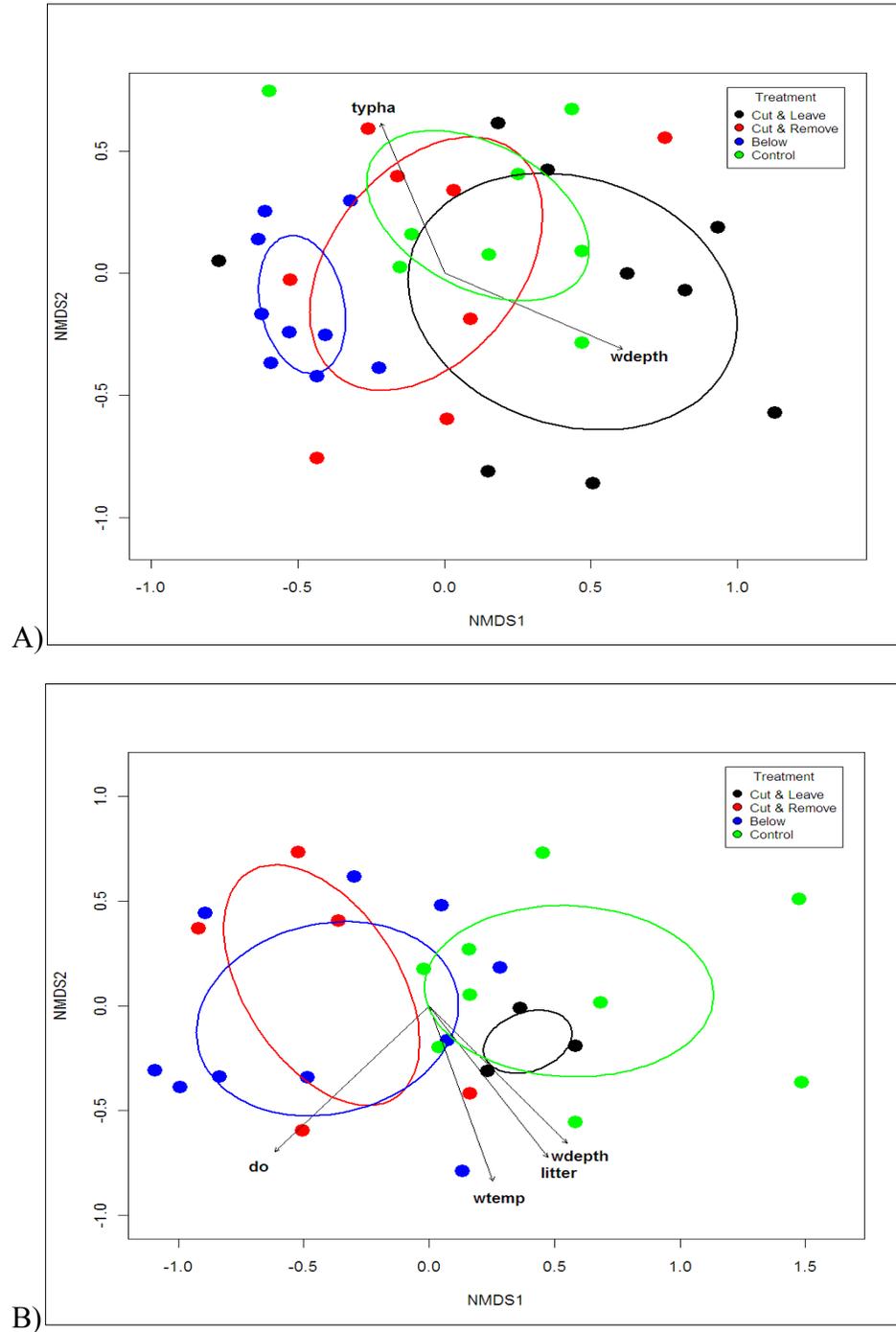
Fig. 10. NMDS ordination of 2012 plant species composition data for A) Cedarville Marsh and B) Munuscong Marsh. Stresses for Cedarville (0.2) and Munuscong (0.2) signify weak relationships. Ordination is measured in two dimensions, and dissimilarity was based on Bray-Curtis (Polar) distances. Vector overlays are significant ($P < 0.05$) and the strength of the relationship of is signified proportionally by the length of the vector. Ellipses represent the standard deviation around the mean for each treatment.

Macroinvertebrate Community

Mean macroinvertebrate biomass, Shannon-Wiener diversity index, and

macroinvertebrate density were not significantly different (P > 0.05) between sample

sites or the four experimental treatments (Table 3 A-C). Insect biomass made up a much

smaller proportion of the total biomass in all four treatments than non-insect biomass;

this was especially true for the cut and leave, and cut and remove treatments (Fig. 11).

See Appendix A for a list of all taxa identified.

Table. 3. Mean (± 1 SE) aquatic macroinvertebrate biomass, density and Shannon-Wiener diversity index (H') for cut & leave, cut & remove, below ground removal (below) and control plots. Comparisons are indicated by letters; means with the same letter are not significantly different (P > 0.05). n = 18.

