SPATIAL AND SEASONAL VARIABILITY IN CRANE CREEK, A DIKED FRESHWATER ESTUARY COMPLEX TRIBUTARY TO WESTERN LAKE ERIE

by

Jennifer Marie Pfaff

A thesis submitted
in partial fulfillment of the requirements
for the degree of Master of Science
(Natural Resources and Environment)
at the University of Michigan
August 2012

Thesis Committee:

Professor Michael Wiley, Chair

Catherine Riseng, Ph.D.

External reviewer:

Kurt Kowalski, Ph.D.

USGS Great Lakes Science Center, Ann Arbor

Abstract

This study examined ecological variation in a diked wetland complex located at the mouth of Crane Creek, a tributary to Lake Erie located in northwest Ohio, USA. The study examined nine locales: five diked wetland pools (four of which were completely isolated from the estuary and one of which was connected to the estuary by a water control structure) and four reaches of the Crane Creek channel estuary from roughly 6 km upstream of the lake down to where the channel meets Lake Erie. In late summer and late fall of 2011 water quality data (TDS, specific conductance, turbidity, chlorophyll concentration, and blue-green algae concentration) and invertebrate community composition samples were collected. Data was analyzed to compare similarities and differences among the nine study sites.

I found that the diked wetland pool sites had a high amount of variability but that they were generally more similar to one another in all parameters than they were to the creek reaches (with the exception of the reach located farthest upstream). I found that the three creek reaches closest to the lake were all very similar to one another, while the most upstream reach was more similar to some of the diked wetland sites. The diked wetland unit that has been connected to the estuary by a water control structure had the greatest number of invertebrate taxa and had some water quality parameters that were similar to the creek (e.g. phytoplankton density) and others that were dissimilar to the creek (e.g. specific conductance, TDS). This hydrologically connected unit was also the only diked pool in which Dreissenid mussels were found.

Acknowledgements

I would like to thank Dr. Mike Wiley for his assistance and guidance throughout this research project. His kindness and patience were essential to my completion of this research. I would also like to thank Dr. Catherine Riseng for her constructive and supportive comments and guidance during my thesis writing. Thank you also to Dr. Kurt Kowalski at USGS for providing helpful comments and insights throughout my thesis project.

Many thanks to those who helped in my data collection, including those at USGS, Margie Chriscinske, Mike Eggleston and Sean Green, as well as Maureen Lynch and Nick Coleman. Their help enabled my data collection to be quick and efficient! I would also like to thank the Wiley lab for use of lab equipment, help with preliminary investigations, and for help with identification questions. I would like to thank the Ottawa National Wildlife Refuge staff and managers for accommodating my data collection and giving me access to refuge records to help in the writing of my thesis. Thank you to Kyung Seo Park for helping me to find access to all of the resources I could have needed.

I would like to thank those who lent personal support as well as academic insights while I completed my thesis project: I thank my roommates and friends for their friendship and support, there are a too many to name. Thank you to my family, especially my mother and grandmother; I would not have been able to accomplish this without their constant encouragement and limitless confidence. Finally I would like thank Nick Coleman for putting up with me through this adventure and helping me in countless ways.

Contents

Introduction

In its natural state, the coastline of the western basin of Lake Erie consisted of marshy wetlands behind barrier beaches, variously interrupted by freshwater estuaries (Herdendorf 1992). To the west laid the Great Black Swamp, a 4000 km^2 expanse of wooded and marshy wetland occupying postglacial Erie lakeplain terraces. However, beginning in the early 1800s with European colonization, this area was drained and cleared for settlement and agriculture, leaving less than 200 km^2 of the original swampland (Herdendorf 1987, Brookhout et al. 1989). Around the turn of the century much of the remaining coastal marshland was purchased by hunting clubs and protected from draining in order to protect the waterfowl that use this area as a migration pathway. Since then, much of this land has been transferred to federal and state control (Herdendorf 1987).

