COASTAL WETLAND FRAGMENTATION BY BOAT CHANNELS AND DRAINAGE OUTLETS

Neil T. Schock

A thesis submitted in partial fulfillment of
the requirements for the degree of
Master of Science

Department of Biology

Central Michigan University
Mount Pleasant, Michigan
May, 2012

Accepted by the Faculty of the College of Graduate Studies,

Central Michigan University, in partial fulfillment of

the requirements for the master's degree

Thesis Committee:

Donald Uzarski, Ph.D.	Committee Chair
Brent Murry, Ph.D.	Faculty Member
Thomas Gehring, Ph.D.	Faculty Member
March 2, 2012	Date of Defense
Roger Coles, Ed.D.	Dean College of Graduate Studies
April 11, 2012	Approved by the College of Graduate Studies

This is dedicated to my wife Anne Marie and my parents, Sid and Marilee. Your love, sacrifice and support made the completion of this endeavor possible.

ACKNOWLEDGEMENTS

I would like to thank my committee members for their willingness to contribute thoughts and expertise, which has greatly enhanced the quality of this research. I would especially like to thank Dr. Don Uzarski for giving me the opportunity to work with him on research that holds great value to me and to society. Dr. Uzarski has developed me into a productive wetland scientist, providing his knowledge in many research related areas. Dr. Uzarski has provided me with great opportunities to advance my abilities and has also been a model for what I strive for in numerous areas outside of science. I feel blessed to be working with someone who is so passionate about his craft and has given me a chance to prove myself when others doubted my abilities. Thank you to Dave Coulter and Cody Webster for your experience and ideas pertaining to data collection techniques and experimental design. I would also like to thank Tom Clement and Jessica Sherman for your time and patience while collecting data and editing manuscripts. Thank you to the following Uzarski lab members who have contributed to the completion of this research: Ryan Wheeler, Carli Gurholt, Chad Blass, Mike O'Leary, Jack Distel and Lee Schoen. Funding sources such as Central Michigan University, Michigan Department of Environmental Quality and the US Environmental Protection Agency provided travel and supplies. Special thanks to William J. Strickler who mentored and provided me with the inspiration to pursue my dreams. Thank you to my parents and my brothers for shaping me into the person I am today. Finally, I would like to thank my wife whose endless love and support has played a vital role in my development as a professional and as a person. I could not have been any more fortunate in picking a life partner.

ABSTRACT

IMPACTS OF BOAT CHANNELS AND DRAINAGE OUTLETS ON GREAT LAKES COASTAL WETLANDS

by Neil T. Schock

Great Lakes coastal wetlands experience long term water level changes. Prolonged low water periods often prompt riparian property owners to manipulate wetland structure and fragment coastal wetland habitat. Boat channel dredging and vegetation removal practices are easily exercised under low water conditions. I hypothesized these habitat alterations would have measurable impacts on habitat conditions, fish, and invertebrate communities. To test this hypothesis I sampled wetlands immediately adjacent to boat channels and compared these disturbed sites with reference sites located 100 to 800m from each disturbance. Fish and invertebrates were sampled from both site types using fyke nets and dip nets respectively. Physical characteristics (water depth and vegetation type) were kept consistent between reference and disturbed sites in order to isolate the disturbance as the responsible component for differences observed in habitat conditions or biotic communities. Among the sites selected, two genera of dominant emergent vegetation (*Typha* and *Schoenoplectus*) were sampled and grouped for analysis. I also analyzed the effects of drainage outlets on the physical/ chemical conditions and biological communities of coastal wetlands. Drainage systems act as corridors for agricultural pollutants to infiltrate coastal wetlands and alter the habitat and biological compositions of areas near outlets. Sites that were less than 800 meters from the mouth of a drainage outlet were considered “near” outlets, whereas sites that were greater than 1500 meters were considered “far” from outlets. Sites were then grouped based on being near or far from drainage outlets to detect differences in habitat characteristics and biotic community compositions. Statistical analyses revealed differences in individual habitat characteristics between reference and disturbed sites. Habitat conditions and invertebrate communities were noticeably different when I compared *Typha* and *Schoenoplectus* sites. Drainage outlets also had a noticeable impact on the observed habitat conditions and biotic communities. Invertebrate communities near outlets were significantly different and fish community differences hovered just above the statistical significance level. These relationships suggest that watershed land managers should make an effort to work with the agricultural community in an attempt to dampen the negative effects that intensive agriculture has on Great Lakes coastal wetlands.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

IMPACTS OF DRAINAGE OUTLETS AND BOAT CHANNELS ON THE HABITAT CONDITIONS AND MACROINVERTEBRATE COMMUNITIES OF GREAT LAKES COASTAL WETLANDS

Introduction

Great Lakes coastal wetlands are valuable habitats that play an important role in the overall health and value of the Great Lakes ecosystem as a whole. In particular they are important habitats for fish, amphibians, reptiles, invertebrates, birds and mammals (Goodyear 1982, Jude and Pappas 1992, Prince 1992, Whitt 1996, Michael 2003, Taft et al. 2005). Wetlands also support habitat conditions that are important for natural processes to take place such as denitrification (Groffman 1994), nutrient conversion and containment (Howard – Williams 1985, Yan et al. 1997, Melzner 1999). These systems experience natural water level fluctuations of varying duration that are necessary to maintain plant and faunal diversity (Burton 1985). Daily water level changes can measure up to 20cm, while seasonal water level fluctuations have been recorded up to 40cm (Bedford 1992). Larger scale water level changes have also been observed throughout these systems. Natural fluctuations, ranging from years to decades, occur on a regular basis in the Great Lakes, but recently, declining water levels have been observed.

In 2003 record lows were recorded and as a result, previously submerged substrates were left exposed. Shoreline landowners (private and public) used this as an opportunity to groom shorelines and dredge boat channels for watercraft navigation and lake access. The removal of aquatic macrophytes coupled with the disruption of hydric soil, eliminates coastal wetland habitat and fragments the remaining wetland complex. It is obvious that these anthropogenic actions have a direct impact on wetland structure, but they also have an indirect effect on the condition of the water in these habitats. Areas where coastal wetland vegetation has been removed, act as corridors for pelagic water to infiltrate the remaining coastal wetlands, which alters the abiotic characteristics of these habitats. A change in invertebrate community composition and a reduced abundance has also been observed in areas were coastal wetland vegetation has been removed (Uzarski et al. 2009). Vegetation community composition changes have also been associated with anthropogenic disturbances (Lougheed et al. 2001) and in some cases, displace rare plant species and their seed banks (Wisheu and Keddy 1991).

The expansion of non-native *Phragmites spp.* and *Typha spp.* has also been correlated with periods of below average water levels (Tulbure et al. 2007). Amsberry et al. (1999) found that when transplanted to deeper water *Phragmites australis* failed to establish, but in drier soils the plants thrived and expanded. This suggests that as water levels recede, exposed wetland substrates are easily colonized by these invasive wetland plant species. The addition of large, highly competitive plants can change wetland habitat structure in as little as two years. This change would presumably alter value and function (Tulbure et al. 2007). Additionally, these very tall, herbaceous wetland plants impede lake viewing and access sparking intensive mowing and raking to remove these large invasive plants leaving the remaining wetland habitat either un-vegetated or heavily fragmented. Coastal areas that no longer maintain wetland plants have significantly lower invertebrate abundance, altered chemical and physical water conditions as well as become susceptible to the establishment of invasive plant species (Amsberry 1999, Uzarski et al. 2009).

Previous research has related vegetation conditions (type, density, and diversity) to characteristics in macroinvertebrate assemblages. In most cases, changes in the type of dominant plant species gave rise to altered invertebrate community compositions (Olsen et al. 1995, Webb et al. 1984, Angradi et al. 2001). This suggests that if vegetation composition is transformed by a disturbance, the macroinvertebrate community in that particular habitat will be altered as well. These changes may have impacts on fish and other organisms that depend on macroinvertebrates as prey items as macroinvertebrates are of particular importance to fish during their early life stages (Wei et al. 2004). In addition to the alteration and elimination of wetland habitat, disturbances in these areas fragment wetland habitat, which also negatively influences wetland value.

Habitat fragmentation increases the edge to center ratio of impacted coastal wetlands. Wetland edges give rise to a decrease in plant densities as well as supporting plants with lower reproductive potential when compared to individuals from the center of a large group (Lienert and Fischer 2003). Similarly, habitat fragmentation has detrimental effects on the abundance and diversity of pollinating insects and as a result, plant species in the remaining habitat have a reduced pollination rate, which gives rise to a decrease in seed production (Rathcke and Jules 1993). Habitat fragmentation can also produce a loss of genetic diversity within some wetland plant communities. This unnatural isolation of habitat can have a significantly negative impact on habitat quality (Hooftman et al. 2003). Common and rare plant species both experience the confounding effects of lost fecundity and decreased genetic diversity due to habitat fragmentation which could permanently inhibit the success of plant communities (Honnay and Jacquemyn 2007).

Heavily fragmented or isolated wetland habitats may tend to portray a functional ecology similar to that of an island. The theory of island biogeography provides insight on specific variables related to fragmentation such as "the degree of isolation"; when a habitat is isolated, organisms from other habitat patches are not likely to immigrate to the isolated habitat. Fragmented patches that are isolated to a lesser degree are likely to have more individuals immigrating to that habitat, which reduces the loss of genetic diversity among populations (MacArthur and Wilson 1963). This degree of isolation and the rate of immigration are primarily dependent on the mobility of a species and its willingness to move to a new habitat. Species richness within a fragmented habitat is dependent on the size of the connected available habitat. This suggests that narrow boat channels should not be as detrimental to macroinvertebrate species richness as long as the remaining habitat patch is large enough to sustain genetic diversity and species richness.

Many of the boat channel disturbances in this study were constructed on the banks of outlet systems that drain the surrounding landscape. Most of these drainages were channelized streams or ditches that cut through agricultural areas associated with elevated nutrient levels and sedimentation. The River Continuum Concept suggests the form of available organic matter in a riverine habitat will have a significant impact on the type of invertebrates that inhabit an area (Vannote et al. 1980). Cummins (1973) grouped invertebrates into categories known as "functional feeding groups" based on the mode of which they acquire their food. When a habitat contains an abundance of course particulate organic matter, invertebrate communities will be made up of primarily shredders. Conversely, when the available organic matter is in the form of fine particulates, the invertebrate community in that area will be dominated by collectors and filterers. Invertebrate communities in disturbed stream systems are often made up of taxa that can more readily obtain fine particulates and dissolved nutrients (Harding 1999). Normally associated with stream ecosystems, the same patterns likely apply in coastal wetlands, especially those adjacent to drainage outlet systems.

Objectives and Hypotheses

The main objective of this study was to determine how the presence of a boat channel affects the chemical and physical components, as well as the macroinvertebrate communities found in coastal wetlands located immediately adjacent to boat channels. Very little research has been done on small scale disturbance and its effect on the macroinvertebrate communities of lentic systems. I hypothesized the macroinvertebrate communities inhabiting the coastal wetland vegetation near boat channels, as well as the habitat conditions near channels would be significantly altered as a result of habitat disturbance.

The second objective of this study was to provide insight on the relationship between the dominant plant species found at a site with the abiotic habitat characteristics and macroinvertebrate communities of these vegetated areas. Past research has suggested that invertebrate community compositions have been significantly correlated with vegetation type (Burton et al. 2004). In southern Brazil, hydrophytes, reed plants and leafy emergent habitats all had distinctly different invertebrate community compositions (de Avila et al. 2011). Four wetland habitat types in southern Minnesota were found to support significantly different macroinvertebrate abundances and diversities inhabiting inland lake wetlands (Olson et al. 1995). I hypothesized the macroinvertebrate communities and habitat conditions of a Great Lakes coastal wetland are a function of the type of dominant emergent vegetation in a given habitat.

The third objective of this study was to detect effects of drainage systems on the abiotic characteristics and the macroinvertebrate communities of nearby coastal wetlands. Previous research has concluded macroinvertebrate community compositions and functional feeding group relative abundance is directly correlated with the form of available organic carbon (Vannote et al. 1980). In addition, the presence of agricultural operations has been shown to alter steam habitat characteristics as well as the invertebrate community compositions (Klotz 1985, Harding et al. 1999, Gleason et al. 2003). Relyea (2005) monitored 25 aquatic organisms that were exposed to four commonly used agricultural chemicals. Two common pesticides and a common herbicide were responsible for the elimination of a diverse group of aquatic organisms. Some organisms were more negatively impacted than others. Reductions in species richness reduction ranged from 15% to 30% in communities depending on which harmful chemicals the organisms were exposed to. I hypothesized the invertebrate communities surveyed "near" (< 800 meters) drainage outlets would be different when compared to the invertebrates collected in locations "far" (> 1500 meters) from outlets. I also expected to see a shift in the relative abundance of functional feeding groups of the organisms near drainage outlets with an increase of collector filterers and a decrease in grazers.

Methods

Site Selection

Sites in this study were selected from a variety of coastal wetlands along Michigan's shoreline including those of northern Lake Michigan, northern Lake Huron and the expansive marshes of Saginaw Bay (Figure 1). Macroinvertebrate samples were collected at 23 reference and disturbed site pairs from Great Lakes coastal wetlands during the periods of 17 June to 18 August 2009, 2010, and 2011. This time period was chosen because macrophytes are mature at this time and macroinvertebrates are at an identifiable stage. Consistency in the timing of invertebrate collection was also important in order to collect organism exhibiting similar life stages. Disturbed sites were located in the wetland vegetation that was located immediately adjacent to an active boat channel. Reference sites were then selected from 100 to 800 meters in either direction up the shoreline from the disturbance (Figure 2). Habitat characteristics such as vegetation type and water depth were kept consistent in order to isolate the boat channel as the responsible factor for differences observed between reference and disturbed sites.

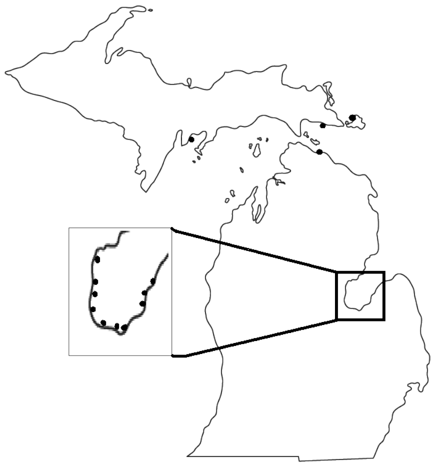

Figure 1. Black dots represent coastal wetland habitats that were sampled.

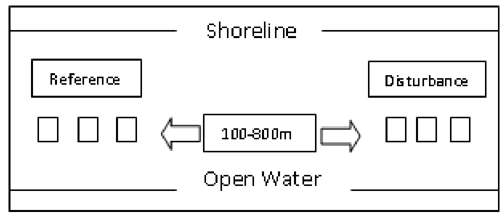

Figure 2. Small boxes represent replicate locations within each wetland site pairing.

Habitat Characteristics

Upon arriving at a site, physical and chemical measurements were measured from the column, 2 meters inside the wetland vegetation using a sonde multi-probe (Yellow Springs Instruments model 6600 V2). I measured *in situ* temperature, pH, turbidity (NTU's), chlorophyll *a* (µg/L), oxidation–reduction potential (mV), specific conductance (µs/cm), and dissolved oxygen (mg/L). Two 500 mL acid washed water bottles were used to sample water from the wetland for chemical analysis. Water from one bottle was used to measure alkalinity by titration. This process was conducted by titrating 0.02 M H2SO4 into 100 milliliters of unfiltered water collected from the wetland. Water from the other bottle was filtered using a 0.45µm Milliipore filter to obtain water suitable for dissolved nutrients analysis which included soluble reactive phosphorus (SRP), nitrates (NO3-N), and ammonium (NH4-N). Dissolved nutrients were analyzed using a BRAN+LUEBBE QuAAtro analyzer. Each nutrient parameter had an associated detection limit on the auto-analyzer, the lowest level that the machine could accurately measure. For all samples that had a value below the detection limit, the detection limit of that specific parameter was divided by half and this value was used for data analysis. The detection limit on the Bran+Luebbe QuAAtro for SRP, nitrate, and ammonium was 0.3μg/L, 1.0 μg/L, and 0.5 μg/L, respectively.

A 10 cm deep soil core was also obtained at each site using a 47mm diameter core sampler. These samples were then bagged and stored on ice until I returned to the lab in which the soils were stored in the freezer. Upon processing, soils were thawed and dried at 60°C for 24 hours and weighed. Soils were then ashed at 550°C for 24 hours and reweighed in order to compute the percentage of organics present in the soil. Water depth and the depth to a resistant layer in the soil were collected using a standard aluminum meter stick. Vegetation stem density was measured using a 0.25 m x 0.25 m quadrat. The amount of wave exposure (modified effective fetch) was calculated using Google Earth™ and the British Columbia Estuary Mapping System© (Resource Inventory Committee 1999; e.g. Burton et al. 2004).

Eq. 1.