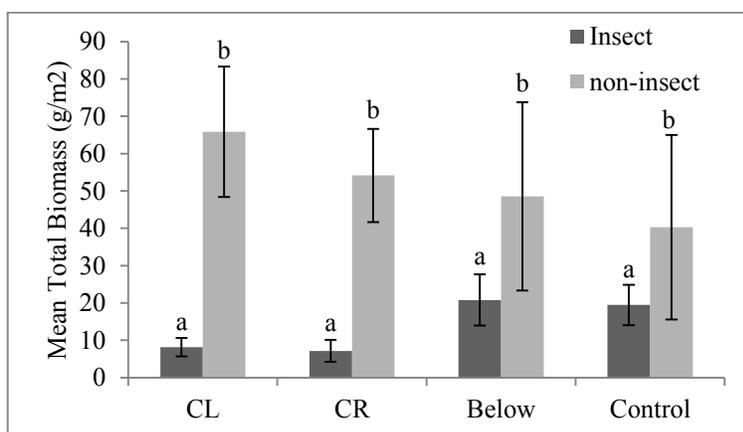|	Cut & Leave	Cut & Remove	Below	Control
Biomass (g/m^2)	560.39(128.5)^a	463.97(102.8)^a	525.06(207.9)^a	452.36 (210.0)^a
Density (no/m^2)	650.55(136.0)^a	577.59(91.7)^a	468.54(67.7)^a	477.89(72.2)^a
H'	1.23(0.1)^a	1.22(0.1)^a	1.17(0.1)^a	1.28(0.1)^a

Fig. 11. Mean total biomass of insect and non-insect organisms for cut & leave (CL), cut & remove (CR), below ground removal (below) and control plots. Error bars represent ± 1 SE. Comparisons are within each group of invertebrates (insect or non-insect); means with the same letter are not significantly different (P > 0.05).

Shredder biomass was significantly greater ($P < 0.05$) in control plots than the

three other treatment types, although shredder biomass represented less than 1% of the

total macroinvertebrate biomass I collected. All other functional feeding groups did not

vary significantly ($P > 0.05$) between the four treatments (Fig. 12). NMDS ordination of

aquatic macroinvertebrate biomass did not show clear differences in aquatic

macroinvertebrate community composition between the four treatments. Overlaid vectors

indicated that water depth ($r^2=0.263$) and plant diversity ($r^2=0.096$) accounted for a

significant ($P < 0.05$) portion of the variation among macroinvertebrate communities