Lake Erie coastal marshlands differ from inland palustrine and lacustrine marshes in that they are subjected to the more substantial physical forces of a Great Lake shoreline. These forces include the more energetic wave action associated with a larger lake fetch, strong coastal currents, large seichal oscillations commonly approaching a meter in amplitude, and long term changes in water levels due to regional variations in precipitation, evaporation and runoff (Herdendorf 1987, Bedford 1992, Brookhout et al. 1989). All of this dynamic action affects both physical and biological characteristics of Lake Erie coastal marshes. Wave action can uproot plants and prevent the establishment of vegetation in places. Seiches can interrupt and reverse the transport of sediments, nutrients, and other dissolved solids through estuaries (Herdendorf 1987, Keough et al. 1999, USGS 2012). Long term lake water level changes cause shifts in the habitable

zone for wetland plants and invertebrate communities (Herdendorf 1987, Gatham and Burton 2011, Kreiger 1992). However, these forces are also necessary for marshes to form and thrive. Most marshes flourish in areas that are physically protected to some extent, usually behind barrier beaches or sand spits.

Natural barriers can be overcome and washed away in years when water levels are exceptionally high or during particularly strong storm events. This is problematic because the natural processes that would have recreated the barrier may have been interrupted by shoreline modifications elsewhere (Kowalski and Wilcox 1999). Because of this some coastal marshes that have lost natural barriers are today protected artificially by manmade dikes and can no longer be classified as "coastal marshes" as they no longer fill that ecological role (Herdendorf 1987, Brookhout et al. 1989, Kowalski and Wilcox 1999, Albert et al. 2005). Manmade dikes are thought to offer a good alternative for wetland managers for the preservation of marshland because they are similar to natural barriers in that they reduce the effects of wave action in a marsh (Herdendorf 1987), and most remaining marshes located near Lake Erie are protected by them (Johnson et al. 1997, Brookhout et al. 1989). These dikes also allow managers to control the water levels in the marshes to encourage vegetation communities that are attractive to waterfowl (Herdendorf 1987, Johnson et al. 1997).

However, diking coastal wetlands also poses problems. Dikes disconnect coastal marshes from the lake and river estuaries, blocking natural fish access to the marshes (Johnson et al. 1997). Many fishes use marshes as feeding, spawning and nursery habitat, as well as for cover from predators (Herdendorf 1987, Herdendorf 1992, Jude and Pappas 1992, Wei et al. 2004). Cvetkovic and Chow-Fraser (2011) estimate that less than 30

percent of pre-European-contact wetlands remain accessible to fishes in the Great Lakes basin, and over half of the former coastal Lake Erie wetlands in Ohio are now behind dikes (Jude and Pappas 1992). Some advocate for the reconnection of diked coastal wetlands to estuaries or the lake so that fishes may enter the wetlands (USGS 2012). While others have found that diked wetlands may not provide the full suite of functions that fish historically utilized in coastal marshes, and therefore removing dikes may not benefit as many fish taxa as was historically the case (Johnson et al. 1997). Using dikes to disconnect wetlands from coastal tributaries can also be disruptive to larger scale nutrient cycling, resulting in more of the nutrients carried by tributary streams to end up in the lake, rather than being transformed or otherwise retained in the wetland (Mitsch and Wang 2000).

Diked wetlands have also been found to support a somewhat different vegetation community than that found in undiked wetlands (Herrick and Wolf 2005, Theit 2002). For example, some invasive species are better able to invade diked wetlands (e.g. purple loosestrife; Herrick and Wolf 2005) while others are better able to invade undiked wetlands (e.g. *Phragmites*; Thiet 2002).

This study focused on the diked wetland complex located at the Ottawa National Wildlife Refuge (ONWR), Oak Harbor, Ohio. Crane Creek, a small tributary to Lake Erie, flows through ONWR and is included in the Maumee River Area of Concern (USEPA). The U.S. Geological Survey Great Lakes Science Center (GLSC) is currently carrying out a detailed study exploring the effects of hydrologic reconnection of two diked wetland units with the Crane Creek estuary. The GLSC project is studying the effects of this reconnection on water quality as well as fish, benthic, and avian

communities in the two adjacent units (USGS 2012). The goal of my thesis research was to provide a broader ecological context for the GLSC study by examining conditions in the greater estuary setting at ONWR (including a larger set of the diked wetland units and a longer extent of the Crane Creek estuary channel) as well as to provide more information on the overall variability of diked wetlands which may be applied to other such sites in the region where reconnection of diked wetlands to naturally fluctuating water sources may be an option.