)

]

Fm = Modified effective fetch
F = Fetch distance
45L = 45° left of perpendicular to the shoreline
45R = 45° right of perpendicular to the shoreline

Macroinvertebrate Collection

Macroinvertebrates were collected from all present microhabitats using D-frame dip nets with a mesh size of 500 µm. Organisms were emptied into white collection trays with 5 x 5 cm grids lining the bottom. When an organism was selected from one of the grids on the white tray, all organisms from that grid were then collected in order to exclude size bias and obtain a diverse representation of organisms for each replicate sample. Specimens were picked using forceps for 30 person minutes per replicate or until 150 organisms were obtained. If the 30 person minutes expired before 150 organisms had been collected, than the next reciprocal of 50 was achieved. Therefore, each replicate contained approximately 50, 100 or 150 organisms. Three replicates were collected at least 10 meters apart within each sample site. All picked organisms were immediately preserved in 95% ethanol and taken back to the lab for identification to the lowest operational taxonomic unit using a stereomicroscope. Macroinvertebrates were identified to various taxonomic levels based on the level of difficulty to accurately identify specific organisms, but most specimens were resolved to the genus level. Individual taxa from each replicate where then stored in separate vials with polyseal caps and preserved in ethanol. The dichotomous keys used for identification were Peckarsky *et al.* (1990) as well as Merritt and Cummins (1996).

Data Analysis

Site specific macroinvertebrate data such as richness, abundance, and Shannon diversity index scores were computed. Shannon diversity represents a combination of species richness and evenness to establish numerical values that represent the proportional distribution of taxa at a given site.

Shannon Diversity Index Eq. 2:

H' = Shannon diversity index
P_i = Proportion of the entire population made up of taxa
S = number of species encountered
$\sum$ = sum from all species

Paired t-tests were run on macroinvertebrate, chemical, and physical data to establish if there were differences in the variables associated with disturbed and reference sites (Minitab version 16, Minitab Inc., U.S.A.). An alpha value of 0.05 was used to determine statistical significance in all tests. Principal component analysis (PCA) was used to visualize the overall physical and chemical habitat conditions of sites. Sites were then graphed on a bi-plot based on the degree of similarity among sampling locations. Locations that had more similar abiotic characteristics were grouped more closely on the bi-plot. This allowed for a visual representation of the sites based on habitat characteristics.

Macroinvertebrate community data was analyzed using non-metric multidimensional scaling (NMDS). This analysis allowed me to plot sites based on how similar their communities were to one another and to use symbols to represent site groups. A multi response permutation procedure (MRPP) was then used to determine if the designated site groups were statistically similar. Statistical analyses were performed using PC-ORD version 5 (MjM Software, Glenden Beach, Oregon, U.S.A.). Paired t-tests were conducted on macroinvertebrate, chemical and physical data to establish if there were any significant differences in the individual variables associated with disturbed and reference sites (Minitab version 16, Minitab Inc., U.S.A.). Percentages of invertebrates separated by functional feeding groups were arcsine square root transformed in order to achieve normality. Values were separated based on their location in relation to drainage outlets, and two sample t-tests were conducted on each functional feeding group.

Results

Boat Channels

I collected 14,954 invertebrates that represented 132 taxa. Invertebrate richness ranged from 9 to 37 taxa per site. Invertebrate richness averaged 22.08 and 21.39 taxa from reference and disturbed sites respectively. Abundance of invertebrates averaged 284.2 at reference sites and 239.9 at disturbed sites. Shannon diversity index of invertebrates averaged 0.972 at reference sites and 0.946 at disturbed sites. From the 20 reference sites that were sampled, 100 invertebrate taxa were collected and identified. Sampling of 23 disturbed wetland sites produced 106 invertebrate taxa. None of the macroinvertebrate data were different when reference and disturbed sites were compared, however some abiotic factors did differ (Table 1). Turbidity levels at disturbed sites averaged 9.22 NTU and reference sites averaged 3.61 NTU. Alkalinity at disturbed sites produced an average of 136.52 (mg/L) and reference sites had an average of 148.13 (mg/L). The organic composition was higher at disturbed sites with an average of 3.07%

whereas reference site soils averaged 2.36%. The average water temperature was 25.22°C at reference sites and was 24.83°C at disturbed sites.

Table 1. Paired t test results of the abiotic characteristics recorded from 23 reference and disturbed (R & D) site pairings. Each disturbed site was located in the wetland vegetation located immediately adjacent to a boat channel and its paired reference site was located 100 – 800 meters from the disturbance.

Parameter	**R&D P-Value**
Dissolved Oxygen	0.834
Turbidity	0.041
Specific Conductance	0.252
Alkalinity	0.003
Redox Potential	0.135
pH	0.195
Chlorophyll a	0.188
Water Temperature	0.081
Stem Density	0.773
Organic Depth	0.164
NH4	0.155
NO3	0.334
SRP	0.248
% Organics	0.031
Invertebrate Richness	0.581
Invertebrate Shannon	0.558
Invertebrate Abundance	0.715

Principal component analysis comparing abiotic parameters of reference and disturbed sites showed no pronounced stratification between the two site types (Figure 3). Axis 1 accounted for 18.5% of the variation observed among the sampling sites. This variation was predominantly driven by specific conductance and the percentage of organics in soil samples. Sites that fell out on the right side of the bi-plot where associated with higher specific conductance levels, while the sites that appeared on the left were associated with high soil organic content. Axis 2 was responsible for 16.9% of the variation observed in the abiotic data. Environmental characteristics associated with separation on axis 2 were redox potential, organic depth and turbidity. Sites that appeared on the upper portion of the PCA bi-plot had elevated redox potential readings, whereas sites that appeared on the lower end of the bi-plot maintained higher turbidity levels and deeper organic depths.

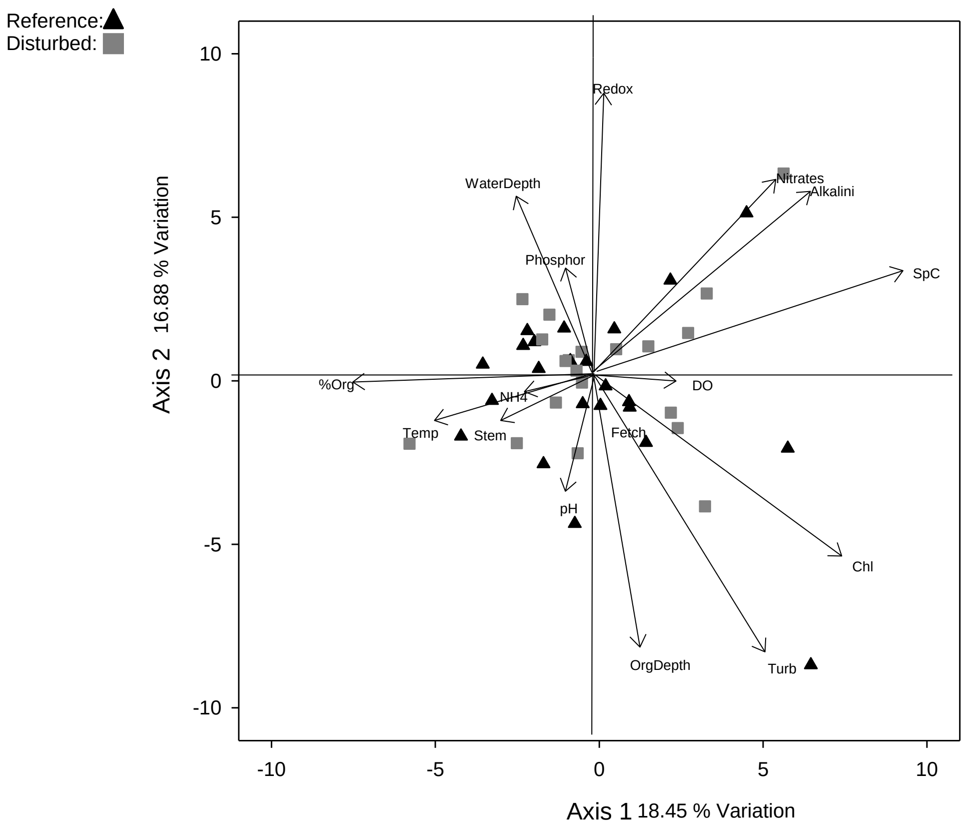

Figure 3. Principal components analysis bi-plot showing the distribution of 23 pairs of sites based on 16 abiotic characteristics. Vectors were inflated to 200% in order to better exemplify the data. Black triangles represent disturbed sites near boat channels and gray square site labels represent reference sites that were located 100-800 meters from the disturbance.

While reference and disturbed sites differed with individual abiotic factors, there was no structuring of invertebrate communities to match this. The NMDS analysis portrayed no detectable difference in the invertebrate communities of reference and disturbed sites (MRPP t = 1.42, p = 0.988) (Figure 4).

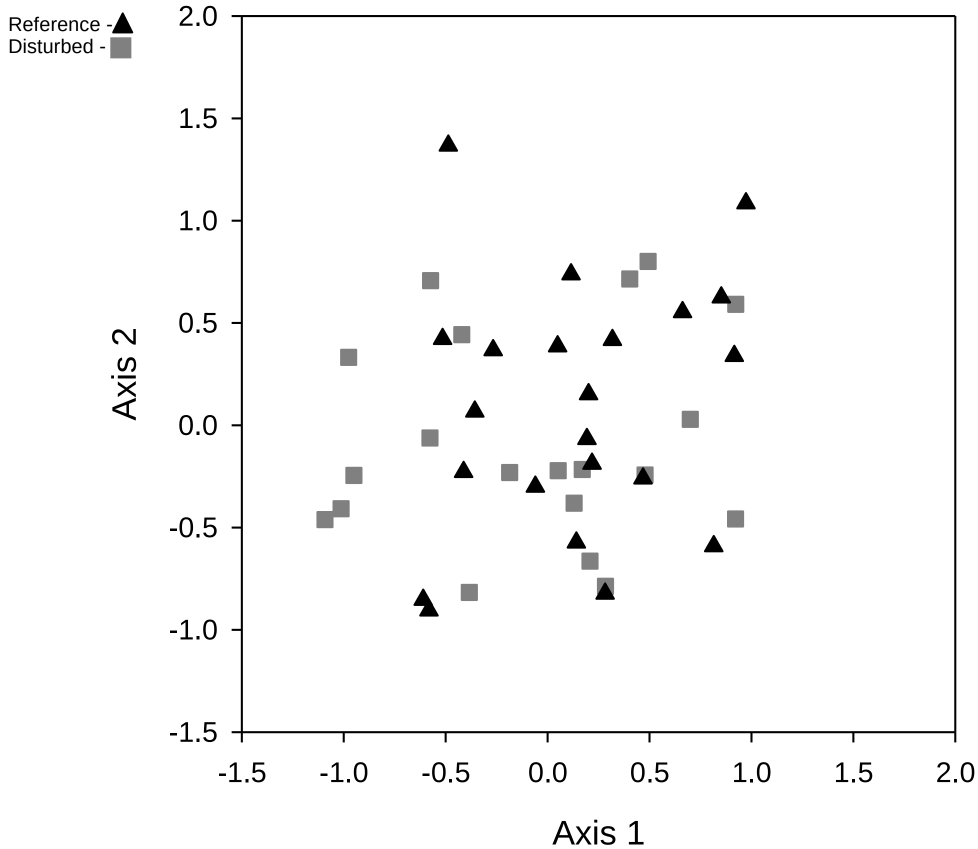

Figure 4. Non-metric multidimensional scaling bi-plot displaying the invertebrate community compositions of disturbed and reference sites. Disturbed sites were located immediately adjacent to a boat channel and its reference site was located 100 - 800 meters away from the disturbance. No significant differences in community compositions were observed between the two site types.

Vegetation Type

Principal components analysis shows that different dominant emergent vegetation types maintained noticeably unique abotic conditions (Figure 5). *Typha* sites were associated with elevated levels of specific conductance and chlorophyll *a* levels.

Schoenoplectus sites contained a larger percentage of organics and higher water temperatures.

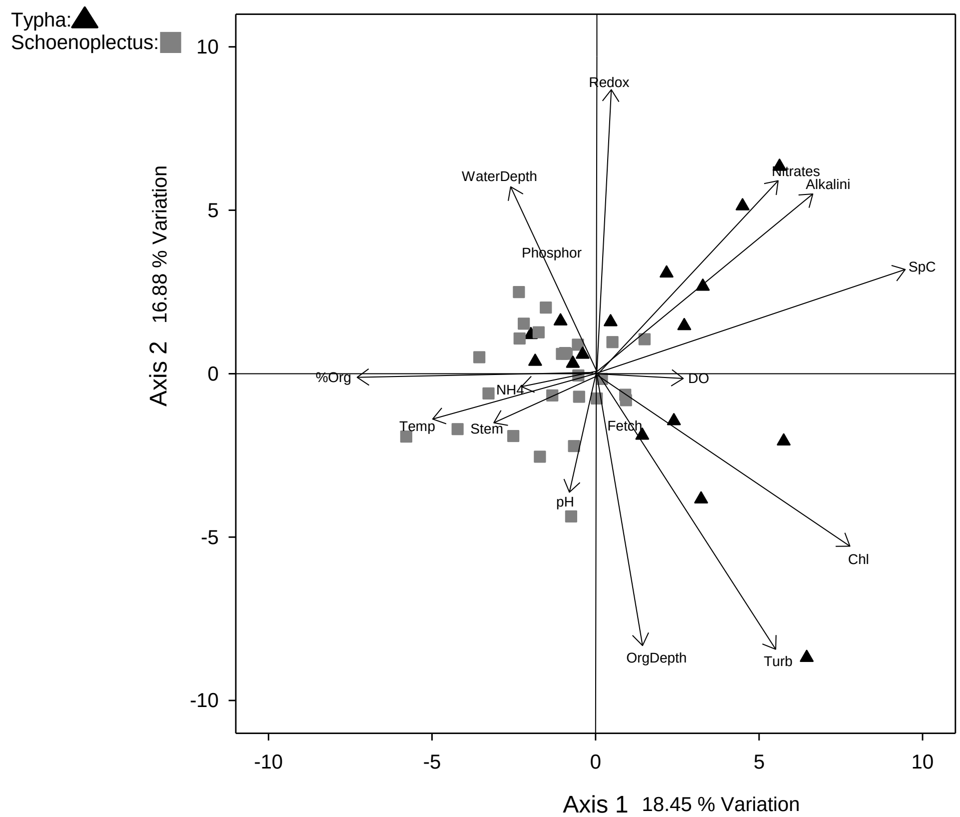

Figure 5. Principal component analysis bi-plot showing the difference in the abiotic characteristics of *Schoenoplectus* and *Typha* sites based on 16 measured parameters. Elevated levels of chlorophyll *a*, specific conductance, alkalinity, and nitrates were observed at *Typha s*ites. *Schoenoplectus* sites had high organic soil concentration, stem density, and temperature.

Non-metric multidimensional scaling revealed that the invertebrate community composition was significantly different between dominant vegetation types (MRPP t = -3.55, p = 0.005). *Typha spp.* and *Schoenoplectus spp*. were the two plant taxa used in this comparison (Figure 6). Invertebrate taxa associated positively with axis 1 were the Odonata genera *Epitheca*, *Libellula*, *Hagenius*, and the gastropod species *Stagnicola caperata*. Taxa associated with the negative side of axis 1 were an unknown Lampyridae and the Ephemeroptera genus *Tortopus*. Axis 2 was positively correlated with the Tricoptera genus *Leptocerus* and was negatively correlated with the Coleoptera genus *Dubiraphia* and the Hemiptera genus *Neocorixa.*

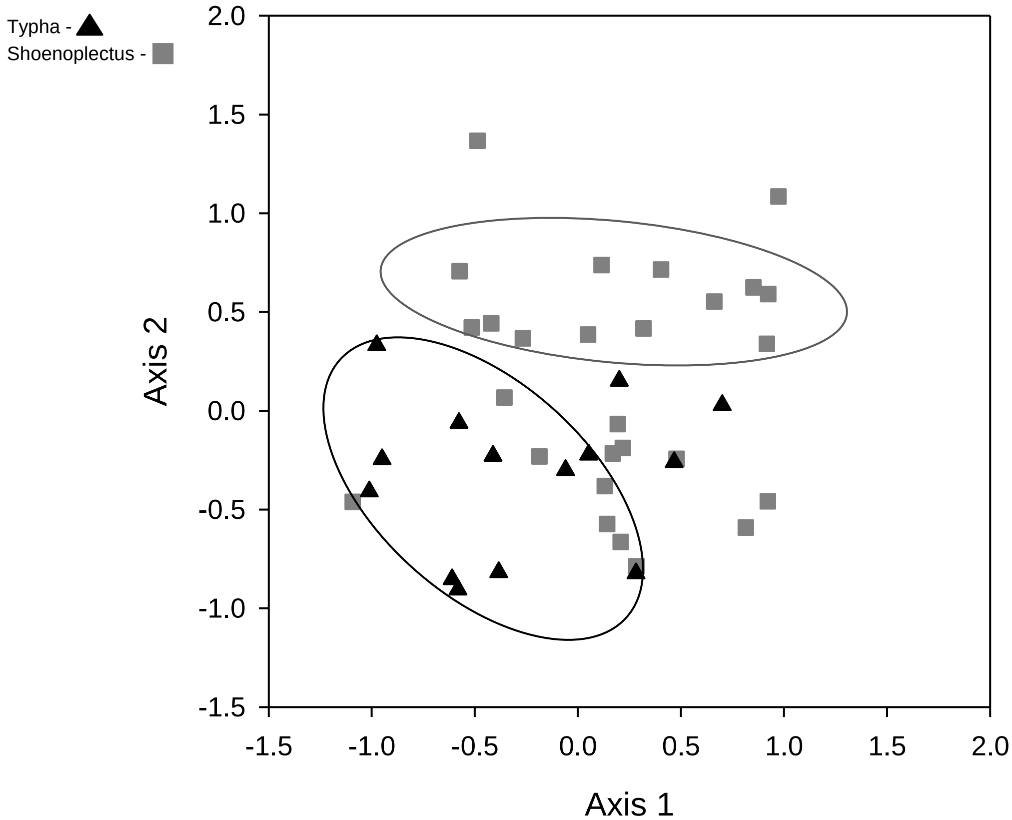

Figure 6. Non-metric multidimensional scaling showing a significant difference in the invertebrate community compositions collected from *Typha* and *Schoenoplectus* dominated habitat types (MRPP t = -3.55, p = 0.005). Invertebrate community compositions were distinctly separated based on dominant vegetation type.