(Fig. 13).
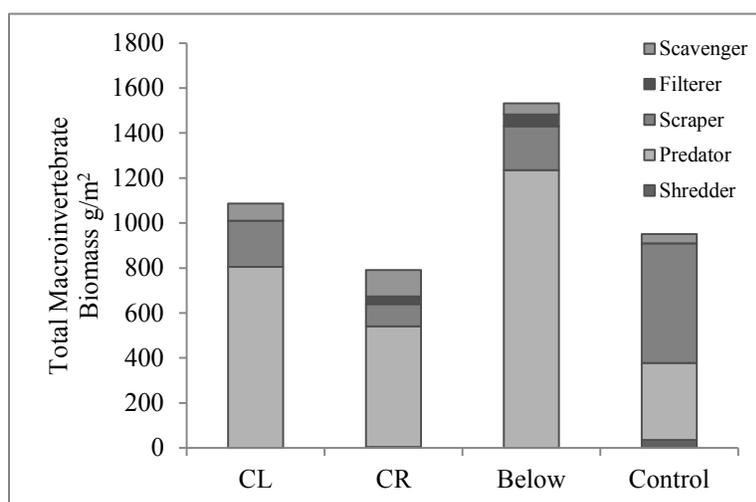
Fig. 12. Total biomass collected from each plot type for June, July and August 2012 represented by functional feeding groups. Total biomass collected is pooled from both study sites and shown for cut & leave (CL), cut & remove (CR), below (Below) and control (Control) plot types.
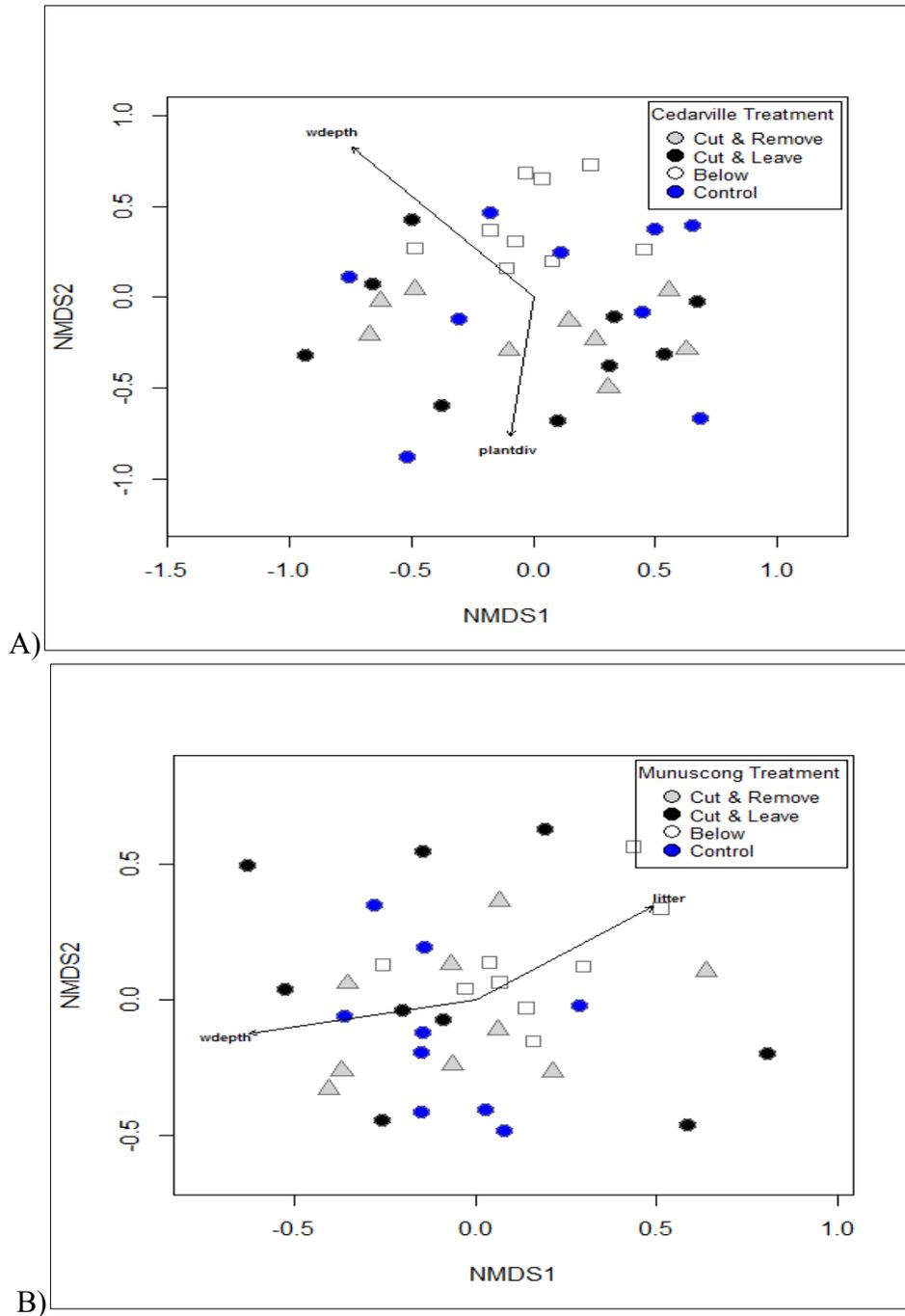
Fig. 13. NMDS ordination of 2012 macroinvertebrate species composition data for A) Cedarville Marsh and B) Munuscong Marsh. Stress for Cedarville (0.2) and Munuscong (0.2) indicate weak relationships. Ordination is measured in two dimensions, and dissimilarity was based on Bray-Curtis (Polar) distances. Vector overlays are significant and the strength of the relationship of is signified proportionally by the length of the vector.

Mean Diptera biomass (Fig 14), predominantly represented by the family Chironomidae, was significantly greater in below-ground treatments than any other treatment ($P = 0.001$). Diptera biomass was also significantly greater in July in comparison with June and August ($P < 0.001$); the treatment × date interaction was the only significant interaction ($P = 0.001$). No other macroinvertebrate taxa differed significantly among treatments.
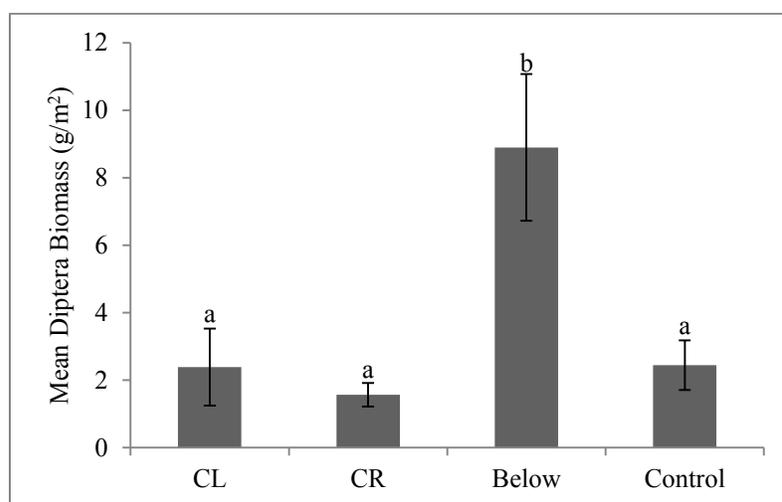

Fig. 14. Mean Diptera biomass collected from the four experimental plot types during June, July and August 2012. Error bars represent ± 1 SE. Comparisons are indicated by letters; means with the same letter are not significantly different ($P > 0.05$).