Understanding the spatial and seasonal variability of these systems is important for many reasons. In the GLSC study at OWNR evaluating reconnection, intensive but spatially limited sampling of two adjacent experimental units is being used to make inferences about management implications for the larger collection of diked units comprising the refuge and similar sites in the region. However, we currently have almost no information on how representative the GLSC study units are, nor do we understand the longitudinal variation that occurs within the Crane Creek estuary system (Kowalski 2010). A better understanding of variability among the units will also help to inform managers about which units have habitats that are best suited to important taxa in the region as well as which habitats are most likely to be influenced by invasive taxa. This in turn should help managers think about which units might be good candidates for future reconnection. It is also important to determine how much longitudinal variation occurs within lower Crane Creek since the nature of water exchanged in any reconnection scheme would depend on the physical locale of the unit relative to water being transported into and out of the Crane Creek estuary. Better information on the longitudinal structure of the estuary could also inform current operational decisions about

pump placement when moving water from the creek into the diked units. When the full range of variability within ONWR is understood and appreciated, managers will be better able to make decisions on the best steps forward to achieve refuge goals.

I examined chemistry, phytoplankton density and the invertebrate community composition in the lower Crane Creek channel and in five diked wetland pools with the specific objective of understanding of the degree of variation that currently occurs in these diverse and fragmented habitats. I focused on phytoplankton and invertebrate assemblages as indicators of the biological community structure. As primary producers, phytoplankton provide an important part of the base of the aquatic foodweb, and thus are relevant to the wildlife production goals of ONWR. Furthermore phytoplankton production is sensitive to both the amount of nutrients available in the water and light availability. Therefore I expected to see significant variation between sites related to differences in nutrient loading, local nutrient cycling, and sediment transport capacity. Previous studies have shown that there are high nutrient concentrations in runoff from agricultural land in the region (Richards and Baker 2002, Kasat 2006). Within Crane Creek phosphorus concentration has been found to be strongly seasonal, with low levels in the spring and increasing levels throughout the summer, while nitrate concentration is more weakly seasonal with a peak in spring (Kasat 2006) and another smaller peak in the late fall (USGS 2012). In 2011 the nitrogen to phosphorus ratio appeared to be highest in the early spring (USGS 2012). This is important in that there may be excess phosphorus during a time in the summer when there is limited nitrogen. This creates an opportunity for high levels of nitrogen-fixing blue-green algae to develop in the creek. The western basin of Lake Erie has historically had problems with hazardous algal blooms (HABs) of

the blue-green algae *Microcystis* (NOAA 2012). The situation in the Crane Creek estuary during the late summer could be contributing to the blooms seen in the lake both in the form of nutrients and *Microcystis* populations. Because of these nutrient trends I expected to see high concentrations of blue-green algae in the creek during the late summer, and I was further interested to see if this pattern would occur in the diked units as well as the creek.

Previous studies have found that productivity in an estuary increases with distance from the riverine source of suspended sediments (Cloern 1987). Therefore I expected to see phytoplankton production to have a negative relationship with turbidity. I also wanted to explore whether there were differences in turbidity between the creek sites and the diked units. I expected higher turbidity in the portion of the creek in this study where there is a more consistent source of suspended sediments from upstream, while in the diked wetland units more variable sources of sediment suspension are important (e.g. wind-mixing, carp rooting).

Lake water generally has lower conductivity and total dissolved solids than river water. Crane Creek experiences seiches with a periodicity of around 12 to 14 hours and can reverse the flow of the lower creek (Herdendorf 1987, Kowalski 2010). During seiche events, lake water is introduced to the estuary and water levels can rise substantially (up to 2 meters) in a few hours (Herdendorf 1987). Because of this, total dissolved solids and conductivity can be used as indicators of the influence of lake water in the creek at a particular time.

The invertebrate community of coastal marshes is important in the diet of the fish, birds, mammals, reptiles and amphibians that live in wetland habitats (Cooper et al. 2007)

and as such serve as an important pathway from primary producers and detritus to higher trophic levels (Kreiger 1992). Many factors influence the composition of the invertebrate community in any particular place. Wetland habitat that has numerous dense patches of submersed or floating vegetation will have a larger and more diverse invertebrate population compared to a site dominated by open water and limited vegetation (Kreiger 1992). The invertebrate community typically relies on the vegetation as habitat rather than a food source, and community composition tends to vary with vegetation community (Batzer and Wissinger 1996). However, others have found water levels (Gatham and Burton 2011) or sediment depth to be more predictive of invertebrate community composition (Cooper et al. 2007). Kreiger (1992) found that the invertebrate community differs depending on whether a coastal wetland experiences intermittent disconnection from the lake due to the development of a barrier beach.