Drainage Outlets

Principal component analysis of sites located "near" (< 800m) drainage outlets had unique abiotic characteristics when compared to sites that were located in areas "far" (>1500 m) from outlets (Figure 7). Sites located near drainage outlets were associated with higher levels of available nitrates and alkalinity, whereas sites far from outlets were very central to the origin with the exception of two outliers. Sites near drainage outlets were more variable with no clear pattern of distribution.

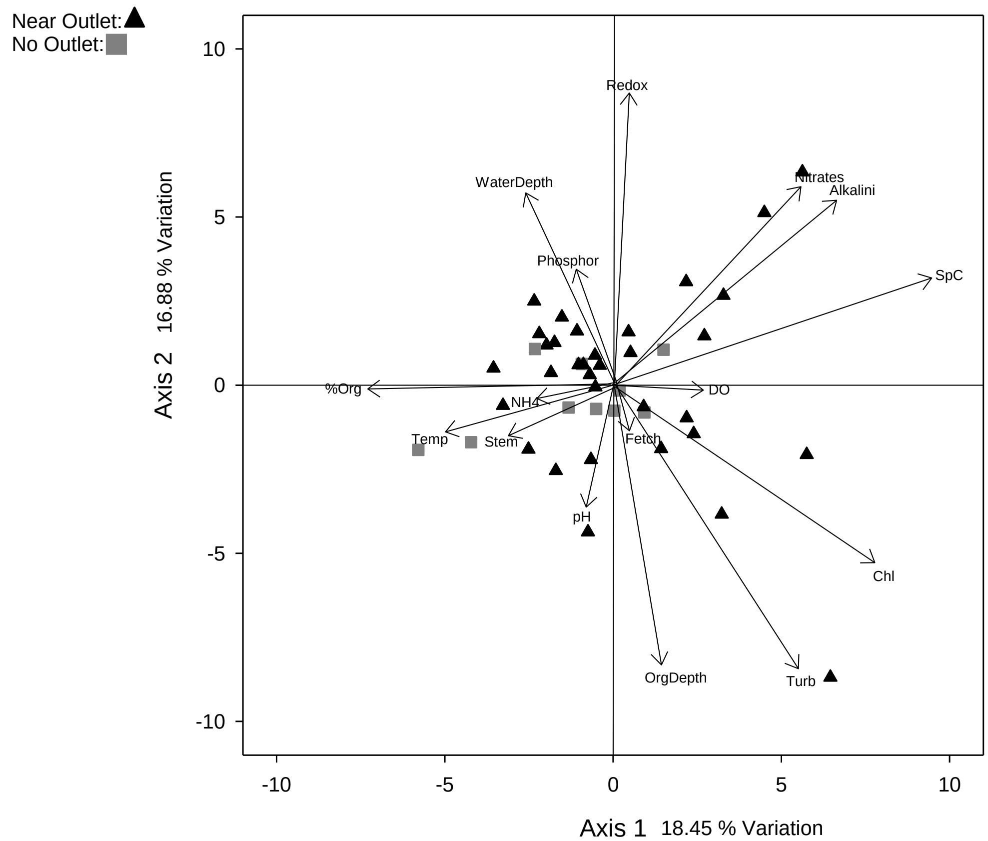

Figure 7. Principal component analysis bi-plot showing the distribution of sites based on abiotic characteristics based on 16 parameters. Sites located far from drainage outlets were more similar to each other when compared to the likeness of sites near outlets. This suggests that the presence of a drainage outlet has an impact on the high variability of chemical and physical readings that were observed in the coastal wetlands near outlets.

Macroinvertebrate community compositions from sites located near drainage outlets were also significantly stratified when compared to site pairs located far from drainage outlets (t = -3.87, p = 0.003) (Figure 8). Taxa associated positively with axis 1 were Odonata genera *Epitheca*, *Libellula*, *Hagenius*, and the gastropod species *Stagnicola caperata*. Taxa associated with the negative side of axis 1 were an unknown

Lampyridae and the Ephemeroptera genus *Tortopus*. Axis 2 was positively correlated with *Leptocerus* and was negatively correlated with the Coleoptera taxa *Dubiraphia* and Hemipteran genus *Neocorixa.*

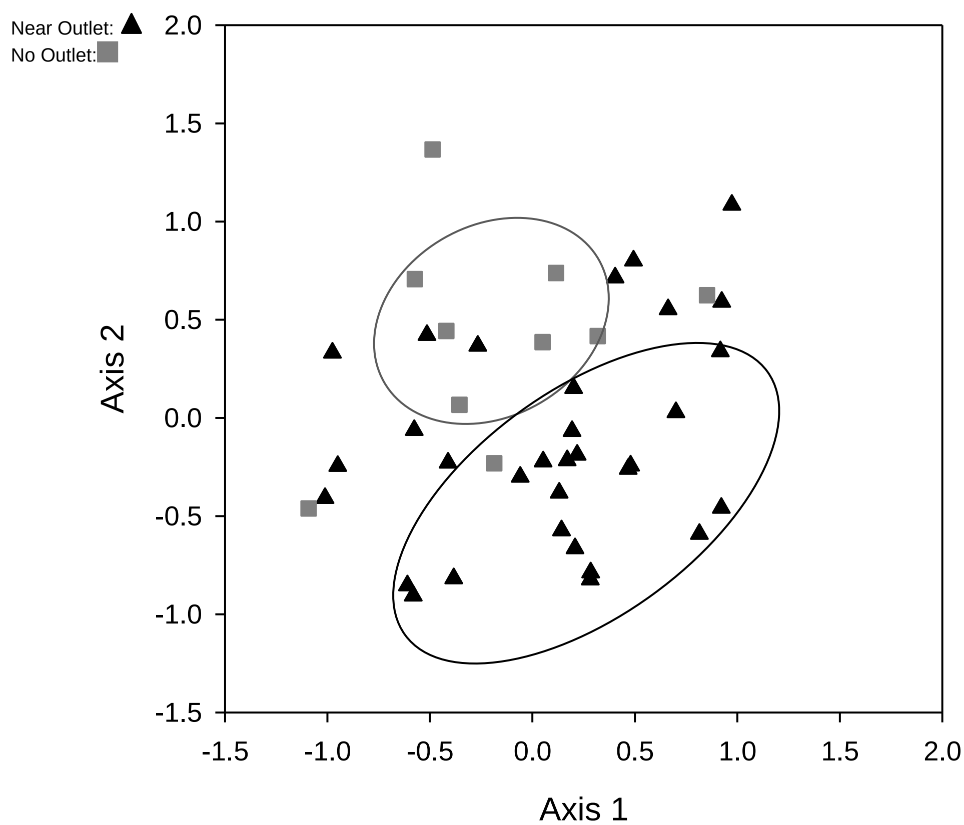

Figure 8. NMDS bi-plot showing a significant difference between the macroinvertebrate communities located near a drainage outlet and macroinvertebrate communities far from a drainage outlet (MRPP t = -3.87, p = 0.003).

Sites far from outlets had a higher relative abundance of grazers and a lower relative abundance of collector filterers when compared to the invertebrate communities located near outlets. The relative abundance of grazers near outlets made up 50% less of the overall invertebrate community when compared to the communities collected far from outlets. The relative abundance of collector filterers near outlets was greatly increased when compared to sites far from outlets (Figures 9 & 10). Two sample t-tests supported these differences in functional feeding group percentages with statistically significant p-values (< 0.05) when comparing the relative abundances for collector filterers and grazers from sites near outlets and sites far from outlets (Table 2).

Table 2. This table displays the two sample t-test results from invertebrate communities collected from sites near outlets and far from outlets. Collector filterers and grazers were significantly different when comparing the invertebrate communities from the two site types.

	Predators	Shredders	Filterers	Gatherers	Grazers
P-value	0.603	0.486	0.024	0.058	0.002

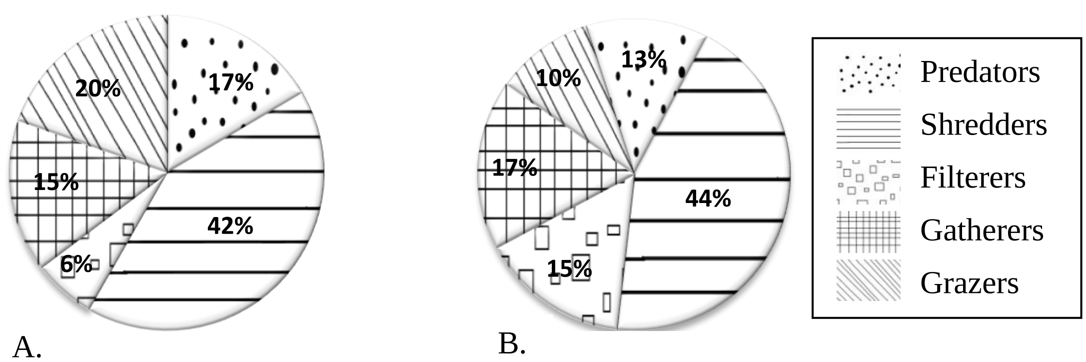

Figure 9. Pie chart (A.) displays the relative abundance of functional feeding groups of the invertebrate communities collected from wetlands far (>1500m) from drainage outlets. Pie chart (B.) displays the proportion of functional feeding groups within the invertebrate community assemblage at sites near (< 800 m) Great Lakes watershed drainage outlets.

Schoenoplectus Sites

Analysis of habitat conditions of sites near drainage outlets and sites far from drainage outlets revealed differences in abiotic characteristics (Figure 10). The chemical and physical habitat conditions near drainage outlets were highly unpredictable compared to the consistent abiotic conditions recorded at sites far from drainages. With the exception of one site pair with abnormally dense *Schoenoplectus* stem density; all sites far from drainages fell out in relatively close proximity to the origin of the PCA bi-plot, with some sites exhibiting heightened chlorophyll *a* levels. Sites located near drainage outlets contained chemical and physical characteristics that were variable, which caused these sites to fall out in all directions from the center point of the bi-plot.

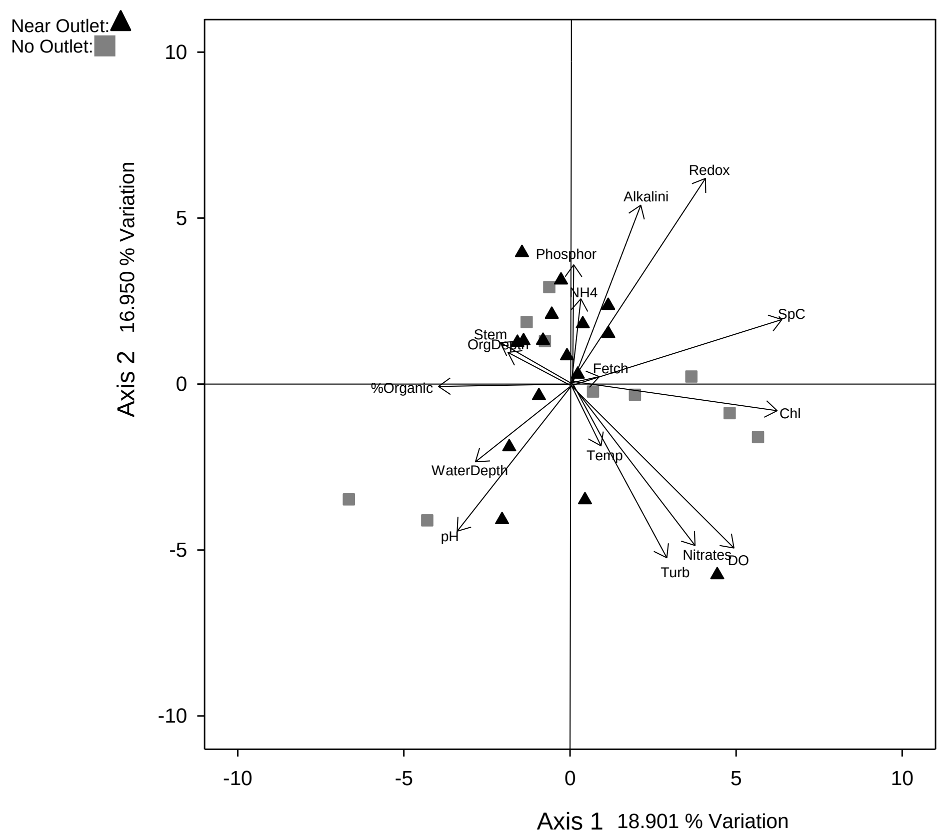

Figure 10. Principal component analysis showing the distribution of *Schoenoplectus* sites based on the abiotic data collected at each location. Sites far from drainage outlets were associated with higher chlorophyll a and specific conductance readings. Elevated levels of nitrates and pH as well as a higher concentration of dissolved oxygen were observed at sites located near outlets.

The distribution of *Schoenoplectus* sites based on the invertebrate communities were statistically segregated based on their location in relation to a drainage outlet

(Figure 11). The invertebrate species associated with the negative side of axis 1 was the Gastropod species *Acella haldemani*. The species linked to sites on the positive side of axis 1 was the Gastropod *Valvata sincera*. Odonata genera *Epitheca, Libellula and Hagenius* as well as the Gastropod *Stagnicola caperata* were linked to the negative side of axis 2. Taxa related to sites on the positive side of axis 2 were the Ephemeroptera genus *Tortopus* and the Gastropod species *Fossaria obrussa.*

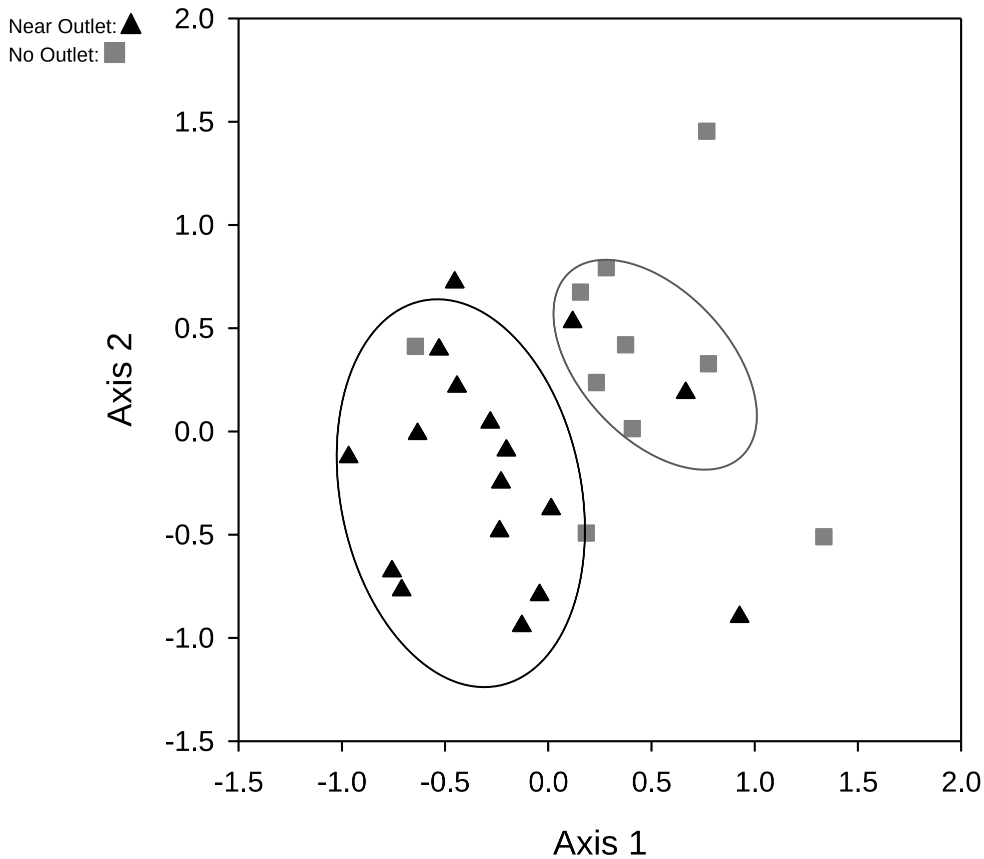

Figure 11. Non-metric multidimensional scaling bi-plot showing the distribution of invertebrate communities among *Schoenoplectus* sites near drainage outlets and *Schoenoplectus* sites far from drainage outlets (MRPP t = -3.64, p = 0.007).