Discussion

Effective management practices are essential to maintaining the health of Great Lakes coastal wetlands. The purpose of this study was to determine how methods for *Typha* harvest treatments within Great Lakes coastal wetlands 1) impacted physicochemical parameters, 2) altered the plant community and 3) affected

macroinvertebrate communities. We found that our methods altered the plant community while having few significant effects on the standing stock of aquatic macroinvertebrates in two coastal wetlands in northern Michigan. The treatments we implemented in this study demonstrate how diverse management strategies may be used to produce different ecological outcomes.

Cut and Remove Treatment

The cut and remove treatment mimicked a large-scale biomass harvest; all of the cut plant biomass and litter were removed from the plot. This treatment increased plant diversity, although not significantly. However, plant response was most likely due to the increased light penetration and soil temperature as a result of removing the shade producing *Typha* litter (Farrer & Goldberg 2009, Vaccaro et al. 2009, Larkin et al. 2011). It appears that the seed bank was stimulated by removing *Typha* litter allowing many of the smaller native species to germinate. It should be noted that water levels in the Great Lakes, especially Lakes Michigan and Huron, during the summer of 2012 were nearly 1-m lower than the historic average (NOAA 2012). As a result, many of the treatment plots had little to no standing water and small emergent plants like *Juncus* spp. and *Eleocharis* spp. were able to propagate. This is consistent with Keddy & Reznicek (1986), who found that low water levels within the Great Lakes create opportunities for buried seeds to germinate. Although not significant, cut and remove plots produced the lowest aquatic macroinvertebrate biomass throughout the summer. Removal of all plant material, including litter, from these plots left relatively little habitat for aquatic macroinvertebrates to colonize. It is likely that the plant community, made up of small emergent plants such

as *Eleocharis* sp. and *Juncus* sp. did not present suitable habitat for many aquatic macroinvertebrates which use plants as both hunting and hiding grounds.

Cut and Leave Treatment

The cut and leave treatment mimicked a common *Typha* management technique where stands of *Typha* are mowed and the cut plants and litter are left on the soil or water surface. Likely because the felled plants and litter covered the soil, we saw slightly reduced plant diversity in these plots as compared to the cut and remove treatments, though the difference was not significant. These plots also harbored the most aquatic macroinvertebrate biomass, although not significantly more than the other three treatments. The difference in macroinvertebrate biomass was most likely due to the high density and biomass of leeches (Hirudinea) within the cut and leave plots. Leeches are significantly larger than any of the other macroinvertebrates I collected and Leech biomass in cut and leave plots was at least four times greater than any other group found within these or any other plot type. The floating felled litter (over open water) may have created desirable habitat for leeches, which can frequently be found clinging to floating plant material near the water surface.

Below Ground Treatment

Open pools developed in below ground removal plots where *Typha* rhizomes were pulled from the soil. This mimicked hemi-marsh conditions (50% open water, 50% vegetated wetland habitat), which are a frequently targeted outcome for *Typha* marsh management to enhance habitat value for waterfowl and marsh breeding birds (Sojda & Solberg 1993). Generally, aquatic macroinvertebrates were not negatively affected by our

below ground removal treatment, potentially allowing for these areas to make suitable habitat for fish and marsh birds.

Below ground treatment resulted in very different environmental and floristic responses than either of the cut treatments. The water was deeper in this treatment, ostensibly because of the significant amount of biomass (*Typha* rhizomes) removed from the soil. Because of the deep water, few emergent plant species grew in these treatment plots. Instead, the plant community was represented by submergent species such as *Myriophyllum* spp., *Utricularia* spp., and *Nymphaea odorata.*

Diptera (represented primarily by Chironomidae) had significantly higher biomass and density in the below ground removal treatment. The high dipteran biomass could be explained by the open water conditions of these treatments, as chironomid adults are attracted to open water habitats to lay their eggs (Merritt, Cummins & Berg 2008). The below ground treatment implementation also left the sediments fairly unconsolidated and many of the Chironomid species I collected burrow into the sediments as larvae, where they remain until the pupal phase. As such, these loose sediments may be ideal for many chironomid species, which are an especially important food resource for many fish and bird species within Great Lakes coastal wetlands (Jude & Pappas 1992, Kostecke et. al 2005).

Implications of Plot Size

The relatively small size (4 × 4-m) of the treatment plots may have lessened the potential impact of our treatments on the aquatic macroinvertebrate communities. Studies examining the impacts of wetland restoration on aquatic macroinvertebrates are generally

conducted in larger plots, anywhere from 10-m× 10-m plots to areas as large as 16

hectares (DeSzalay & Resh 1997, Kostecke et al. 2005, Davis & Bidwell 2008). It is

conceivable that one year after management implementation the aquatic

macroinvertebrate community would be more variable in larger plots due to different

colonization strategies and rates of wetland macroinvertebrates. For example treatments

implemented at larger scales may have a greater proportion of aerial dispersing

macroinvertebrates such as Odonates, while lacking in slowly dispersing organisms such

as Gastropods (Brown et al. 1997, Brady et al. 2002).