Both birds and fishes are important predators on invertebrates. In the absence of fish predation the invertebrate community is more likely to include large predators (Batzer and Wissinger 1996). Therefore I anticipated some correlation between fish access and the presence of large invertebrate predators. Particularly important invertebrate taxa for waterfowl include midges, caddisfly larvae, odonate nymphs, adult and larval beetles, and water boatmen (Batzer and Wissinger 1996). Generally, invertebrates are most important for waterfowl at wintering and breeding sites, with larger waterfowl eating large invertebrates and small water fowl eating both large and small invertebrates (Batzer and Wissinger 1996). Because of this, I would expect see these taxa in the diked wetland pools that are actively managed for the attraction waterfowl.

It is my hope that this study will contribute to a better understanding of how diked wetland units differ from undiked sites in coastal estuaries of Lake Erie. I began with the preliminary hypothesis that there would be some clear differences between the chemistry and biology of river-connected estuary and the diked pool units, as well as differences among the various pools related to connectivity to the nutrients and sediment being transported by Crane Creek. I also expected to see differences related to spatial gradients within the open estuary.

Materials and Methods

Study Site

The Ottawa National Wildlife Refuge Crane Creek wetland complex is a Lake Erie coastal drowned rivermouth wetland; however, most of the wetland pools in the complex can be considered "protected" wetlands, in that they are free from the influence of the seiche and river currents (Keough et al. 1999, Figure 1). These wetlands are fed (passively in connected wetlands or by pumping in diked wetlands) by Crane Creek, a small Lake Erie tributary that flows northeast through Wood, Lucas, and Ottawa counties in Ohio. The mainstem of Crane Creek is approximately 32.2 kilometers with a catchment roughly 143.5 square kilometers (Kasat 2006). The catchment is roughly 2 percent forest, 6 percent urban and 80 percent agriculture (Kasat 2006). Overall, Crane Creek shows elevated nutrient concentrations, with higher concentrations upstream than downstream, likely due to agricultural runoff in the upper reaches and interaction with lake water in the lower reaches (Kasat 2006).

The climate is characteristic for the region, with an average temperature of -3.6˚ C in January and 23.1˚ C in July (National Weather Service Forecast Office 2012). All data was collected in 2011, which had the second warmest July on record, as well as the 10th warmest November. 2011 was also the wettest year on record, receiving 124.4 cm of precipitation, which is 37.4 cm above the annual average (National Weather Service Forecast Office 2012).

Within ONWR I sampled the length of the lower reach of Crane Creek from the State Route 2 bridge east to Lake Erie (roughly 6.5 km) and in five diked pools (2A, 2B, MS5, MS6, and MS7). I divided the creek into four sections that were visually distinct

from one another (CC1, CC2, CC3, CC4; Figure 1). The creek sample reaches range in length from roughly 2.5 km to 0.2 km, while the pools range in area from 26.3 to 103.6 hectares (Table 1). Note that these reaches of Crane Creek are not representative of the creek as whole (Kasat 2006), but are only representative of the estuarine portion of the creek.

Site CC1 is farthest from the lake (most upstream) of all of the creek sites. It runs from State Route 2 east to Stange Road. It is narrower than much of the lower creek and has patches of emergent vegetation within its channel as well as in small, connected, vegetated pools. CC2 is next farthest from the lake, and it runs from Stange Road northeast to a narrow stretch where pool 2A meets pool 2B. It is narrow in some areas but largely opens up to a very wide flat channel area. The channel is edged in many places with riprap dikes, but it does still contain large stands of emergent vegetation and small islands. CC3 is the second closest to the lake, and it runs from the end of CC2 northeast to the strait that connects Crane Creek to Lake Erie. It is very similar to CC2 in its edges and plant community, but it is much more open and lagoon-like. CC4 is adjacent to the lake and is simply the narrow, roughly 200 m strait that connects Crane Creek to the lake. It has mostly sand edges (though the sand consists mostly of tiny shards of Dreissenid mussel shells) with some areas of seawall; there is little emergent vegetation.