Discussion

Boat Channels

Boat channels had an impact on only a few of the measured chemical and physical conditions of the coastal wetland habitat near the channel. These disturbed areas had increased water turbidity and percentage of organics in the soil, as well as decreased water alkalinity. Due to the fact that 14 of my 23 disturbed sites were located near a drainage outlet, it is not a surprise to see significantly higher levels of turbidity or soil organics in these disturbed areas. These outlet systems in most cases drained land used for agricultural purposes, which provided an increased sedimentation rate as well as high nutrient inputs. The drainage impact, in conjunction with boat traffic, makes it easy to understand why these particular characteristics were higher at disturbed sites. A decrease in alkalinity at disturbed sites was probably due to an increase in pelagic water mixing. Although these particular abiotic characteristics were different between reference and disturbed sites, there were no overall site abiotic differences detected. Additionally, there were no significant differences in the macroinvertebrate community compositions when comparing reference and disturbed sites. Uzarski et al. (2009) concluded that disturbed areas where coastal wetland vegetation had been completely removed, maintained lower invertebrate abundance as well as altered community compositions. Even though the experimental design of this study and the Uzarski study were comparable, the “disturbed” area analyzed in this study still maintained dense vegetation, which would explain the lack of differences observed between reference and disturbed site types. Literature suggests that habitat fragmentation has negative effects on wetland plant and amphibian communities (Rathcke and Jules 1993, Lehtinen 1999, Lienert 2003) and although significant alterations in macroinvertebrate communities were not observed, this does not mean boat channels are exempt from negatively affecting these organisms in some way.

Vegetation Types

When sites were grouped by dominant vegetation type, the PCA displayed an obvious difference between *Schoenoplectus* and *Typha* sites. *Typha* sites were associated with elevated levels of nitrates, alkalinity, specific conductance, chlorophyll *a*, and turbidity. *Schoenoplectus* sites generally had increased stem densities, water temperatures, and percentage of organics in the soil. Previous research suggests that vegetation type is a strong predictor of macroinvertebrate community compositions, and that certain alterations in vegetation structure has given rise to changes in macroinvertebrate communities as a result (Olsen et al. 1995, Webb et al. 1984, and Angradi et al. 2001). This was supported with the relationship between the sampled vegetation type and the macroinvertebrate community compositions in this study. The NMDS bi-plot of macroinvertebrate community compositions showed a significant difference in communities collected from *Typha* habitats when compared to communities sampled from *Schoenoplectus* habitats.

Drainage Outlets

Similarly, the presence of a drainage outlet had a significant impact on the chemical, physical and biological components of the sampled Great Lakes coastal wetlands. Sites located near the mouth of drainage outlets had significantly different habitat conditions and macroinvertebrate community compositions. The habitat parameters measured near drainage outlets were highly variable, whereas sites located far from outlets displayed more moderate variation among abiotic readings. Drainage systems contribute large amounts of fine particulates and dissolved nutrients to Great Lakes coastal wetlands, which in turn has long lasting effects on the biotic communities that inhabit these areas. The negative effects of agriculture on aquatic systems have been extensively documented (Harding et al. 1999, Gleason et al. 2003, Relyea 2005). I observed changes in the overall invertebrate community composition, as well as a shift in the relative abundance of invertebrate functional feeding groups. The increase in the relative abundance of collector filterers and the decrease in grazers at sites near outlets were presumably caused by increases in turbidity and fine particulate organic matter.

Some agricultural practices more intensely affect nearby aquatic systems than others. Most row crops need to be planted in well drained, nutrient rich soils. Farmers run tiles under the surface of their row crop farm ground to drainage systems in order to eliminate standing water as quickly as possible after a rainfall event or as the ground thaws in the spring. In addition to the inputs of sediment and nutrients, row crop agricultural practices contribute chemicals associated with pesticides and herbicides (Jayne et al. 1998). Intensive farming practices normally involve "Roundup ready" crops, which are tolerant to the effects of herbicides. This allows farmers to spray their fields multiple times during the early phases of the growing season, in order to eliminate unwanted plant species (*i.e.* everything but the row crop). As it pertains to my study, harmful chemicals may have played a role in the observed alteration of community compositions, but a significant reduction of macroinvertebrate species richness was not associated with sites near drainage outlets.

All *Typha* sites were located within 800m of drainage outlets, which made it difficult to attribute macroinvertebrate community composition changes within *Typha* habitat to the presence of a drainage outlet due to the fact that it could not be tested with my data set. Some literature suggests that vegetation type (Olsen et al. 1995, Webb et al. 1984, Angradi et al. 2001) and altered forms of organic matter (Vannote 1980, Harding 1999) potentially have impacts on macroinvertebrate community composition. To help provide more insight on this relationship, I used NMDS analysis on *Schoenoplectus* sites only, and tested the difference between sites near drainage outlets and sites not near drainage outlets. *Schoenoplectus* sites near outlets maintained significantly different macroinvertebrate communities than communities from *Schoenoplectus* wetlands far from drainage outlets. The results from this analysis suggested that drainage outlets play a significant role in altering macroinvertebrate communities of *Schoenoplectus* habitats. This helps support the idea that vegetation type as well as outlet presence have a significant impact on predicting the macroinvertebrate community compositions of Great Lakes coastal wetlands.

My results suggest that boat channels do not play a significant role in altering the macroinvertebrate community compositions or the overall abiotic integrity of Great Lakes coastal wetlands. However, it is likely the altered habitat conditions caused by drainage outlets have made any impacts of boat channels undetectable. The results imply water from drainage outlets affect coastal areas up to 800 meters from the outlet. The excess nutrients and chemicals transported through the drainages of agricultural watersheds are a significant threat to Great Lakes coastal wetland ecosystems, in particular the macrophyte (Keddy and Reznicek 1986, Melzer 1999) and macroinvertebrate (Vannote 1980, Harding 1999) assemblages. As a result, coastal wetlands near outlets support altered abiotic conditions as well as macroinvertebrate communities that are much different than what were originally found in these areas. Significant changes in coastal wetland biota could produce lasting negative effects on the Great Lakes as a whole. Macroinvertebrate abundance and distribution has a direct influence on the prey availability and growth rates for local fish species (Keast 1984, Bottom and Jones 1990). Invertebrates that serve as prey items in Great Lakes coastal wetlands have a significant influence on wetland value and are the building blocks of a multimillion dollar fishing industry.

Undoubtedly, the detrimental effects of boat channels exist in the form of habitat loss and fragmentation. U.S. EPA (2011) estimates 50% of the original Great Lakes coastal wetlands have been lost and the remaining coastal wetland habitats have been significantly degraded through fragmentation, nutrient pollution, and the introduction of exotics. Boat channel management goals should be focused on reducing the loss of habitat by consolidating boat traffic as well as setting regulations against the consistent disruption of relatively pristine wetlands. It is important that we maintain the integrity of the remaining coastal wetlands and make a noble attempt to reclaim some of the wetland area that has been subject to anthropogenic disturbance.

Drain Management

Management of drainage systems should be focused on land usage and the control of non-point source pollution by the implementation of agricultural best management practices (BMP's). BMP's are ways to help control surface erosion, as well as containing nutrient and agricultural chemical pollution (Logan 1993, Cook et al. 1996). Incorporating manure management practices to livestock operations located within transferrable vicinity of the watershed drainage is a cost effective way to reduce nutrient pollution to aquatic systems. Conservation tilling is also an effective practice to reduce erosion and nutrient loss. In some cases conservation tilling may boost crop production yield by increasing soil moisture, but may also cause declines in crop yield due to the competition with undesirable plants (Sharpley and Withers 1994). Working with the agricultural community and implementing BMP's should be the primary management goal pertaining to drainage systems. Simple, cost effective measures such as a 10 meter wide, vegetation buffer strip along the riparian areas of a drainage system could significantly reduce the input of sediments, excess nutrients and harmful chemicals. More extensive management such as the construction of a catchment basin to develop "buffer wetlands" for the filtering of runoff could be an effective tool in regards to watershed input control (Brix and Schierup 1989, Yan et al. 1997). Conservation programs and financial contributions could make these economical efforts possible, with the potential of intensely increasing the ecological value of coastal wetlands. These issues are very important to coastal wetland ecosystem management and should be integrated into basin wide land management practices.

Understanding the impacts of watershed contributions to the Great Lakes and how those inputs affect coastal wetlands should play a large part in promoting a management plan that will contribute the most useful protective measures for these systems. Further implications of this study are to obtain samples from a number of *Typha* sites that are far from drainage systems in order to determine if the macroinvertebrate community compositions from these sites are similar to the communities found at *Typha* sites located near drainage outlets. This information would give us more insight on the influence of drainages on the macroinvertebrate communities living in *Typha* habitat specifically. Coastal wetlands are the last line of defense for the containment of contaminants coming off the landscape entering the Great Lakes. They are important ecosystems that provide services such as nutrient attenuation, carbon cycling, water quality improvement, shoreline erosion protection, floodwater storage and groundwater recharge (Neiring 1978, Howard-Williams 1985, Yan et al. 1997, Choi and Wang 2004). These habitats are also important for the biodiversity and development of fish, invertebrates, amphibians, reptiles, birds and mammals (Goodyear 1982, Whitt 1996, Michael 2003, Taft et al. 2005). The management of anthropogenic disturbance is the greatest challenge Great Lakes managers face. As scientists, it is important that we make information sharing a large part of our overall goal, in order to raise general public awareness of the functions and values that our coastal wetlands provide, and how these systems are negatively impacted by our actions.

LITERATURE CITED

Amsberry L., Baker M.A., Ewanchuk P.J., and Bertness M.D. (1999) Clonal integration and the expansion of *Phragmites australis. Ecological Applications.* 10:4. 1110–1118.

Angradi T.R., Hagan S.M., and Able K.W. (2001) Vegetation type and the intertidal macroinvertebrate fauna of a brackish marsh: *Phragmites* vs *Spartina*. *Wetlands*. 21:1. 75-92.

Bedford K.W. (1992) The physical effects of the Great Lakes on tributaries and wetlands. *Journal of Great Lakes research*. 18, 571-589.

Bottom D.L. and Jones K.K. (1990) Species composition, distribution, and invertebrate prey of fish assemblages in the Columbia River Estuary. *Progress in Oceanography*. 25:1-4. 243-270.

Brix H. and Schierup H.H. (1989) The use of aquatic macrophytes in water-pollution control. *Ambio*. 18:2. 100-107.

Burton T.M., Uzarski D.G., Genet J.A., (2004) Invertebrate habitat use in relation to fetch and plant zonation in northern Lake Huron coastal wetlands. *Aquatic Ecosystem Health and Management Society.* 7 : 249-267.

Burton T.M., (1985) The effects of water level fluctuations on Great Lakes coastal marshes In: *Coastal Wetlands. Proceedings of the First Great Lakes Coastal Wetlands Colloquium* (eds H. H. Prince & F.M. D’Itri) pp. 3-13. Lewis Publishers, Chelsea, Michigan.

Choi Y. and Wang Y. (2004) Dynamics of carbon sequestration in a coastal wetland using radiocarbon measures. *Global Biogeochemical Cycles*. 18. 1-12.

Cook M.G., Hunt P.G., Stone K.C., and Canterberry J.H. (1996) Reducing diffuse pollution through the implementation of agricultural best management practices: a case study. *Water science and technology : a journal of the International Association on Water Pollution Research*. 33: 4-5. 191-196.

Cummins K.W. (1973) Trophic relations of aquatic insects. *Annual review of Entomology*. 18: 183-206.

de Avila A.C., Stenert C. and Maltchik L., (2011) Partitioning macroinvertebrate diversity across different spatial scales in southern Brazil coastal wetlands. *Wetlands*. 31. 459-469.

Gleason R.A., Euliss N.H., Hubbard D.E., and Duffy W.G. (2003) Effect of sediment load on the emergence of aquatic invertebrates and plants from wetland soil egg and seed banks. *Wetlands*. 23:1. 26–34.

Goodyear C.D., Edsall T.A., Ormsby-Dempsey D., Moss G.D., Polanski P.E. (1982) Atlas of spawning and nursery areas of Great Lakes fishes. Vol. 1-A summary of geographic area. FWS/OBS-82/52, U.S. Fish and Wildlife Service, Washington, D.C.

Groffman P.M., (1994) Denitrification in freshwater wetlands. *Current Topics in Wetland Biogeochemistry*. 1. 15-35.

Harding, J. S., Young, R. G., Hayes, J. W., Shearer, K. A. and Stark, J. D. (1999), Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology*. 42: 345–357.

Honnay O. and Jacquemyn H. (2007) Susceptibility of common and rare plant species to the genetic consequences of habitat fragmentation. *Conservation Biology*. 21:3. 823-831.

Hooftman D.A., van Kleunen M., and Diemer M. (2003) Effects of habitat fragmentation on the fitness of two common wetland species, *Carex davalliana* and *Succisa pratensis*. 134. 350-359.

Howard-Williams Clive (1985) Cycling and retention of nitrogen and phosphorus in wetlands: a theoretical and applied perspective. *Freshwater Biology*. 15:4. 391-431.

Jayne D.B., Hatfield J.L. and Meek D.W. (1998) Water quality in Walnut Creek watershed: herbicides and nitrate in surface waters. *Journal of Environmental Quality* 28:1. 45-59.

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

Keast A. (1984) The introduced aquatic macrophyte, *Myriophyllum spicatum* as habitat for fish and their invertebrate prey. *Canadian Journal of Zoology*. 62:7. 1289-1303.

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

Klotz R.L. (1985) Factors controlling phosphorus limitation in stream sediments. *Limnology and Oceanography.* 30:3. 543–553.

Lehtinen R. M., Galatowitsch S. M., Tester J. R., (1999) Consequences of habitat loss and fragmentation for wetland amphibian assemblages. *Wetlands*. 19. 1-12.

Lienert J. and Fischer M. (2003) Habitat fragmentation affects the common wetland specialist *Primula farinose* in north-east Switzerland. *Journal of Ecology*. 91:4. 587-599.

Logan T.J. (1993) Agricultural best management practices for water pollution control: current issues. *Agriculture, ecosystems & environment*. 46:1-4. 223–231.

Lougheed V.L., Crosbie B. and Chow-Fraser P. (2001) Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin: latitude, land use, and water quality effects. *Canadian Journal of Fisheries and Aquatic Sciences*. 58:8. 1603-1612.

MacArthur R.H., and Wilson E.O. (1963) An equilibrium theory of insular zoogeography. *Evolution*. 17:4. 373–387.

Melzner A. (1999) Aquatic macrophytes as tools for lake management. *Hydrobiologia*. 395-396:0. 181–190.

Merritt R.W. and Cummins K.W., (1996) An introduction of the aquatic insects of North America. Third Edition. Kendall/ Hunt Publishing Company.

Michael, J. H. (2003). Nutrients in salmon hatchery wastewater and its removal through the use of a wetland constructed to treat off-line settling pond effluent. *Aquaculture*. *226*(1/4), 213-225.

Neiring W. (1978) Wetland Values. *Proceedings of the National Wetland Protection Symposium*. 6. 7-8

Olsen E.J., Engstrom E.S.,Doeringsfield M.R., and Bellig R. (1995) Abundance and distribution of macroinvertebrates in relation to macrophyte communities in a prairie marsh, Swan Lake, Minnesota. *Journal of Freshwater Ecology*. 10:4. 325 –335.

Peckarski B.L., Fraissinet P.R., Penton M.A., Conklin D.J., (1990) Freshwater macroinvertebrates of northeastern North America. Comstock Publishing Associates.

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

Rathcke B.J., and Jules E.S., (1993) Habitat fragmentation and plant pollinator interactions. *Current Science*. 65. 273-277.

Relyea R.A. (2005) The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. *Ecological Applications*. 15:2. 618-627.

Resources Inventory Committee, (1999) British Columbia Estuary mapping System. Land Use Coordination Office for the Coastal Task Force Report, Resources Inventory Committee, Victoria B.C.

Sharpley A.N. and Withers P.J. (1994) The environmentally-sound management of agricultural phosphorus. *Fertilizer Research*. 39. 133–146.

Simon T.P. (1998) Modification of an index of biotic integrity and development of reference condition expectations or dunal, palustrine wetland fish communities along the southern shore of Lake Michigan. *Aquatic Ecosystem Health and Management*. 1:1. 49-62.

Taft, O. W., & Haig, S. M. (2005). The value of agricultural wetlands as invertebrate resources for wintering shorebirds. *Agriculture, Ecosystems & Environment*. *110*(3/4), 249-56.

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

U.S. Environmental Protection Agency. (2011) http://www.epa.gov/solec/indicator_sheets/coastal_wetlands.pdf

Uzarski D.G., Burton T.M., Kolar, R.E., and Cooper M.J., (2009) The ecological impacts on fragmentation and vegetation removal in Lake Huron's coastal wetlands. *Aquatic Ecosystem Health and Management Society*. 12,1.45-62.

Vannote R.L., Minshall W.G., Cummins K.W., Sedell J.R., and Cushing C.E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*. 37:1. 130-137.

Webb N.R., Clarke R.T., and Nicholas J.T. (1984) Invertebrate diversity on fragmented *Calluna*-heathland: effects of surrounding vegetation. *Journal of* Biogeography. 11. 41-46.

Wei A., Chow-Fraser P., and Albert D., (2004) Influences of shoreline features on fish distribution in the Laurnetian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 61, 7. 1113-1123.