Management Implications

The primary reason for advocating the management strategies used in this study is

to reduce pollution and promote sustainability in wetland management. The aquatic

macroinvertebrate sampling I completed was part of a larger project investigating

sustainable wetland restoration. Many management and restoration strategies use

destructive practices, including herbicide application and fire, to control invasive plant

species. Along with non-polluting plant removal, this project is exploring the feasibility

of converting invasive plant species, like *Typha*, into energy by turning harvested

biomass into methane gas through anaerobic digestion. Turning invasives into energy is

one way in which we can ensure ecosystem restoration leans toward carbon neutrality.

Wetland management practices are continually being improved upon, and sustainable

restoration is a logical next step. The potential to produce clean energy in the form of

methane gas could mean that this management strategy produces a larger net benefit than

many of the other management strategies currently in practice and make these methods

more desirable in an energy conscious world.

APPENDIX A

LIST OF AQUATIC MACROINVERTEBRATE TAXA

Amnicola sp.

Anax sp.

Armiger crista

Aquarius sp.

Atrichopogon sp.

Bacobtrella phalera

Belastoma sp.

Bezzia sp.

Bithynia tentaculata

Buenoa sp.

Caecidotae sp.

Caenis sp.

Carabidae

Ceratapagonidae pupa

Chironominae pupa

Chironomini

Chrysops sp.

Chrysomelidae

Coleoptera

Colymbetes sp.

Corisella sp.

Corixinae

Crangonyx sp.

Culex sp.

Cyphon sp.

Dasyhelea sp.

Derovatellus sp.

Dina sp.

Diptera pupa

Dolichopodidae

Dytiscidae

Eclipidrilus sp.

Entombrynidae

Empididae

Enallagma sp.

Enochus sp.

Erpobdellidae

Ferrissia sp.

Fossaria sp.

Glossiphonia
     complanata

Gyraulus sp.

Helobdella elongata

Helobdella papillata

Helobdella stagnalis

Helobdella triserialis

Hesperocorixa sp.

Hyalella sp.

Hydracarina

Hydrophillidae

Lampyridae

Lepidoptera

Leptoceridae

Leucorrhinia sp.

Libellulidae

Limnephilidae

Lirceus sp.

Lumbriculus sp.

Lymnea sp.

Mansonia sp.

Menetus sp.

Miathyria sp.

Micronecta sp.

Microvelia sp.

Moorebdella sp.

Naididae

Nehalennia sp.

Nematoda

Notiphila sp.

Notonecta sp.

Notonectadae

Odontomyia sp.

Oecetis sp.

Orthocladiinae

Paraplea sp.

Parapoynx sp.

Physella sp.

Pilaria sp.

Placobdella sp.

Placobdella papillifera

Placobdella ornata

Planorbella sp.

Planorbula sp.

Polycentropus sp.

Promenetus sp.

Rhantus sp.

Semicerura sp.

Sphaeridae

Staphlynidae

Stlyaria lacustris

Stylodrilus heringianus

Sympetrum sp.

Syrphidae

Tanypodinae

Tanytarsini

Trichocorixa sp.

Trichoptera

Tropisternus sp.

Tubificidae

Veliidae

REFERENCE LIST

Albert, D. A., D. A. Wilcox, J. Ingram, and T. A. Thompson. 2005. Hydrogeomorphic classification for Great Lakes coastal wetlands. *Environmental Science and Biology Faculty Publications*. Paper 50.

Anderson, A. M., D. A. Haukos and J. T. Anderson. 1999. Diet composition of three anurans from the playa wetlands of northwest Texas. *Copeia* 1999: 515-520.

Apfelbaum, S. 1985. Cattail (*Typha* spp.) management. *Natural Areas Journal* 5: 9-17.

Balcombe, C. K., J. T. Anderson, R. H. Fortney, and W. S. Kordek. 2005. Aquatic macroinvertebrate assemblages in mitigated and natural wetlands. *Hydrobiologia* 541:175-188.

Batzer, D. P., R. B. Rader, and S. A. Wissinger (eds). 1999. *Invertebrates in Freshwater Wetlands of North America: Ecology and Management*. John Wiley & Sons, INC. New York, New York, USA.

Baumgärtner, D. and K. Rothhaupt. 2003. Predictive length-dry mass regressions for freshwater invertebrates in a pre-alpine lake littoral. *International Review of Hydrobiologia* 88:453-463.

Benke, A. C., A. D. Huryn, L. A. Smock, and J. B. Wallace. 1999. Length-mass relationships for freshwater macroinvertebrates in North America with particular reference to the southeastern United States. *Journal of North American Benthological Society* 18:308-343.