Pools were selected because they were adjacent to the creek and had sufficient water to navigate a kayak during the summer sampling event. All of the diked wetland pools included in this study have artificially managed water levels and are entirely isolated from Crane Creek or Lake Erie, with the exception of pool 2B which is

connected to Crane Creek by means of a water control structure (discussed further below). In the unconnected pools water is gained and lost through precipitation, evaporation, and pumping of creek water into or out of the pools. This causes the water levels in these pools to be variable throughout the year depending on management actions. Since 2011 had unusually high precipitation, only pools MS6 and MS7 required water inputs, while pool MS5 required water to be withdrawn. Pool 2A receives inputs from throughflow of an adjacent pool (not included in this study), and therefore did not receive any direct inputs or outputs of water during 2011. Pool 2B fluctuated with water levels in the creek (Huffman 2011). The management goals for the pools largely consist of maximizing various avian habitats and minimizing the colonization of various invasive plants. Pools 2A and 2B are located on public walking trails, and so these pools also have goals of attracting other wetland fauna along with the avifauna for wildlife viewing.

Pool 2A is 26.3 hectares and consists of open water and emergent vegetation, as well as areas of high amounts of submersed aquatic vegetation (SAV). This pool is largely edged in riprap, though most of it has been grown over in terrestrial and emergent vegetation. Pool 2A is adjacent to Crane Creek, south of the easternmost part of site CC2. The stated goals of the refuge for pool 2A are to "attract a variety of waterfowl, water birds, wetlands animals, and invertebrates to provide opportunities for wildlife viewing." The refuge is currently working to create and maintain a vegetation community that consists mainly of perennial plants in the hopes that this will limit the spread of invasive plant species (Huffman 2011).

Pool 2B is 38.4 hectares and consists largely of emergent vegetation and open water with large patches of SAV. This pool is mostly edged with riprap, though most of

it has been grown over in terrestrial or emergent vegetation. Pool 2B is adjacent to Crane Creek, south of the westernmost part of site CC3. It is also adjacent to pool 2A. The refuge has similar goals for 2B as those for 2A: to "attract a variety of waterfowl, shorebirds, water birds, and wetland animals for wildlife viewing." The refuge is also currently working to encourage more variety in the perennial plant community in this pool (presently it is dominated by smartweed). They are also trying to maintain a variety of depths in order to provide deep water habitat for both fish and invertebrates, as well as shallow emergent habitat for waterfowl and wading birds (Huffman 2011). In March 2011 a water control structure was opened to carry out the GLSC project (described above) that allows water and aquatic organisms (with the exception of some large fishes for several weeks in late spring due to the presence of 9.4 cm by 5.2 cm carp grates) to flow freely between Crane Creek and Pool 2B (USGS 2012). Previous to this reconnection pool 2B received water inputs passively from the same adjacent pool as 2A (Huffman 2011).

Pool MS5 is a 103.6 hectare pool that was largely open mudflat with very little standing water and deep, soft, wet sediments at the time of data collection. This pool is entirely edged in riprap with little or no vegetation. Previous to this draw down, a few large patches of emergent vegetation were established, and there were also deeper areas that still contained SAV. Pool MS5 is adjacent to Crane Creek, north of site CC2 and west of the lagoon connected to site CC3. The stated goals of the refuge for this unit are to "provide a resting and feeding area for migratory birds" and to prevent the colonization of purple loosestrife (Huffman 2011).

Pool MS6 is 28.3 hectares and has some open water and emergent vegetation, but

also has larger areas of higher ground. This pool is edged mostly in riprap, though some of it has been overgrown by emergent vegetation. Pool MS6 is adjacent to Crane Creek, north of the easternmost part of site CC1. The stated goals of the refuge for this unit are to "provide foraging and resting habitat for migratory birds as well as brood habitat" (Huffman 2011).

Pool MS7 is 38.0 hectares and consists largely of higher ground areas, including a formerly shrubby patch that still has standing dead trees. Sampling for this study occurred in the deep channel that runs along the northern edge of the pool and has large amounts of SAV. The edge of this channel is largely vegetated on the side that is adjacent to higher ground and riprap on the side that is adjacent to the dike. Pool MS7 is adjacent to Crane Creek, south of site CC2, though there are large stands of emergent vegetation between the edge of the pool and the creek channel. The stated goals of the refuge for this pool are to "provide migratory bird foraging and resting habitat" including "provid[ing] a gradient of water levels" for nesting habitat. The refuge is also working to manage water levels to prevent invasive plant colonization (Huffman 2011).