Wisheu I. C., and Keddy P.A., (1991) Seeds banks of a rare wetland plant community: distribution patterns and effects of human induced disturbance. *Journal Vegetation Science*. 2. 181-188.

Whitt M.B., (1996) Avian breeding use of coastal wetlands on the Saginaw Bay of Lake Huron. M.S. Thesis. Department of Fisheries and Wildlife, Michigan State University, East Lansing, Mi.

Yan W., Yin C., and Tang H. (1997) Nutrient retention by multipond system: Mechanisms for the control of nonpoint source pollution. *Journal of Environmental Ecology*. 27:5. 1009–1017.

CHAPTER II

IMPACTS OF DRAINAGE OUTLETS AND BOAT CHANNELS ON THE HABITAT CONDITIONS AND FISH COMMUNITIES OF GREAT LAKES COASTAL WETLANDS

Introduction

Great Lakes coastal marshes are important habitats for fish, amphibians, reptiles, invertebrates, birds and mammals (Goodyear et al. 1982, Jude and Pappas 1992, Michael 2003, Taft et al. 2005) and are among some of the most productive and biologically diverse ecosystems on the planet. An estimated 50 percent of Great Lakes coastal wetlands have been lost across the entire Great Lakes basin with some areas having lost up to 90 percent of their coastal wetlands (U.S. EPA 2011). These areas are exposed to natural water level fluctuations of varying degree. Seasonal water level fluctuations (20 to 40cm) and short term weather-induced water level changes (10 to 20cm) are commonly observed across Great Lakes coastal wetlands (Bedford 1992). Over long periods (years to decades), water levels of the Great Lakes have fluctuated up to 150 cm. This environmental variability has had major effects on the plant, invertebrate and fish communities found in coastal wetlands (Burton 1985, Keough et al. 1999).

Long term declines in water levels have exposed Great Lakes wetlands to increasing anthropogenic disturbance that disrupt coastal ecosystems. Shoreline landowners see periods of lowered levels as an opportunity to develop areas for commercial navigation (i.e., boat channels) and protect property from coastal erosion (i.e., seawalls). This causes Great Lakes coastlines to be subjected to intensive development, which potentially threatens these coastal wetland ecosystems and their functions. When wetland vegetation is removed, the physical and chemical characteristics within the disturbed area become altered due to the fact this ecosystem is more exposed to open water conditions. As a result, the biological community also shifts due to the change in habitat conditions (Uzarski et al. 2009). These physical alterations have a dramatic effect on important processes that take place in wetlands such as denitrificaiton, a valuable component of nutrient flow in wetlands (Groffman 1994).

The anthropogenic reduction of natural water fluctuations through the manipulation of lake levels is thought to have a negative effect on macrophyte species richness and diversity (Albert and Minc 2004). These physical interactions have impacted many types of wetland inhabiting organisms that depend on the natural fluctuation of Great Lakes water levels (Keddy and Reznicek 1986, Chow-Fraser et al. 1998). In particular, anthropogenic disturbances promote the success of invasive plant species. These alterations in wetland structure play a significant role in promoting species assemblages that are quite different from the original fauna presumed to inhabit these coastal wetland areas (Wisheu et al. 1991).

Alterations in plant communities give rise to significant changes in invertebrate communities within manipulated wetlands (Olsen et al. 1995, Angradi et al. 2001, Burton 2004, de Avila et al. 2011). These community changes can have negative implications on the population dynamics of higher trophic levels due to the fact that invertebrates are important food sources for Great Lakes fishes and migrating waterfowl (Murkin and Batt 1987, Bottom and Jones 1990, Mallory et al. 1994, Marklund et al. 2002). Waterfowl spend a large portion of their time in and around wetlands and depend on quality wetland habitat conditions to survive. A harvestable population of these migrants provides hundreds of jobs throughout the nation (Ducks Unlimited 2009). The loss or degradation of coastal wetland habitat has the potential to negatively affect migratory waterfowl populations inhabiting Great Lakes coastal wetlands (Prince et al. 1992). Recreational jobs and licensing revenue is lost as a result and the local economy is negatively affected. The state of Michigan in 2008 sold 1,259,129 fishing licenses, which grossed 22,203,876 dollars (Mi. DNR, 2009). Ninety five percent of commercially targeted fish and shellfish species are dependent on wetland habitats (U.S. EPA 2011). These relationships make wetland resources of great economical and social importance to the state of Michigan and the interactions between people and wetland ecosystems should be a significant concern for wetland, fish, and wildlife managers.

Habitat Fragmentation

These important transitional zones between the terrestrial landscape and open water are often subjected to detrimental human impacts. Disturbance from creating boat channels, beach grooming, and lake viewing have all contributed to an increase in wetland habitat fragmentation. The fragmentation of coastal wetland habitat produces an increased edge to center ratio of the remaining wetland vegetation. Edge wetland habitats support much lower plant densities comprised of individuals with lower fecundity when compared to plants living in the center of a wetland (Lienert and Fischer 2003). A loss of genetic diversity within some fragmented wetland plant communities has been documented due to a lowered number of reproductive individuals in the isolated population. The seclusion of wetland plant communities can have a significantly negative impact on habitat quality of the remaining wetland (Hooftman et al. 2003). The loss of complexity and connectivity among wetland habitats is detrimental to the diversity and richness of many indigenous amphibian species (Lehtinen et al. 1999). The abundance and diversity of pollinating insects using wetland habitat is negatively affected by fragmentation. As a result, plant species in the remaining habitat exhibit a reduced pollination rate, which gives rise to lower seed production (Rathcke and Jules 1993).

Fish have also been negatively affected by habitat fragmentation. Due to the high frequency of stream habitat fragmentation (i.e., damns), many stream fish populations have suffered the effects of isolation. A survey of the cutthroat trout *Oncorhynchus clarki henshawi* in Nevada revealed these threatened fish were present in 89% of the un-fragmented stream basins, but were only present in 32% of the isolated stream basins (Dunham et al. 1997). Although the effects of isolation in a Nevada river are different than in coastal wetlands, Great Lakes fish may be negatively impacted by habitat fragmentation as well.

More than 80 Great Lakes fish species use coastal wetland habitat and over 50 of these species are dependent upon Great Lakes coastal wetland habitats for a portion of their life cycle (Jude and Pappas 1992, Wilcox 1995). Boat channels are a very common type of coastal wetland fragmentation throughout the Great Lakes, especially in regions were human population is prevalent along lake shorelines. State and city municipalities construct boat channels and marinas in order to create revenue and to provide local residents and tourists with access to the Great Lakes. In areas with private landowners occupying shoreline, boat channels are normally widespread. Most landowners with boats prefer to have a personal dock on their property, which in many cases are coupled with a boat channel. These areas for watercraft navigation are created by the excavation of wetland soil or from the mechanical removal of wetland plants. In some instances where small residential lots line the shore, up to 8 personal boat channels were observed within 600 meters of coastal wetland habitat. This practice not only heavily fragments coastal wetland habitat, but eliminates valuable coastal wetland area. Research pertaining to Great Lakes coastal wetlands provides insight on how anthropogenic disturbance, in the form of boat channels, affect the abiotic conditions and fish communities of wetlands adjacent to impacted areas.

Over half of the boat channels surveyed in this study were associated with launches that had been constructed on the banks of drainage outlets. These watershed drainage systems were commonly channelized with drainage tiles emptying into them in order to quickly drain the agricultural land that they cut through. Current intensive farming practices contribute large amounts of nutrients, chemicals and sediment to surrounding watersheds and in many instances have been observed to degrade aquatic habitats, increase the rate of nutrient enrichment, or eutrophication, and negatively affect the biota living in these areas (Klotz 1985, Harding et al. 1999, Gleason et al. 2003, Sharpley et al. 2003, Relyea 2005). Macrophyte community composition has been known to change due to the altered habitat conditions attributed to agricultural land use (Egerson et al. 2004).

Literature suggests that macrophyte structure has a direct impact on abundance and community structure of fish observed in littoral zones. In southern Ontario, areas with more dense submerged aquatic vegetation contained a much higher abundance of fish than did areas with sparse or no submerged macrophytes (Randall et al. 1996). Habitat containing dense vegetation provides higher food availability and better refugia from predators for highly sought after prey fish species than does sparsely or un-vegetated habitats (Crowder and Cooper 1982). Most research pertaining to fish use to vegetation density and habitat complexity, but a correlation between fish community composition and the dominant vegetation type has not been addressed thoroughly.

Objectives and Hypotheses

The main objective of this study was to evaluate how boat channels affect coastal wetland habitat conditions and the fish communities using the wetlands adjacent to these disturbances. I hypothesized that boat channels within Great Lakes coastal wetlands have a significant impact on the abiotic conditions and fish communities adjacent to these disturbances. I used species diversity data to compare fish assemblages from boat channels and adjacent wetlands. Increased levels of diversity and richness are thought to be indicators of quality habitat (Gotelli and Colwell 2001). Species community composition, Shannon diversity index, richness and abundance were used to assess how boat channels affect wetland fish assemblages.

My second hypothesis was to determine if the dominant emergent wetland vegetation type is a predictor of fish community composition in coastal wetlands. Uzarski (2005) found vegetation type to be the most effective forecaster of fish communities in wetlands. If fish prefer specific invertebrates as prey items, prey availability along with vegetation may influence habitat usage by fish assemblages. Four wetland vegetation habitat types in southern Minnesota were documented to support significantly different invertebrate communities (Olsen et al.1995). Since invertebrate communities are often correlated with vegetation type, macrophyte structure could be indirectly responsible for determining fish habitat selection in wetlands.

My third hypothesis was that the presence of a drainage outlet had a significant impact on the habitat conditions and fish communities of wetlands near outlets. Research has documented that increased eutrophication of freshwater systems has decreased overall fish diversity, altered community compositions, and decreased prey availability for some important game fish species (Hayward and Margraf 1987, Helminen et al. 1998, Tammi et al. 1999).

This study provides valuable information on the biological, physical and chemical responses to boat channel disturbance throughout Great Lakes wetland habitats. It will give insight into how habitat manipulation and anthropogenic inputs affect wetland habitat and fish communities. By acquiring and applying these data, managers will be able to use the information from this study to aid in building models and other habitat assessment techniques specific to wetland production and value. Identifying trends in habitat conditions and biological communities will allow habitat managers to more accurately predict the effects that boat channels and other coastal wetland disturbances have on the overall quality of Great Lakes coastal wetland habitat.

Methods

Sample Sites

Throughout the study period (June 17 – August 18, of 2009, 2010, and 2011) I monitored habitat conditions and fish community compositions of various Great Lakes coastal wetlands in northern Lake Huron, northern Lake Michigan as well as in Saginaw Bay (Figure 13). Soil, water, physical and chemical habitat data were collected at all 43 coastal wetland sites.

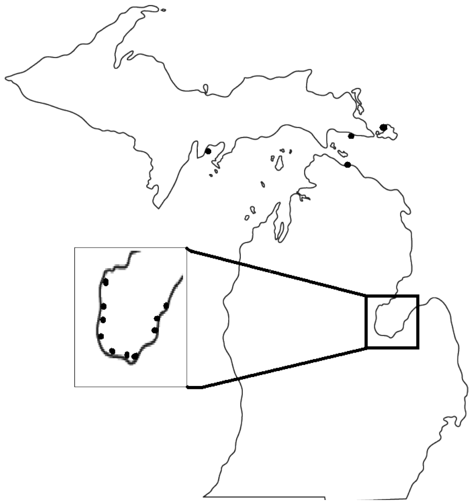

Figure 12.Black dots represent coastal wetland habitats that were sampled.

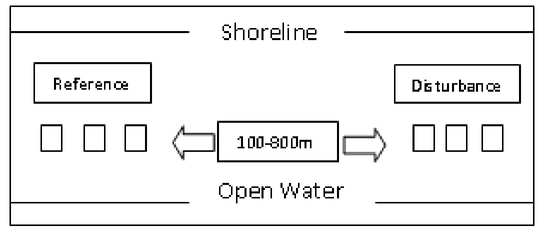

Figure 13. Small boxes represent replicate fyke net locations within each wetland site.

Boat Channels

Twenty three of the sampled wetlands were located immediately adjacent to boat channels. Each of the "disturbed" wetland sites was paired with a reference site that was not directly affected by boat traffic or habitat fragmentation. Undisturbed or "reference" sites were located between 100 and 800 meters from the disturbed wetland site (Figure 13). I paired each disturbed site with a reference site in order to establish a state of initial conditions and to serve as the control for this experiment. There were only 20 reference sites because some areas had multiple boat channels that were sampled within the 800 m range of a reference. In order to isolate the disturbance as the responsible constituent for changes observed in biotic and abiotic conditions, characteristics such as depth, vegetation type, and wave exposure were kept consistent between disturbed and reference site pairings. Reference sites were then selected in the most closest replicate wetland area in either direction of the boat channel. Disturbed sites were located in the wetland vegetation as close to the boat channel as possible. Sites were always located just inside the wetland, near the outer edge of the emergent wetland zone.

Sites were categorized based on the dominant emergent vegetation type present. All sites sampled in this study were easily designated as being dominated by *Schoenoplectus* or *Typha* due to the fact that there was virtually no mixing of the two plant taxa at any site. Sites were then delineated based on their location relative to drainage outlets. Sampling locations that were located within 800m of the mouth of a drainage ditch were designated as being “near” outlets whereas sites greater than 1500m from drainage outlets were categorized as being “far” from drainage outlets.

Habitat Characteristics

The first step when arriving at a site was to collect data on the physical and chemical characteristics of the water without disrupting water chemistry characteristics. Basic physical parameters were collected from the middle of the water column at each site using a water quality sonde (Yellow Springs International, model 6600 V2). Water quality data were collected from the boat before any other characteristics were measured minimize influence from my presence. The YSI data sonde was used to measure temperature, pH, turbidity (NTU’s), chlorophyll *a* (µg/L), oxidation–reduction potential (mV), specific conductance (µs/cm), and dissolved oxygen (mg/L). Two water samples were collected in 500 mL acid washed bottles and were stored on ice. These water samples were used to measure alkalinity and dissolved nutrient levels for each wetland. Alkalinity was determined by titrating 100 mL of unfiltered water with 0.02 M sulfuric acid (H2SO4) to a pH of 4.5. The other water sample was filtered through 0.45µm

Millipore filters, stored in a new acid washed bottle and frozen for lab analysis of nutrient content. Measurements of dissolved nutrients included soluble reactive phosphorus (SRP), NO3-N, and NH4-N, which were detected in the lab using a BRAN+LUEBBE QuAAtro auto-analyzer. For all samples that had a value below detection limit, the detection limit of that specific parameter was divided by half and this value was used for data analysis. The detection limit on the Bran+Luebbe QuAAtro for SRP, nitrate, and ammonium was 0.3μg/L, 1.0 μg/L, and 0.5 μg/L, respectively.

Soil samples were collected at each site using a 47 mm diameter core sampler. The top 10 cm of substrate was placed into a Ziploc bag and stored on ice until they were frozen upon return to the lab. For analysis, soil samples were thawed, and then dried at 60 ºC for a minimum of 24 hours to remove all water content. The dry samples were weighed and then ashed in a muffle oven at 550 ºC for 24 hours in order to burn off all organic content present in the sample. The remaining inorganic content of the samples was reweighed to determine the percentage of organics present in the soil at each site.

Physical components such as water depth and organic depth were measured three times at each site using a standard aluminum meter stick. Vegetation stem density was also measured three times at each site using a 0.25 x 0.25 m quadrat made of 2.5 cm diameter tubing that was haphazardly placed over wetland vegetation. The number of plants within each quadrat was enumerated and then the three replicates were averaged in order to determine the stem density per quarter meter squared at each site. The amount of wave exposure (modified effective fetch) was calculated using the British Columbia Estuary Mapping System (Eq. 3) (Resource Inventory Committee 1999; e.g. Burton et al. 2004). This made it possible to quantify the amount of wave action at each site and include the modified effective fetch from each site as one of our 16 measured chemical/physical parameters in the assessment of habitat conditions.

Eq. 3.

]

Fm = Modified effective fetch
F = Fetch distance
45L = 45^{o} left of perpendicular to the shoreline
45R = 45^{o} right of perpendicular to the shoreline

Where Fm represents modified effective fetch and each F is the effective fetch measured at the angle given in Eq. 3. (45^{o} or 90^{o}) (Left or Right) in respect to the shoreline. Google Earth™ (2010) was used to measure all fetch distances.

Fish sampling

Fish assemblages were sampled from the same location abiotic data were collected. Fyke nets were set in replicates of three at each dominant emergent vegetation zone with a minimum 10 meters distance between each net in order to avoid capture competition between nets. Two sizes of fyke nets with a mesh size of 4.8 mm were used in this study to optimize the efficiency of fish capture in a variety of water depths. When wetland water depth was between 20 and 50 cm, small fyke nets (45.72 x 91.44 cm) were used. Wetlands that had water depths between 50 and 100 cm were sampled using larger fyke nets (121.92 x 91.44 cm). Water less then 20cm deep was not sampled because fish in this shallow water could not get past the throat of the net in order to be caught. Reference and disturbed site pairings were always fished with the same size net because they had similar water depths.