Blossey, B., L. C. Skinner, and J. Taylor. 2001. Impact and management of Purple Loosestrife (*Lythrum salicaria*) in North America. *Biodiversity and Conservation* 10:1787-1807.

Boers, A. M., and J. B. Zedler. 2008. Stabilized water levels and *Typha* invasiveness. *Wetlands* 28:676 – 685.

Brown, S. C., K. Smith, and D. Batzer. 1997. Macroinvertebrate responses to wetland restoration in northern New York. *Community and Ecosystem Ecology* 97:1016-1024.

Burton, T. M., D. G. Uzarski, J. P. Gathman, J. A. Genet. 1999. Development of a preliminary invertebrate index of biotic integrity for Lake Huron coastal wetlands. *Wetlands* 19:869-882.

Burton, T. M., C. A. Stricker, D. G. Uzarski. 2002. Effects of plant community composition and exposure to wave action on invertebrate habitat use of Lake Huron coastal wetlands. *Lakes & Reservoirs: Research and Management* 7:255-269.

Burton, T. M. and D. G. Uzarski. 2009. Biodiversity in protected coastal wetlands along the west coast of Lake Huron. *Aquatic Ecosystem Health & Management* 12:63 – 76.

Cardinale, B. J. 1996. The effects of a pelagic-littoral mixing gradient on an epiphytic invertebrate community. Master's Thesis, Michigan State University.

Christensen, J. R., and W. G. Crumpton. 2010. Wetland invertebrate community responses to varying emergent litter in a prairie pothole emergent marsh. *Wetlands* 30:1031-1043.

Cole, R. A. and D. L. Wiegman. 1983. Relationships among zoobenthos, sediments, and organic matter in littoral zones of western Lake Erie and Saginaw Bay. *Journal of Great Lakes Research* 9:568-581.

Conway, C. J. 2009. Standardized North American marsh bird monitoring protocols, version 2009-2. Wildlife Research Report #2009-2. U.S. Geological Survey, Arizona Cooperative Fish and Wildlife Research Unit, Tucson, AZ.

Cvetkovic, M., A. Wei, and P. Chow-Fraser. 2010. Relative importance of macrophyte community versus water quality variables for predicting fish assemblages in coastal wetlands of the Laurentian Great Lakes. *Journal of Great Lakes Research* 36:64-73.

Davis, C. A., and Bidwell, J. R. 2008. Response of aquatic invertebrates to vegetation management and agriculture. *Wetlands* 28:793-805.

de Szalay F. A., and V. H. Resh. 1997. Responses of wetland invertebrates and plants important in waterfowl diets to burning and mowing of emergent vegetation. *Wetlands* 12:149-156.

Dvořaki, J., and E. P. H. Bestz. 1982. Macro-invertebrate communities associated with the macrophytes of Lake Vechten: structural and functional relationships. *Hydrobiologia* 95:115-126.

Edwards, F. K., R. B. Lauridson, L. Armand, H. M. Vincent, I. J. Jones. 2009. The relationship between length, mass, and preservation time for three species of freshwater leeches (Hirudinea). *Fundamental and Applied Limnology* 173:321-327.

Ehrenfeld, J. G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. *Ecosystems* 6:503 – 523.

Farrer, E.C. and D.E. Goldberg. 2009. Litter drives ecosystem and plant community changes in cattail invasion. *Ecological applications* 19:398-412.

Fausch, K. D., M. E. Power, and M. Murakami. 2002. Linkages between stream and forest food webs: Shigeru Nakano's legacy for Ecology in Japan. *TRENDS in Ecology and Evolution* 17:429-434.

Freyman, M. J. 2008. The effect of litter accumulation of the invasive cattail *Typha × glauca* on a Great Lakes coastal marsh. Master's Thesis. Loyola University Chicago.

Frid. C.L.J., W. U. Chandrasekara, and P. Davey. 1999. The restoration of mud flats invaded by common cord-grass (*Spartina anglica*, CE Hubbard) using mechanical disturbance and its effects on the macrobenthic fauna. *Aquatic Conservation: Marine and Freshwater Ecosystems* 9:47-61.

Galatowitsch, S. M., N. O. Anderson, and P. D. Ascher. 1999. Invasiveness in wetland plants in temperate North America. *Wetlands* 19:733-755

Gardner, S. C., C. E. Grue, W. W. Major III, and L. L. Conquest. 2001. Aquatic invertebrate communities associated with Purple Loosestrife (*Lythrum salicaria*), Cattail, (*Typha latifolia*), and Bulrush (*Scirpus acutus*) in Central Washington, USA. *Wetlands* 21:593-601.

Gathman, J. P., and T. M. Burton. 2011. A Great Lakes coastal wetland invertebrate community gradient: relative influence of flooding regime and vegetation zonation. *Wetlands* 31:329-341.

Gray, M. J., R. M. Kaminski, G. Weerakkody, B. D. Leopold, and K. C. Jensen. 1999. Aquatic invertebrate and plant responses following mechanical manipulations of moist-soil habitat. *Wildlife Society Bulletin* 27:770-779.