Data Collection

I collected water chemistry and primary production data as well as invertebrate samples twice during 2011. One collection in the late summer (25 August to 13 September), and one collection in the late fall (5 November to 6 November).

Primary production data was collected using a YSI-6600 V2-2 compact data sonde equipped with optical probes for recording chlorophyll (µg/L and relative fluorescence units, RFU) and phycocyanin containing blue-green algae (BGA; cells/mL and RFU) concentrations. The sonde was maintained and calibrated according to

manufacturer's recommendations (YSI Incorporated 2011). The sonde was mounted in a kayak that was navigated throughout each site at a rate as continuous as possible given conditions. A bilge pump hung over the side of the kayak just below the water surface and brought water through the flow-through sonde chamber at a continuous rate. Each minute, the sonde recorded the concentrations of chlorophyll and blue-green algae along with other water quality parameters (temperature, specific conductance, total dissolved solids, turbidity, and dissolved oxygen) throughout each navigation period. A Magellan Professional GPS unit was also present in the kayak and logged GPS points continuously during the collection periods. The data collected by the sonde were joined to the GPS points using the timestamps of the logs of the two devices in ArcMap 10, where the data was also plotted to verify the locations of all data points. A handheld recording device was used to note times during collection when problems occurred (e.g. the bilge pump became blocked with vegetation or was drawing in sediment), and these points were removed from the data set. At some sites very few data points were collected. MS6 was limited by the smaller area of open water, while CC4 was limited due both to its small size and the stronger currents in associated with being so close to the lake.

Chlorophyll and BGA concentrations were corrected according the YSI 6-Series User manual (YSI Incorporated 2011), and all concentrations discussed here are corrected values. Both chlorophyll and BGA were corrected for turbidity, and BGA was also corrected for chlorophyll concentration. Initially I determined the liner relationship between concentration values and RFU values recorded by the sonde for both chlorophyll ([chlorophyll RFU] = 0.2673*[chlorophyll µg/L] + 0.0011) and BGA ([BGA RFU] = 0.0006*[BGA cells/mL] - 0.0032). I then corrected the measured chlorophyll

concentration values for turbidity (0.03 µg/L chlorophyll per NTU) and then used these corrected chlorophyll concentrations to find corrected chlorophyll RFU values using the linear relationship determined earlier. I then used the corrected chlorophyll concentrations to correct the BGA concentrations for both turbidity and chlorophyll (77 cells/mL BGA per µg/L chlorophyll). Finally I used these corrected BGA concentrations to find corrected BGA RFU values using the linear relationship determined earlier. Some negative corrected BGA concentrations were generated due to high levels of both turbidity and chlorophyll at some sites (CC1 and MS6 particularly).

The goal of invertebrate collection was a qualitative assessment of the community composition at each site. Invertebrate samples were collected at each site by a team of two people. The team entered each site and collected invertebrates for one-hour (two total person-hours) using D-frame nets and trays. For two sites (2B and MS5) during the summer collection event the team consisted of four members who collected for thirty minutes, again totaling two person-hours. Collectors scooped along the bottom surface, along pool and channel edges, and in vegetation, and then picked through their catch in trays to collect the invertebrates after each scoop. Samples were stored in jars of 70% ethanol. In cases when it seemed that no new taxa were being collected over a long period, collection ceased even if the allotted sample time had not been reached. Effort was made to ensure that as many taxa as possible were collected and that all habitat types in the site were sampled. Efforts were also made to minimize over-collection of any one taxa; each crew member attempted to stop collection of a taxa after five individuals were collected. This is a modified version of the methods used in Wiley et al. (1988). Incidental knowledge of taxa present in particular locations (2A, 2B, CC2, and CC3) was

also included in the taxa lists used for this analysis.

The preserved organisms were identified in the lab according to Hilsenhoff (1995), Merritt et al. (2008), and Pennak (1989). As many taxa as possible were identified to genus, because identification to order or family is often insufficient to elucidate ecosystem effects (Batzer and Wissinger 1996). All insects were identified to genus (with the exception of some immature individuals as well as Ceratopogonidae and Chironomidae), decapods were identified to genus when possible (otherwise family), gastropods and bivalves were identified to family (with the exception of Dreissenid mussels which were identified to genus), annelids were identified to class, and springtails and water mites were identified to Collembola and Hydrachnida, respectively.