Nets were set so the mouth of the net was facing the wetland and the 7.3 m lead of the net was fishing inside the emergent wetland vegetation. The 1.8 m wings of the net were set at 45° angles outward from the mouth of the net. Each net was set for one net-night in order to survey fish taxa using the habitat at varying times of the day. All fish captured in the fyke nets were measured (total length), enumerated, identified to species, and released. Any fish caught that were unidentifiable were humanely euthanized by sedation in MS-222 and preserved in formalin to be identified later in the lab using a dichotomous fish key (Corkum 2010).

Data analysis

Biological community data were analyzed with non-metric multidimensional scaling (NMDS), which was used to plot fish community composition data. A multi-response permutation procedure (MRPP) was then implemented to determine if the designated site groups of data were statistically similar. Principal components analysis (PCA) was used to compare the habitat conditions between sites in order to determine whether disturbances or vegetation types had an impact on the abiotic characteristics of sites. These statistical analyses were conducted using PC-ORD version 5 (MjM Software, Glenden Beach, Oregon, U.S.A.). Paired t-tests were conducted on abiotic data to establish if there were any significant differences in the physical and chemical characteristics of disturbed and reference wetlands (Minitab version 16, Minitab Inc., U.S.A.). An alpha value of 0.05 was used to establish significance in all statistical tests. Species richness was tabulated for each site, which is the total number of species collected. Abundance was calculated by adding the total number of fish caught from each of the three nets. Shannon diversity (*H'*) was computed as a calculation that combines species richness and evenness to represent the proportional representation of species at a given site.

Eq. 4.

H' = Shannon diversity index
P_i = Proportion of the entire population made up of taxa
S = number of species encountered
∑ = sum from all species

Each of these community assessment parameters are associated with habitat quality so paired t-tests were conducted in order to help provide insight on the differences between the fish communities sampled at disturbed and reference sites.

Results

Boat Channels

Differences between the physical and chemical characteristics among reference and disturbed sites were determined from paired t- tests (Table 3). Average turbidity levels at disturbed sites were 9.22 NTU and turbidity at reference sites averaged 3.61 NTU. Alkalinity was higher at reference sites with an average of 148.13 (mg/L) while disturbed sites produced an average of 136.52 (mg/L). Soil organic levels were higher at disturbed sites with an average of 3.07% whereas reference site soils averaged 2.36%. Water temperature tended to be warmer at reference sites with an average of 25.22ºC at reference sites and 24.83ºC at disturbed sites although the difference was not significant (p = 0.081).

Table 3. Paired t-test were conducted on the data collected from 23 reference and disturbed site pairings during the summers of 2009 - 2011. Each disturbed site was located in wetland vegetation located immediately adjacent to a boat channel. The reference sites paired with each of the disturbed sites were located 100 – 800 meters from the disturbance.

Parameter	R&D P Value
Dissolved Oxygen	0.834
Turbidity	0.041
Specific Conductance	0.252
Alkalinity	0.003
Redox Potential	0.135
pH	0.195
Chlorophyll a	0.188
Water Temperature	0.081
Stem Density	0.773
Organic Depth	0.164
NH4	0.155
NO3	0.334
SRP	0.248
% Organics	0.031
Fish Richness	0.066
Fish Shannon	0.174
Fish Abundance	0.795

Principal component analysis of site abiotic characteristics showed no trends or patterns based on reference and disturbed site groupings (Figure 14). Axis 1 accounted for 18.5% of the variation observed in the data. Increased specific conductance and heightened levels of chlorophyll *a* were the characteristics associated with sites on the positive side of axis 1. Sites that fell out on the negative side of axis 1 were related to high soil organic content and increased water temperature. Axis 2 was responsible for 16.9% of the disparity observed in the abiotic data. Measurements associated with the positive side of axis 2 had increased redox potential and nitrate concentration. Sites that fell out on the negative side of axis 2 were linked with deeper organic depths and high turbidity readings.

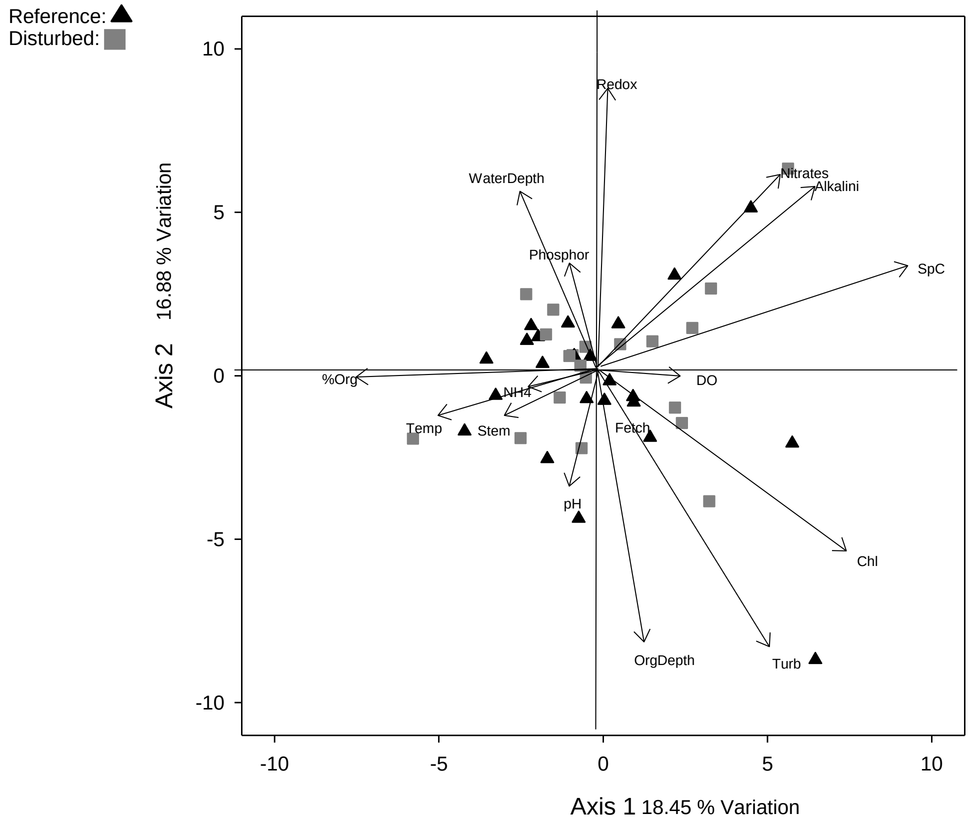

Figure 14. Principal component analysis displaying the similarities of 43 sites based on 16 abiotic characteristics. Sites with more similar chemical and physical habitat conditions are plotted more closely on the bi-plot.

I collected 12,057 fish during the summers of 2009, 2010, and 2011. Fyke net sampling of the 20 reference sites that were selected produced 6,538 fish representing 32 species. Sampling of 23 disturbed wetland sites produced 5,519 fish representing 38 species. The average Shannon diversity index score was 0.522 (std.err. = 0.035) and 0.587 (std. err. = 0.048) for reference and disturbed sites, respectively. The average fish richness at reference sites was 7.65 (std.err. = 0.726) and 9.39 (std. err. = 0.863) at disturbed sites. A paired t-test on the fish species richness of reference and disturbed sites did show a relative difference between site types (p = 0.066). Although the difference of fish richness between reference and disturbed sites was statistically insignificant, a trend of increased richness at disturbed sites was evident. Of the 23 reference and disturbed site pairings, 15 disturbed sites had higher fish species richness than its reference site. Six reference sites had higher richness than the paired disturbed site and 2 sites had the same number of fish species (Figure 15). NMDS analysis portrayed no detectable difference in the fish communities of reference and disturbed sites (MRPP t = 1.14, p = 0.915) (Figure 16).

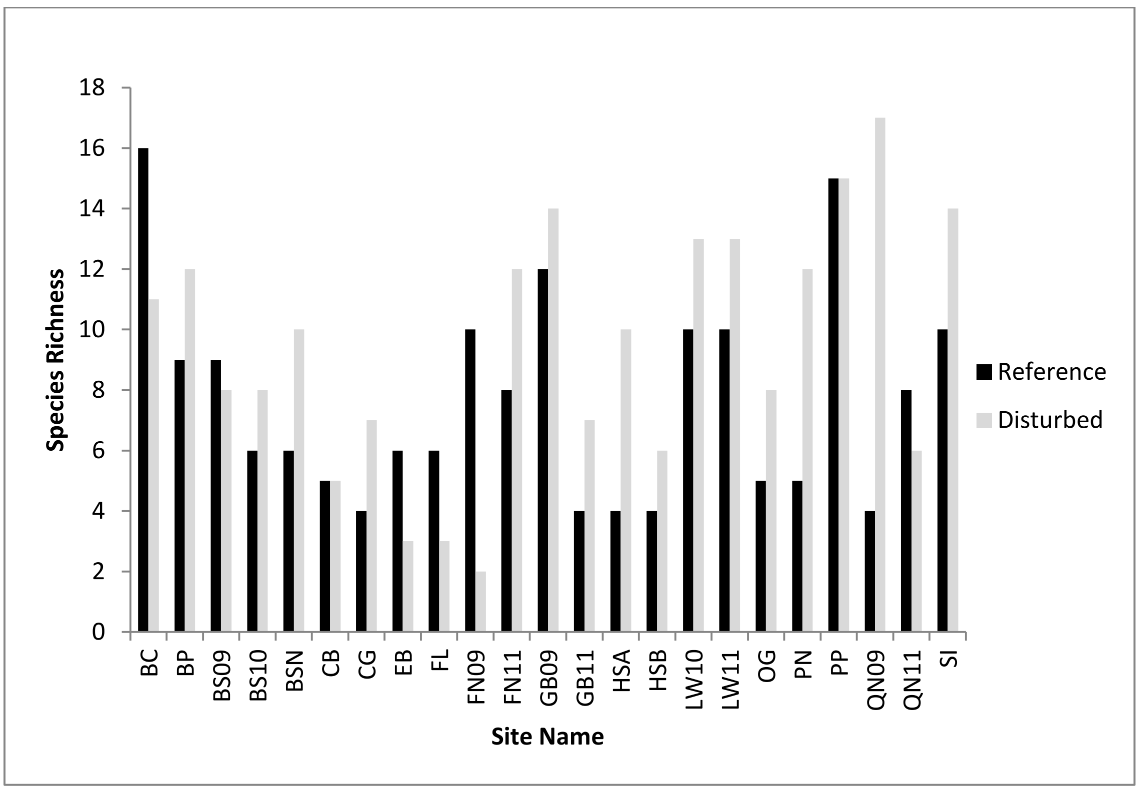

Figure 15. Bar chart showing the fish species richness of 23 site pairings. Fyke net sampling of reference and disturbed sites produce a total of 43 Great Lakes fish species. In most instances (15 of 23), disturbed sites produced a more diverse fish community. There were 6 instances when the reference site had a more diverse fish community than its disturbed site. Two sites had the same number of species when comparing reference and disturbed communities.

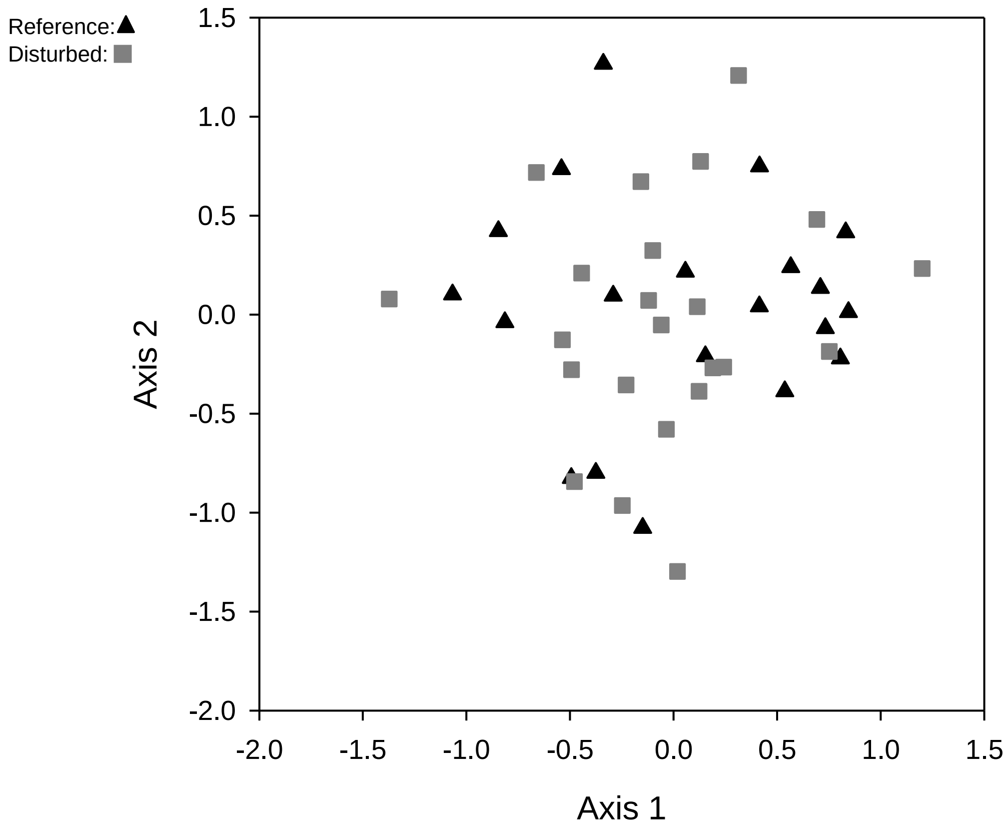

Figure 16. NMDS bi-plot showing the distribution of reference and disturbed sites based on the fish communities collected at each site using fyke nets. Most reference and disturbed site pairs were plotted relatively close to one another which suggest that the differences in the fish communities inhabiting reference and disturbed site are similar.

Vegetation Type

Principal components analysis displayed a noticeable difference in the physical and chemical characteristics of different vegetation types (Figure 17). *Typha* sites were associated with higher levels of specific conductance and chlorophyll *a*. *Schoenoplectus* sites had a larger percentage of organics in the soil, higher stem density and increased water temperatures.

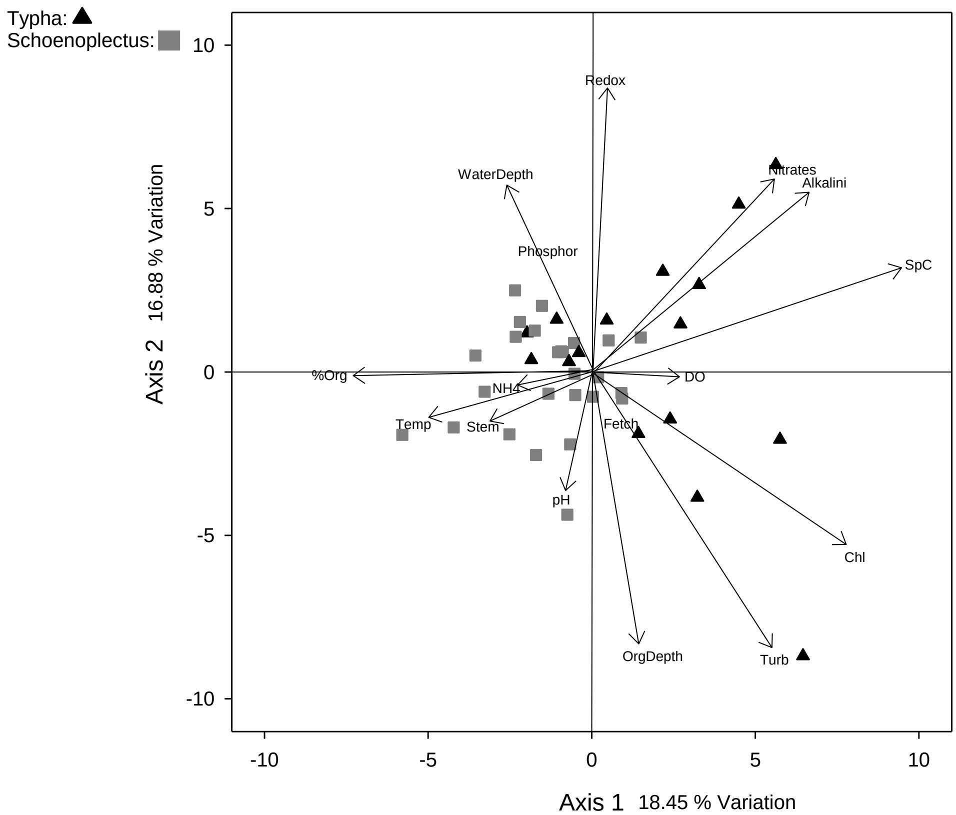

Figure 17. Principal component analysis bi-plot displaying the difference in habitat characteristics of *Schoenoplectus* and *Typha* sites based on 16 measured parameters. Sites with the most similar physical and chemical characteristics were located in the closest proximity on the bi-plot. *Typha* and *Schoenoplectus* habitats tended to sustain vastly different habitat conditions.

Although habitat conditions were unique based on the dominant type of emergent vegetation type at each site, fish community composition did not show any trends or patterns based on macrophyte types. *Typha* and *Schoenoplectus* sites contained statistically similar fish communities (MRPP t = 0.771, p = 0.767) (Figure 18).