Burton, T. M. et al. (eds). 2008. Great Lakes Coastal Wetlands Monitoring Plan. Great Lakes Coastal Wetlands Consortium. Available via web. http://www.glc.org/wetlands/final-report.html.

Gruner, D. S. 2003. Regressions of length and width to predict arthropod biomass in the Hawaiian Islands. *Pacific Science* 57:325-336.

Hall, R. O., M. F. Dybdahl, M. C. Vanderloop. 2006. High secondary production of introduced snails in rivers. *Ecological Applications* 16:1121-1131.

Holomuzki, J. R., and D. M. Klarer. Invasive reed effects on benthic community structure in Lake Erie coastal marshes. *Wetlands Ecology and Management* 18:219-231.

Huener, J. D. and J. A. Kadlec. 1992. Macroinvertebrate response to marsh management strategies in Utah. *Wetlands* 12:72-78.

Jude, D. J., and J. Pappas. 1992. Fish utilization of Great Lakes coastal wetlands. *Journal of Great Lakes Research* 18:651-672.

Kashian, D. R., and T. M. Burton. 2000. A comparison of macroinvertebrates of two Great Lakes coastal wetlands: testing potential metrics for an index of ecological integrity. *Journal of Great Lakes Research* 26:460-481.

Keddy, P. A. and A. A. Reznicek. 1986. Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. *Journal of Great Lakes Research* 12:25-36.

Keddy, P. A. 2010. Wetland Ecology, 2nd Edition. Cambridge University Press. New York, NY.

Kostecke, R. M., L. M. Smith, and H. M. Hands. 2005. Macroinvertebrate response to cattail Management at Cheyenne Bottoms, Kansas, USA. *Wetlands* 25:758-763.

Krecker, F. H. 1939. A comparative study of the animal population of certain submerged aquatic plants. *Ecology* 20:553-562.

Krull, J. N. 1979. Aquatic plant-macroinvertebrate associations and waterfowl. *Journal of Wildlife Management* 34:707-718.

Larkin, D. J., M. J. Freyman, S. C. Lishawa, P. Geddes, N. C. Tuchman. 2011. Mechanisms of dominance by the invasive hybrid cattail *Typha × glauca*. *Biological Invasions* 14:65-77.

Lishawa, S. C., D. A. Albert, N. C. Tuchman. 2010. Water level decline promotes Typha × glauca establishment and vegetation change in Great Lakes coastal wetlands. *Wetlands*. 30:1085-1096.

Levin, L. A., C. Neira, and E. D. Grosholz. 2006. Invasive cordgrass modifies wetland trophic function. *Ecology* 87:419-432.

Lonsdale, W. M. 1999. Global patterns of plant invasions and the concept of invasibility. *Ecology* 80:1522-1536.

McCune, B. and J. B. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR, U.S.A.

Merritt, R. W., M. E. Benbow, P. L. Hudson. 2002. Wetland macroinvertebrates of Prentiss Bay, Lake Huron, Michigan: diversity and functional group composition. *The Great Lakes Entomologist* 35:149-160.

Merritt, R. W., K. W. Cummins, and M. B. Berg. 2008. An Introduction to the Aquatic Insects of North America, 4th edition. Kendall and Hunt Publishing Company, Dubuque, IA, USA.

Miserendino, M. L. Length-mass relationships for macroinvertebrates in freshwater environments of Patagonia (Argentina). *Ecología Austral* 11:3-8.

Mitchell, M. E., S. C. Lishawa, P. Geddes, D. J. Larkin, D. Treering, and N. C. Tuchman. 2011. Time-Dependent impacts of cattail invasion in a Great Lakes coastal wetlands complex. *Wetlands* 31:1143-1149.

Nakano, S. and M. Murakami. 2001. Reciprocal subsidies: dynamic interdependence between terrestrial and aquatic food webs. PNAS 98:166-170.

Niemi, G. J., V. J. Brady, T. N. Brown, J. J. H. Ciborowski, N. P. Danz, D. M. Ghioca, J. M. Hanowski, T. P. Hollenhorst, R. W. Howe, L. B. Johnson, C. A. Johnston, E. D. Reavie. 2009. Development of ecological indicators for the U.S. Great Lakes coastal region – a summary of application in Lake Huron. *Aquatic Ecosystem Health & Management* 12:77-89.

NOAA. 2012. Great Lakes Water Level Observations. Interactive Great Lakes Water Level Dashboard (version 1.5). Available via web: http://www.glerl.noaa.gov/data/now/wlevels/levels.html

Olson, E. J., E. S. Engstrom, M. R. Doeringsfeld, and R. Bellig. 1995. Abundance and distribution of macroinvertebrates in relation to macrophyte communities in a prairie marsh, Swan Lake, Minnesota. *Journal of Freshwater Ecology* 10:325-335.

Osland, M. J., E. Gonzales, and C. J. Richardson. 2001. Restoring diversity after cattail expansion: disturbance, resilience, and seasonality in a tropical dry wetland. *Ecological Applications* 21:715-728.