Data Analysis

Patterns in primary production and other abiotic factors were initially explored using Data Desk © version 6.1 to visually compare the mean values for the parameters collected by the sonde. Temperature and dissolved oxygen data was not used in analysis due to the varying times of day at which data was collected. All analyses of primary production and abiotic factors were carried out separately for the summer and fall collection events because the goal of the analysis was to determine differences between the sites, which may be obscured by combining data from different seasons.

Two multivariate analyses of variance (MANOVA) were carried out using IBM© SPSS© Statistics version 20, one for each sampling event, using five parameters (conductance, total dissolved solids (TDS), turbidity, chlorophyll concentration, and BGA concentration) to compare differences between sites.

Analyses of variance (ANOVA) were carried out using IBM© SPSS© Statistics

version 20, for each sampling for each of five parameters (conductance, TDS, turbidity, chlorophyll concentration, and BGA concentration). Scheffe post hoc tests were performed to determine the differences between each site.

Principal components analyses (PCAs) were carried out with IBM© SPSS© version 20 using the primary production and abiotic parameters. One analysis used only the summer dataset and one used only the fall dataset. Each analysis used the mean values at each site for specific conductance, TDS, turbidity, chlorophyll concentration, and BGA concentration.

Invertebrate taxa characteristics were analyzed using a database compiled by Wiley's lab at the University of Michigan School of Natural Resources and Environment. This database compiled information on tolerance/sensitivity, feeding guilds, and various other aspects of classification for invertebrates. Because the invertebrates in this study were collected qualitatively and not quantitatively, all data was analyzed as the presence or absence of taxa rather than the number of individuals from each taxa collected. Taxa lists were generated by combining the data collected from both sampling events.

Presence-absence data was used to calculate Sørensen's similarity quotients for each pair of sites. This is calculated by dividing the number of taxa found at both sites by the sum of the total number of taxa found at each site. Perfectly similar sites generate a similarity quotient of 1 while perfectly dissimilar sites generate a similarity quotient of 0.

Additionally, a PCA was carried out using the invertebrate taxa lists from each site, each taxa being a variable in the analysis. This analysis required the removal all taxa that were present at all sites (*Caenis*, Chironomidae, Hirudinea, *Hyalella*, *Ischnura*, Lymnaeidae, *Notonecta*, Physidae, and *Trichocorixa*) because these variables had zero

variance. This resulted in the use of 84 taxa for this analysis.

Results

Primary Production and Abiotic Factors

Mean chlorophyll and BGA concentrations were generally higher in the summer than in the fall. Chlorophyll concentration in the summer ranged from 1.28 RFU in CC4 to 19.84 RFU in MS6 (nominally from 4.78 to 74.19 µg/L chlorophyll), but in the fall from 0.80 RFU in 2A to 8.46 RFU in MS5 (Table 2). Mean BGA concentration in the summer ranged from -0.20 RFU in MS6 to 9.91 RFU in MS5, and in the fall ranged from 0.16 RFU in CC1 to 1.53 RFU in MS5 (nominally from 275 to 2552 cells/mL). Turbidity was also generally higher in the summer, ranging from 38.50 NTU in 2B to 141.95 NTU in MS5. In the fall, turbidity ranged from 10.20 NTU in MS6 to 67.60 NTU in CC4.

Average summer and fall chlorophyll concentrations decreased slightly in the creek channel as it approached the lake (Figure 2). Average summer chlorophyll concentration in pools 2A and 2B were similar to CC2, while the remaining pools MS5, MS6, and MS7 had higher average chlorophyll concentrations. In the fall average chlorophyll concentrations were uniformly low across all sites, with the exception of MS5 which had similar values for summer and fall.

In the summer sampling event, there was a sharp increase in average BGA concentration from upstream to downstream in the sites in the creek (Figure 3) which is consistent with the fact that a major algal bloom occurred in the western basin of the lake during the sampling event (NOAA 2011). Similarly, average TDS in the sites in the channel decreased from upstream to downstream, further supporting the idea of an influx of lake water at this time. However, BGA concentrations in the pools were also elevated during the summer sampling. Average BGA concentration in pools 2A and 2B in the

summer were similar to those found in CC2, while MS5 and MS7 had average BGA concentrations much higher than even CC4 (which is closest to the lake) in the summer sampling (MS5 had an average BGA concentration of over 16500 cells/mL). MS6 had the lowest average BGA concentration during the summer. During the fall all sites had similarly low average BGA concentrations, the highest being found in MS5.