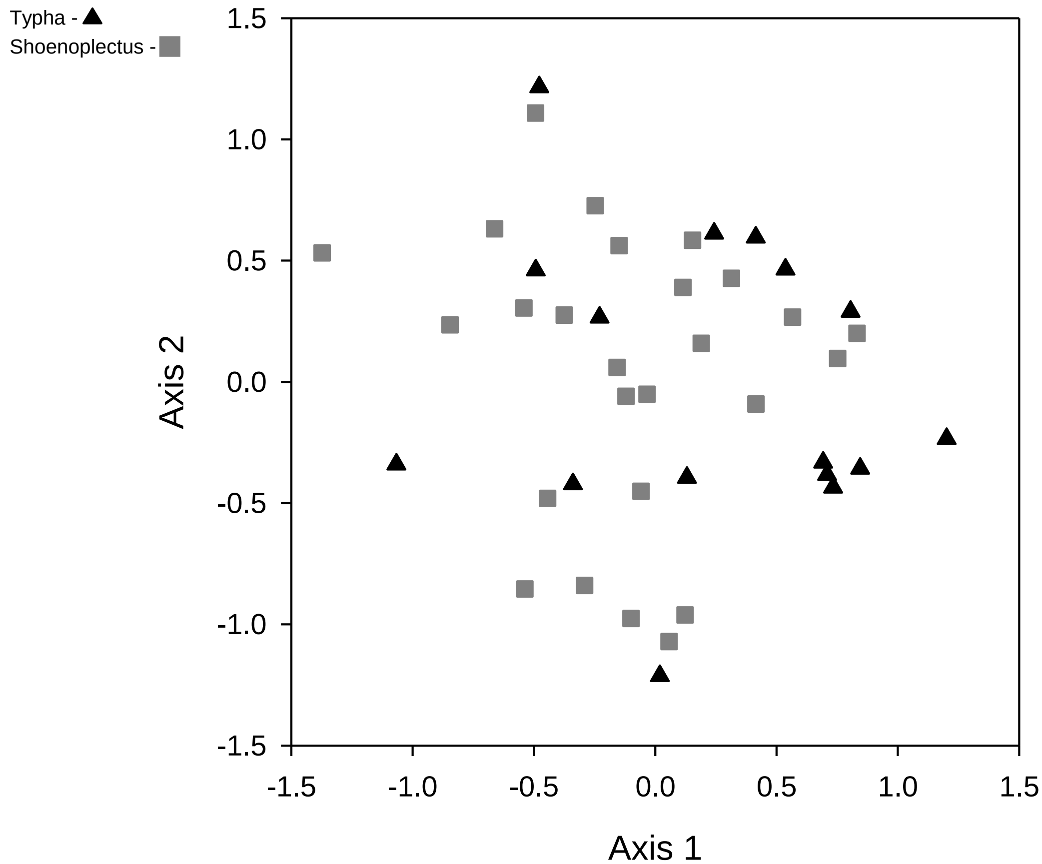

Figure 18. Non-metric multidimensional scaling of the fish communities collected during the summers of 2009, 2010, and 2011. The somewhat random distribution of sites on the bi-plot suggests that *Typha* and *Schoenoplectus* habitats support similar fish community compositions.

Drainage Outlets

Principal components analysis of sites located near drainage outlets (<800 m) had very unpredictable physical and chemical characteristics when compared to sites that were located in areas far from drainage outlets (>1500 m) (Figure 19). Sites located near drainage outlets had higher nitrates, alkalinity, chlorophyll *a* and turbidity, whereas sites far from outlets were associated with more moderate habitat condition levels with the exception of one site pair having very dense stem density and warm water temperatures.

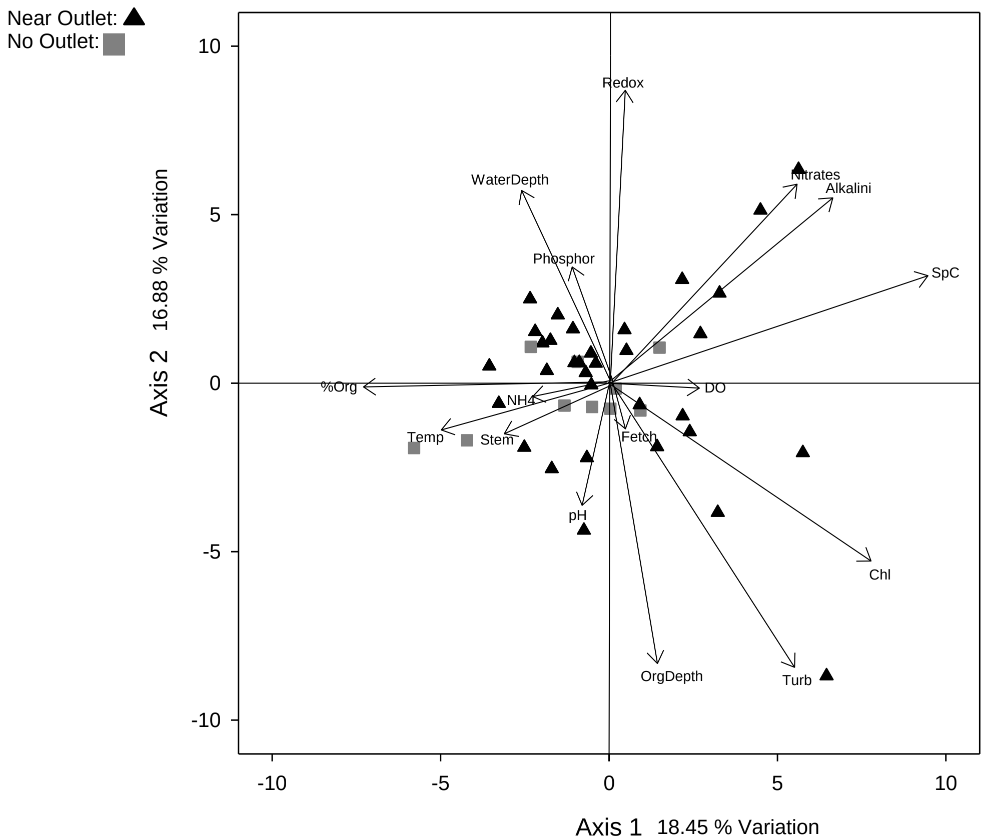

Figure 19. Principal component analysis showing the distribution of sites based on their location in proximity to watershed drainage outlets. Sites near outlets are highly variable when considering a wide variety of parameters. One pair of sites not located near an outlet gravitated away from the center point due to the fact that it had a noticeably higher stem density, water temperature and percentage of soil organics.

The fish communities collected near drainage outlets were not significantly different from the communities found at sites distant from drainage outlets, but the two site groupings did show some apparent differences in overall community composition (MRPP t = 0.918, p = 0.157) (Figure 20). Fish species associated with the negative side of axis 1 were black bullheads (*Ameiurus melas*), bluntnose minnows (*Pimephales notatus*) and sand shiners (*Notropis stramineus*). Fish associated with the positive side of axis 1 were central mudminnows (*Umbra limi*), gizzard shad (*Dorosoma cepedianum*), and golden shiners (*Notemigonus crysoleucas*). Taxa associated with the negative side of axis 2 were green sunfish (*Lepomis cyanellus*) and northern pike (*Esox lucius*). Species associated with the positive side of axis 2 were the freshwater drum (*Aplodinotus grunniens*), spotted gar (*Lepisosteus oculatus*), round goby (*Neogobius melanostomus*) and rock bass (*Ambloplites rupestris*).

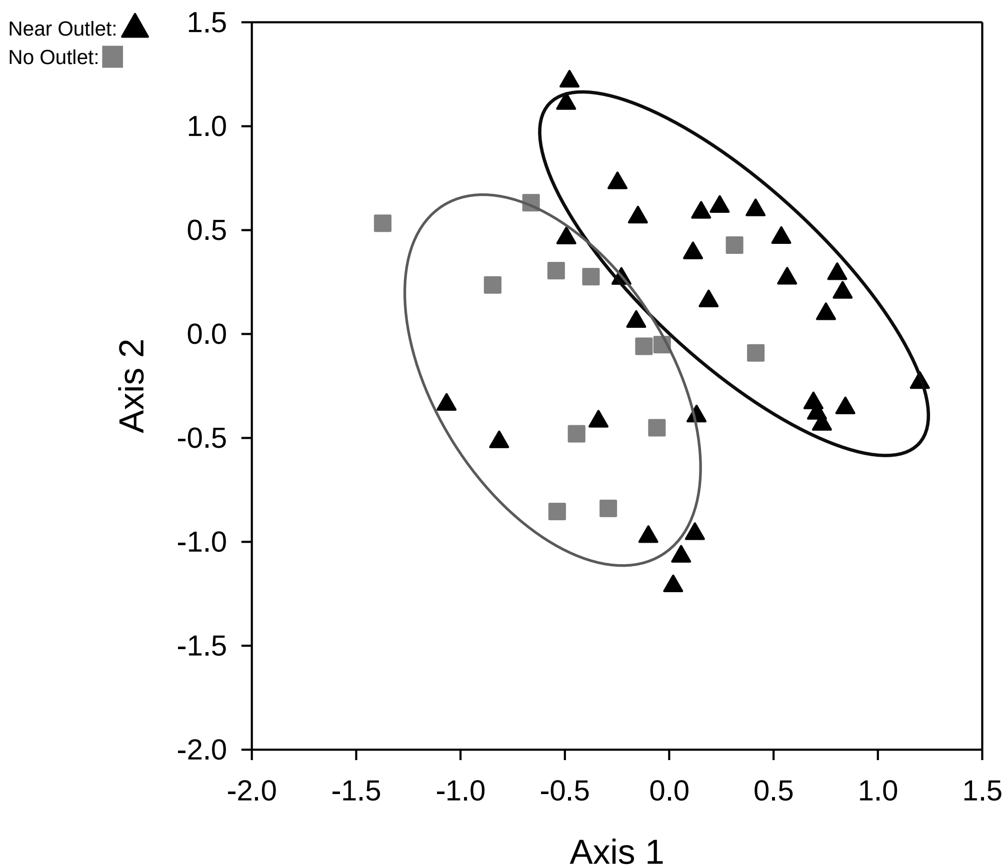

Figure 20. Non-metric multidimensional scaling bi-plot showing the distribution of sites based on fish community compositions collected in coastal wetland habitat. Although the two site groupings were not statistically different, stratification of compositions near outlets and far from outlet did seem evident.

Schoenoplectus Sites

Fyke net sampling of *Schoenoplectus* sites near outlets were not statistically different from *Schoenoplectus* sites not near drainage outlets (MRPP t = -1.43, p = 0.089), but stratification between the two groups was evident (Figure 21). Fish species associated with the negative side of axis 1 were the johnny darter (*Etheostoma nigrum*), golden shiner (*Notemigonus crysoleucas*), and white sucker (*Catastomus commersonii*). Taxa linked with the positive side of axis 1 were the brook stickleback (*Culaea inconstans*), spotted gar (*Lepisosteus oculatus*) and bluegill (*Lepomis macrochirus*). Fish species related to the negative side of axis 2 were the central mudminnow (*Umbra limi*) and northern pike (*Esox lucious*). Sites falling out on the positive side of axis 2 were associated with having channel catfish (*Ictalurus punctatusi*), bluntnose minnows (*Pimephales notatus*) and banded killifish (*Fundulus diaphanus*).

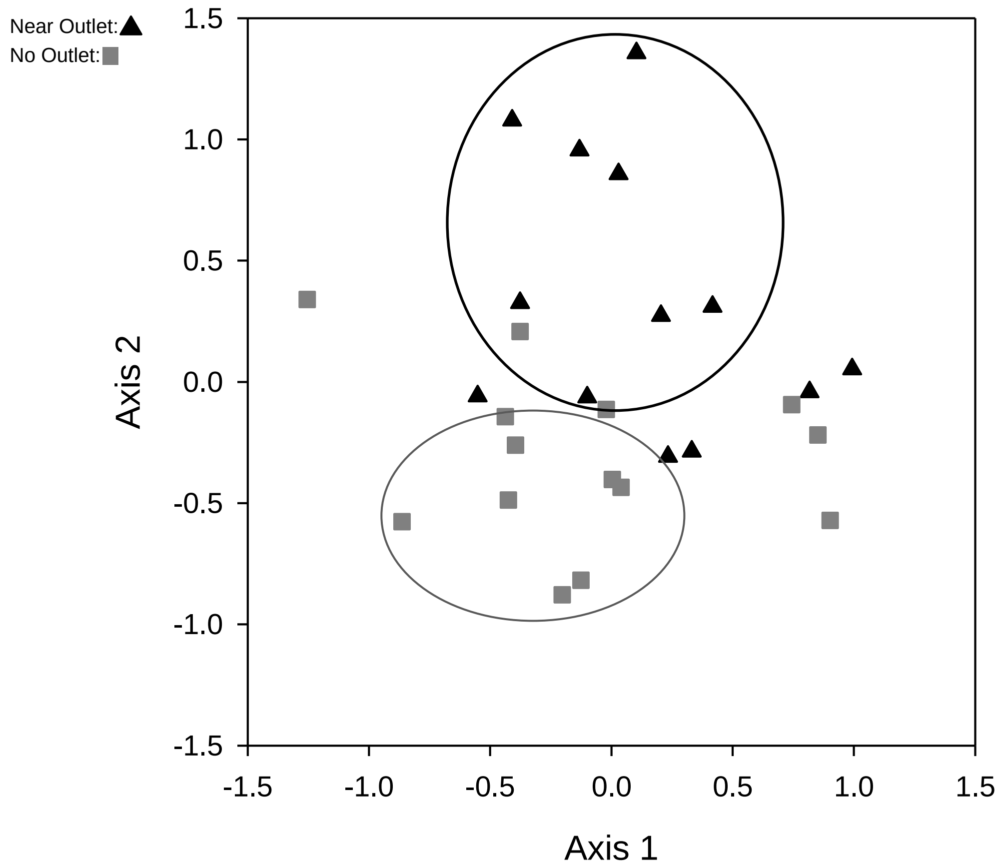

Figure 21. Non-metric multidimensional scaling of *Schoenoplectus* sites with fish communities color coded based on their location in relationship to drainage outlets. Sites designated as near outlet and no outlet were not statistically different but grouping of sites did seem apparent.

Discussion

I analyzed the habitat characteristics and the fish communities of coastal wetlands near boat channels. These habitats displayed significantly higher turbidity levels when compared to reference sites. Altered water clarity at impacted sites was probably due to a combination of influences, but primarily boat traffic, nutrient inputs and suspended sediments from agricultural runoff through drainage canals. As stated previously, 15 of the 23 boat channels were coupled with a drainage outlet and in most cases these systems cut through an agriculturally dominated landscape. It is relatively unclear how fish communities respond to increased turbidity, but the literature suggests that the responses to changes in water clarity are species and system specific. Some systems have shown to exhibit fish species composition shifts in areas where turbidity is increased (Brazner and Beals 1997) and that fish communities dominated by visual predators, frequently shifted to communities dominated by cyprinids with increased turbidity (Sandstrom and Karas 2002). Others have found that increased turbidity in freshwater environments is needed for the success of small planktivorus fish, however, this turbidity must be due to plankton and not inorganic sediment. High turbidity is less than for optimal for large piscivores due to the inability of visual predators to locate prey (Utne-Palm 2002). Trebitz et al. (2007) found that most of the fish species inhabiting Great Lakes coastal wetlands were tolerant or moderately tolerant of relatively high turbidity, with only a few species being intolerant.

Alkalinity was significantly lower at impacted sites, which was likely due to the increased mixing of pelagic water into the wetland from the increased amount of edge habitat. Water temperature tended to be higher at reference sites although a significant p-value was not observed during paired t-test analysis. Of the 23 reference and disturbed site pairings, 18 disturbed sites had lower water temperatures than reference sites. This was also likely a result of increased pelagic water mixing at disturbed sites due to the increased amount of edge habitat created from the boat channel. The percentage of soil organics in wetlands near boat channels was significantly higher as well. The presence of drainage outlets probably contributed to the observed heightened levels of organics in the soils near boat channels. Boat traffic may have also increased the soil organics in these areas due to the fact that boat propellers chop up emergent and submergent wetland plants that eventually become part of the soil composition. Although these habitat conditions were altered, it is presumed that the fish communities are not greatly impacted by these changes. It is important to collect in situ habitat data when sampling, but this information only gives a snapshot of these variables. Monitoring the biological communities living in coastal wetlands is important because it gives researchers a more comprehensive indication of the habitat conditions at a particular site.

Non-metric multidimensional scaling analysis of fish communities collected from reference and disturbed sites suggested that the presence of a boat channel did not have a significant impact on the fish assemblages. Overall, fish richness tended to be higher at impacted sites with paired t-test results approaching a statistical difference in richness between reference and disturbed sites. The increased amount of edge habitat near boat channels may have played a role in the increased richness of fish using these areas. The edge effect principle (Wiens 1976) is an ecological concept that relates the transitional areas between habitats with heightened species richness, a widely accepted principal among ecologists (Harris 1988). Although the edge effect principal has been widely observed among many terrestrial habitat types, while observing a wide range of species, organism richness increases in edge habitat has not been frequently documented in aquatic systems. Tanner (2005) determined the crustaceans living in seagrass habitat were statistically more abundant near edges than in the middle of a habitat patch, but richness was not measured. The change in bathymetry caused by some boat channels may have also played a role in the increased fish richness at impacted sites. Pittman (2007) concluded that bathymetric variability was the number one predictor of saltwater fish richness. Richness likely increased near the artificial edges by creating an area where wetland obligates and pelagic communities overlap. This is not likely positive since it is at the expense of wetland obligate habitat that is much more rare than pelagic habitat.