Phillips, E. C. 2008. Invertebrate colonization of native and invasive aquatic macrophytes in Presque Isle bay, Lake Erie. *Journal of Freshwater Ecology* 23:451-457.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and Economic Costs Associated with Non-indigenous Species in the United States. Accessible via web. http://www.news.cornell.edu/releases/Jan99/species_costs.html.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics* 52:273-288.

Prince, H. H., P. I. Padding, and R. W. Knapton. 1992. Waterfowl use of the Laurentian Great Lakes. *Journal of Great Lakes Research* 18:673-699.

Relyea, R. A. 2005. The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. Ecological Applications 15:618-627.

Riffell, S. K., B. E. Keas, and T. M. Burton. 2001. Area and habitat relationship of birds in Great Lakes coastal wet meadows. *Wetlands* 21:492-507.

Rose, C and W. G. Crumpton. 1996. Effects of emergent macrophytes on dissolved oxygen dynamics in a prairie pothole wetland. *Wetlands* 16:495-502.

Schultz, R. and E. Dibble. 2012. Effects of invasive macrophytes on freshwater fish and macroinvertebrate communities: the role of invasive plant traits. *Hydrobiologia* 684:1-14.

Smith, S. 1987. *Typha*: Its taxonomy and the ecological significance of hybrids. Archive für *Hydrobiologie* 27: 129-138.

Sojda, R. S. and K. L. Solberg. 1993. *Management and Controls of Cattails*. Fish and Wildlife Leaflet. U. S. Fish and Wildlife Service.

Stead, T. K., J. M. Schmid-Araya, and A. G. Hildrew. 2003. All creatures great and small: patterns in the stream benthos across a wide range of metazoan body size. *Freshwater Biology* 48:532-547.

Stoffels, R. J., G. P. Closs & C. W. Burns, 2003. Multiple scales and the relationship between density and spatial aggregation in littoral zone communities. *Oikos* 103: 81–92.

Tronstad, L. M., B. P. Tronstad, and A. C. Benke. 2005. Invertebrate responses to decreasing water levels in a subtropical river floodplain wetland. *Wetlands* 25:583-593.

Tuchman, N.C., D.J. Larkin, P. Geddes, R. Wildova, K. Jankowski, and D.E. Goldberg. 2009. Patterns of environmental change associated with *Typha × glauca* invasion in a great lakes coastal wetland. *Wetlands* 29:94-975.

Tulbure, M. G., C. A. Johnston, D. L. Auger. 2007. Rapid invasion of a Great Lakes coastal wetland by non-native *Phragmites australis* and *Typha*. *Journal of Great Lakes Research* 33:269-279.

Vaccaro, L, B. Bedford, C. Johnston. 2009. Litter accumulation promotes dominance of invasive species of cattails (*Typha* spp.) in Lake Ontario wetlands. *Wetlands* 29:1036-1048

Vail, L. M. 2009. Soil nutrient changes following a *Typha × glauca* invasion in a Great Lakes coastal wetland. Master's Thesis. Loyola University Chicago.

Vepraskas, M. J., and S. P. Faulkner. 2001. Redox Chemistry of Hydric Soils. pp. 85 – 105 in J. L. Richardson and M. J. Vepraskas eds. *Wetland Soils* CRC Press, Boca Raton, FL.

Voigts, D. K. 1970. Aquatic invertebrate abundance in relation to changing marsh vegetation. *The American Midland Naturalist* 95:313-322.

Warren, R. S., P. E. Fell, J. L. Grimsby, E. L. Buck, G. C. Rilling, and R. A Fertik. 2001. Rates, patterns, and impacts of *Phragmites australis* expansion and effects of experimental *Phragmites* control on vegetation, macroinvertebrates, and fish within tidelands of the lower Connecticut River. *Estuaries*. 24: 90-107.

Wieten, A. C., M. J. Cooper, A. D. Parker, and D. G. Uzarski. 2012. Great Lakes coastal wetland habitat use by seven turtle species: influences of wetland type, vegetation, and abiotic conditions. *Wetlands Ecology and Management* 20:47-58.

Woo, I. and J.B. Zedler. 2002. Can nutrients alone shift a sedge meadow towards dominance by the invasive *Typha × glauca*? *Wetlands* 22:509-521.

Zedler, J. B. and S. Kercher. 2004. Causes and consequences of invasive plants in wetlands: opportunities, opportunists, and outcomes. *Critical Reviews in Plant Sciences* 23:431-452

VITA

Kimberly Greene received a Bachelor's of Science in Ecology and Evolutionary Biology from The University of Michigan in 2009. From 2007 through 2010 she worked in a Great Lakes Ecology research laboratory largely analyzing crustacean and rotifer zooplankton samples for a Great Lakes biomonitoring project. In January 2011 she entered the Loyola University Chicago Biology Graduate Program. She presented her research at the Wisconsin Wetlands Association annual meeting in February 2012.