It has been documented that the dominant vegetation type is also a strong predictor of the fish communities that live in a particular habitat (Dean et al. 2000, Uzarski et al. 2005). In contrast to these findings, fish communities collected from *Typha* and *Schoenoplectus* vegetation in this study were statistically similar and did not support distinct fish communities. This may have been due to the fact that other habitat characteristics were more influential in shaping the observed fish communities, but detailed analyses are needed to detect any other responsible habitat variables. Non-metric multidimensional scaling of fish community compositions near outlets was not significantly different far from outlets communities, but the stratification of sites based on location in relation to drainages was obvious. The mobility of fish, in conjunction with the sampling of different vegetation types, may have produced just enough variation in the data to dampen significant conclusions based on fish habitat selection. Analysis of the fish communities collected at *Schoenoplectus* sites exhibited an even more evident stratification of sites based on the relative distance from drainages (Figure 21). It was not possible to evaluate the effects of drainage outlets on the fish communities at *Typha* sites due to the fact that all sites dominated by *Typha* were located near outlets, so a comparison could not be made.

Aquatic habitats subjected to high nutrient inputs experience large increases in phytoplankton abundance, which shade out submerged macrophytes and negatively affect their success. Agricultural induced eutrophication has decreased aquatic plant species richness, as well as shifted wetland communities from being dominated by submerged macrophytes to an emergent macrophyte dominated community (Lougheed 2001, Egerson et al. 2004). In some cases, large algal blooms have been responsible for the loss of submerged macrophytes completely (Morris et al. 2003). Motor boat traffic has been observed to be directly responsible for the loss of submerged macrophyte density, height and percent coverage due to physical disruption (Asplund and Cook 1997). This loss of habitat is detrimental to the success of local fish species. Areas that contain more dense submerged vegetation provide higher food availability, better cover for juvenile fish hiding from predators and relatively abundant and diverse fish communities (Crowder and Cooper 1982, Randall et al. 1996).

The agricultural degradation of Finnish inland water resources has produced alterations in their freshwater fish communities, with overall fish composition shifting toward cyprinid dominance (Tammi et al. 1999). It is unclear if the increased turbidity and habitat degradation associated with drainage systems has a positive or negative influence on the fish communities of coastal wetlands, but an altered community seems to be present in wetlands near drainages. Large amounts of nutrients and agricultural chemicals' being put into aquatic systems has been detrimental to other aquatic organisms as well. Nutrient pollution, toxic algal blooms, anoxic periods, extreme turbidity, harmful chemical addition and rapid sedimentation rates are all issues related to agricultural runoff that negatively impact the productivity and biodiversity of aquatic systems (Klotz 1985, Harding et al. 1999, Rouse et al. 1999, Rabalais 2002, Gleason et al. 2003, Havens et al. 2003, Relyea 2005).

The implementation of agricultural best management practices (BMP's) has proven to be effective in dampening the effects of intensive agriculture on nearby aquatic systems (Brix and Schierup 1989, Logan 1993, Sharpley and Withers 1994, Cook et al. 1996, Yan et al. 1997). These practices range in intensity from maintaining a strip of vegetation along the riparian areas of drainages, to creating "buffer wetlands", which act as filters for nutrients and other agricultural pollutants. Maintaining riparian vegetation is very easy and cost effective, although it is frequently not incorporated into agricultural practices, even in areas within relatively close proximity to the Great Lakes. Sustaining riparian vegetation has shown to produce positive results pertaining to overall habitat quality and fish community health in watersheds where this BMP was incorporated (Wichert and Rapport 1998, Wang et al. 2006).

The conclusions from this study suggest that boat channels change the habitat conditions of coastal wetlands, but drastic changes in fish communities were not observed, which was presumably due to the high level of fish mobility. The construction of boat channels results in lost wetland area and leaves the remaining habitat fragmented. In many instances, boat traffic should be consolidated in order to lessen the degree of fragmentation in areas where multiple channels are bisecting the same wetland complex. High quality wetlands should be treated as such and boat traffic should be eliminated from these areas. Watershed land management should be of immense interest to Great Lakes fish and wildlife managers. Simple land management techniques should be implemented in order to dampen the negative effects of agriculture on Great Lakes coastal wetland habitat, fish and fauna especially in areas near the Great Lakes shoreline. The manipulation, fragmentation and elimination of coastal wetlands should be strictly managed along all Great Lakes shoreline and prohibited in the remaining high quality coastal wetlands.

LITERATURE CITED

Albert D.A. and Minc L.D. (2004) Plants as regional indicators of Great Lakes coastal wetland health. *Aquatic Ecosystem Health and Management*. 7:2. 233-247.

Angradi T.R., Hagan S.M., and Able K.W. (2001) Vegetation type and the intertidal macroinvertebrate fauna of a brackish marsh: *Phragmites* vs *Spartina*. *Wetlands*. 21:1. 75-92.

Asplund T.R. and Cook C.M. (1997) Effects of motor boats on submerged aquatic macrophytes. *Lake and Reservoir Management*. 13:1. 1-12.

Bedford K.W. (1992) The physical effects of the Great Lakes on tributaries and wetlands. *Journal of Great Lakes Research*. 18, 571-589.

Bottom D.L. and Jones K.K. (1990) Species composition, distribution, and invertebrate prey of fish assemblages in the Columbia River Estuary. *Progress in Oceanography*. 25:1-4. 243-270.

Burton T.M., (1985) The effects of water level fluctuations on Great Lakes coastal marshes. In: *Coastal Wetlands. Proceedings of the First Great Lakes Coastal Wetlands Colloquium* (eds H. H. Prince & F.M. D'Itri) pp. 3-13. Lewis Publishers, Chelsea, Michigan.

Brazner J.C. and Beals E.W. (1997) Patterns in fish assemblages from coastal wetland and beach habitats in Green Bay, Lake Michigan: a multivariate analysis of abiotic and biotic forcing factors. *Canadian Journal of Fisheries and Aquatic Sciences*. 54:8. 1743-1761.

Brix H. and Schierup H.H. (1989) The use of aquatic macrophytes in water-pollution control. *Ambio*. 18:2. 100-107.

Burton T.M., Uzarski D.G., Genet J.A., (2004) Invertebrate habitat use in relation to fetch and plant zonation in northern Lake Huron coastal wetlands. *Aquatic Ecosystem Health and Management Society.* 7: 249-267.

Chow-Fraser P., Lougheed V., Le Theic V., Crosbie B., Simser L. and Lord J. (1998) Long-term response of the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario. *Wetlands Ecology and Management*. 6:1. 19-42.

Cook M.G., Hunt P.G., Stone K.C., and Canterberry J.H. (1996) Reducing diffuse pollution through the implementation of agricultural best management practices: a case study. *Water science and technology: a Journal of the International Association on Water Pollution Research*. 33: 4-5. 191-196.

Corkum L.D., (2010) Fishes of essex county and surrounding waters. University of Windsor. Friesens Corporation, Altona, Manitoba.

Crowder L.B. and Cooper W.E. (1982) Habitat structural complexity and the interaction between bluegills and their prey. *Ecology*. 63:6. 1802-1813.

de Avila A.C., Stenert C. and Maltchik L. (2011) Partitioning macroinvertebrate diversity across different spatial scales in southern Brazil coastal wetlands. *Wetlands*. 31. 459-469.

Dean T.A., Haldorson L., Laur D.R., Jewett S.C., and Blanchard A. (2000) The distribution of nearshore fishes in kelp and eelgrass communities in Prince William Sound, Alaska: associations with vegetation and physical habitat characteristics. *Environmental Biology of Fishes.* 57. 271-287.

Ducks Unlimited. http://www.ducks.org/ContactDU.aspx

Dunham J.B., Vinyard G.L., and Rieman B.E. (1997) Habitat fragmentation and extinction risk of Lahontan cutthroat trout. *North American Journal of Fisheries Mangement*. 17.4: 1126-1133.

Egerson C.J., Kopaska J.A., and Downing J.A. (2004) A century of change in macrophyte abundance and composition in response to agricultural eutrophication. *Hydrobiologia.* 524. 145-156.

Gleason R.A., Euliss N.H., Hubbard D.E., and Duffy W.G. (2003) Effect of sediment load on the emergence of aquatic invertebrates and plants from wetland soil egg and seed banks. *Wetlands*. 23:1. 26 – 34.

Goodyear C.D., Edsall T.A., Ormsby-Dempsey D., Moss G.D., and Polanski P.E. (1982) Atlas of spawning and nursery areas of Great Lakes fishes. Vol. 1-A summary of geographic area FWS/OBS-82/52, U.S. Fish and Wildlife Service, Washington, D.C.

Gotelli N.J. and Colwell R.K (2001) Quantifying biodiversity: Procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*. 4: 379-391.

Groffman P.M., (1994) Denitrification in freshwater wetlands. *Current Topics in Wetland Biogeochemistry*.1. 15-35.

Harding, J. S., Young, R. G., Hayes, J. W., Shearer, K. A. and Stark, J. D. (1999), Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology*. 42: 345–357.

Harris L.D. (1988) Edge effects and conservation of biotic diversity. *Conservation Biology*. 2:4. 330-332.

Havens K.E., James T.R., East T.L., and Smith V.H. (2003) N:P ratio, light limitation, and cyanobacterial dominance in a subtropical lake impacted by non-point source nutrient pollution. *Environmental Pollution*. 122. 379-390.

Hayward R.S., and Margraf J.F. (1987) Eutrophication effects on prey size and food available to yellow perch in Lake Erie. *Transactions of the American Fisheries Society*. 116:2. 210-223.

Helminen H., Karjalainen J., Kurkilahti M., Rask M., Sarvala J. (1998) Eutrophication and fish biodiversity in Finnish lakes. 27th Congress in Dublin 1998. Proceedings. 27:1. 194 – 199.

Hooftman D.A., van Kleunen M., and Diemer M. (2003) Effects of habitat fragmentation on the fitness of two common wetland species, *Carex davalliana* and *Succisa pratensis*. 134. 350-359.

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

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

Keough J. R., Thompson T.A. Guntenspergen G. R. & Wilcox D. A. (1999) Hydrogeomorphic factors and ecosystem responses in coastal wetlands on the Great Lakes. *Wetlands.* 19. 821-834.

Klotz R.L. (1985) Factors controlling phosphorus limitation in stream sediments. *Limnology and Oceanography.* 30:3. 543–553.

Lehtinen R. M., Galatowitsch S. M., Tester J. R., (1999) Consequences of habitat loss and fragmentation for wetland amphibian assemblages. *Wetlands*. 19. 1-12.

Lienert J. and Fischer M. (2003) Habitat fragmentation affects the common wetland specialist *Primula farinose* in north-east Switzerland. *Journal of Ecology*. 91:4. 587-599.

Logan T.J. (1993) Agricultural best management practices for water pollution control: current issues. *Agriculture, Ecosystems & Environment*. 46:1-4. 223 – 231.

Lougheed V.L., Crosbie B. and Chow-Fraser P. (2001) Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin: latitude, land use, and water quality effects. *Canadian Journal of Fisheries and Aquatic Sciences*. 58.8: 1603-1612.

Mallory M.L., Blancher P.J., Weatherhead P.J., and McNicol D.K. (1994) Presence or absence of fish as a cue to macroinvertebrate abundance in boreal wetlands. *Hydrobiologia*. 279-280: 345-351.

Marklund O., Sandsten H., Hansson L.A. and Blindow I. (2002) Effects of waterfowl and fish on submerged vegetation and macroinveterbrates. *Freshwater Biology*. 47. 2049-2059.

Michael, J. H. (2003). Nutrients in salmon hatchery wastewater and its removal through the use of a wetland constructed to treat off-line settling pond effluent. *Aquaculture*. *226*(1/4). 213-25.

Michigan DNR. http://www.michigan.gov/dnr/0,1607,7-153-10364---,00.html

Morris K., Bailey P.C., Boon P.I., and Hughes L. (2003) Alternative stable states in the aquatic vegetation of shallow urban lakes.II. Catastrophic loss of aquatic plants consequent to nutrient enrichment. *Marine and Freshwater Research*. 54. 201-215.

Murkin H.R. and Batt B.D.J. (1987) Interactions of vertebrates and invertebrates in peatlands and marshes. *Memoirs of the Entomological Scoiety of Canada*. 119:(140) 15-30.

Olsen E.J., Engstrom E.S.,Doeringsfield M.R. and Bellig R. (1995) Abundance and distribution of macroinvertebrates in relation to macrophyte communities in a prairie marsh, Swan Lake, Minnesota. *Journal of Freshwater Ecology*. 10:4. 325 –335.

Pittman S.J., Christensen J.D., Caldow C., Menza C., Menaco M.E. (2007) Predictive mapping of fish species richness across shallow-water seascapes in the Caribbean. *Ecological Modeling*. 204:1-2. 9-21.

Prince H.H., Padding P.I., and Knapton R.W. (1992) Waterfowl use of the Laurentian Great Lakes. *Journal of Great Lakes Research*. 18:4. 673 – 699.

Rabalais N.N. (2002) Nitrogen in aquatic systems. *Ambio: A Journal of the Human Environment.* 31:2. 102-112.

Randall R.G., Minns C.K., Cairns V.W. and Moore E.J. (1996) The relationship between an index of fish production and submerged macrophytes and other features at three littoral areas in the Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*.53:S1. 35-44.

Rathcke B.J., and Jules E.S., (1993) Habitat fragmentation and plant pollinator interactions. *Current Science*. 65. 273-277.

Relyea R.A. (2005) The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. *Ecological Applications*. 15:2. 618-627.

Rouse J.D., Bishop C.A., and Struger J. (1999) Nitrogen pollution: an assessment of its threat to amphibian survival. *Environmental Health Perspectives*. 107:10. 799-803.

Sandstrom A. and Karas P. (2002) Effect of eutrophication on young of year freshwater fishes communities in coastal areas of the Baltic. *Environmental Biology of Freshwater Fishes*.63: 89-101.

Sharpley A.N. and Withers P.J. (1994) The environmentally-sound management of agricultural phosphorus. *Fertilizer Research*. 39. 133 – 146.

Sharpley A.N., Daniel T., Sims T., Lemunyon J., Stevens R., and Perry R. (2003) Agricultural phosphorus and eutrophication. United Stated Department of Agriculture. http://ddr.nal.usda.gov/bitstream/10113/26693/1/CAT30907360.pdf

Tammi J., Lappalainen A., Mannio J., Rask M., and Vourenmaa J. (1999) Effect of eutrophication on fish and fisheries in Finnish lakes: a survey based on random sampling. *Fisheries Management and Ecology*. 6:3. 173 – 186.

Tanner J.E. (2005) Edge effect on fauna in fragmented seagrass meadows. *Austral Ecology*. 30. 2: 210-218.

Taft, O. W., & Haig, S. M. (2005). The value of agricultural wetlands as invertebrate resources for wintering shorebirds. *Agriculture, Ecosystems & Environment*. *110*(3/4), 249-56.

Trebitz A.S., Brazner J.C., Brady V.J., Axler R. and Tanner D.K. (2007) Turbidity tolerances of Great Lakes coastal wetland fishes. *North American Journal of Fisheries Management*. 27:2. 619-633.

U.S. Environmental Protection Agency (2011) http://www.epa.gov/solec/indicator_sheets/coastal_wetlands.pdf

Utne-Palm A.C. (2002) Visual feeding of fish in a turbid environment: Physical and behavioral aspects. *Marine and Freshwater Behavior and Physiology*.35:1-2. 111-128.

Uzarski D.G., Burton T.M., Cooper M.J., Ingram J.W., and Timmermans S.T.A. (2005) Fish habitat use within and across wetland classes in coastal wetlands of the five Great Lakes: Development of a fish-based index of biotic integrity. *Journal of Great Lakes Research*. 31:S1. 171-187.

Uzarski, D.G., Burton, T.M., Kolar, R.E., and Cooper, M.J., (2009) The ecological impacts on fragmentation and vegetation removal in Lake Huron's coastal wetlands. *Aquatic Ecosystem Health and Management Society*. 12,1.45-62.

Wang L., Lyons J., Kanehl P. (2006) Habitat and fish responses to multiple agricultural best management practices in a warm water stream. *Journal of the American Water Resources Association*. 42:4. 1047-1062.

Wichert G.A. and Rapport D.J. (1998) Fish community structure as a measure of degradation and rehabilitation of riparian systems in an agricultural drainage basin. *Environmental Management*. 22:3. 425-443.

Wiens J.A. (1976) Population responses to patchy environments. *Annual Review of Ecology and Systematics*. 7. 81-120.

Wilcox D.A. (1995). The role of wetlands as nearshore habitat in Lake Huron. *The Lake Huron Ecosystem, Fisheries and Management*, eds. M. Munawar, T. Edsall, and J. Leach. pp. 223-245. Amsterdam: SPB Academic Publishing.

Wisheu I. C., and Keddy P.A. (1991) Seeds banks of a rare wetland plant community: distribution patterns and effects of human induced disturbance. *Journal of Vegetation Science*. 2. 181-188.

Yan W., Yin C., and Tang H. (1997) Nutrient retention by multipond system: Mechanisms for the control of nonpoint source pollution. *Journal of Environmental Ecology*. 27:5. 1009–1017.