MANOVAs for each sampling period using all of the primary production and water quality parameters found there were significant differences among the study sites ($p < 0.001$; F-statistics for summer: Pillai's Trace F = 54.843, Wilks' Lambda F = 101.021, Hotelling's Trace F = 168.129, Roy's Largest Root F = 613.760; F-statistics for fall: Pillai's Trace F = 27.473, Wilks' Lambda F = 88.342, Hotelling's Trace F = 201.699, Roy's Largest Root F = 713.139). ANOVAs for individual water quality and phytoplankton metrics also found significant differences between sites for all variables in both sampling events ($p < 0.001$; Table 3 and Table 4).

However, post hoc contrasts showed that the differences were not necessarily consistent in time or space. For example creek sites CC2, CC3, and CC4 were not significantly different from one another in either specific conductance or TDS in either summer or fall (Figure 4 and Figure 5). Likewise, for these parameters the pools were relatively similar to one another with the exception of pool MS5 in the summer. The pool sites generally had higher values for these two parameters than the creek sites (with the exception of CC1, the most upstream site).

During both sampling events turbidity was fairly similar between sites, however there were some differences (Figure 6). Pool MS5 had significantly higher turbidity than all other sites during the summer sampling. The rest of the sites showed a high degree of

similarity during the summer. In the fall there was also a large amount of statistical similarity in terms of turbidity with the lower creek and MS5 having significantly higher turbidities.

Chlorophyll concentration in MS5 was significantly different from all other sites for both summer and fall sampling (Figure 7). In the fall MS5 was higher than all other sites, while in the summer it was higher than all sites except MS6 and MS7. Pools MS6 and MS7 had significantly higher chlorophyll concentrations than other sites for summer sampling. Chlorophyll was low at all sites in the fall.

There were significant differences in blue-green algae concentrations longitudinally within the creek, with the lower creek sites (CC3 and CC4) being higher than the upper sites (CC1 and CC2; Figure 8). The two upper creek sites (CC1 and CC2) were significantly different from one another, but were similar to many pool sites (2A, 2B, and MS6). However MS5 and MS7 had higher BGA levels than the other pools, with MS7 being statistically similar to CC4. In the fall most sites had uniformly low BGA concentrations, but again MS5 was significantly higher than the other units, as was the case for chlorophyll concentration.

In a PCA of primary production and water quality parameters using the summer data, the first two components explained 45.5 and 33.2 percent of the variation in my dataset (Tables 5 and 6). When sites were plotted on these components it shows that the three lower creek sites (CC2, CC3, and CC4) were very similar, while the upper creek site (CC1) was chemically distinct (Figure 9). MS5 appears to stand out from all other sites, while the remaining pool sites fall somewhere in between the lower creek and CC1.

In a PCA of primary production and water quality parameters using the fall data,

the first two components explained 52.0 and 43.3 percent of the variation (Tables 7 and 8). When these components were plotted, the pattern was similar to the patterns in the summer with a clear separation between the lower creek sites (CC2, CC3, and CC4) and the upper creek site (CC1; Figure 10), MS5 was isolated, and the remaining pools fall between the lower creek sites and CC1, but they do appear to be more similar to CC1, the most upstream segment (especially pool MS6).

Invertebrate Community

A total of 93 taxa were collected at ONWR in the two sampling events or observed incidentally (Table 9). Pool 2B, the experimentally reconnected unit, was the most diverse with 57 taxa collected; 39 taxa were found in 2A, 21 in MS5, 33 in MS6, and 46 in MS7. The Crane Creek sites showed a similar range of diversity as the diked units. CC1, the most upstream site was most diverse with 47 taxa. Diversity declined downstream with only 26 taxa found at the confluence with Lake Erie. More taxa were collected in the summer than in the fall at all sites (Table 10). Most of the taxa collected are known to be fairly pollution tolerant, with average tolerance indices for the taxa collected at each site ranging from 6.93 in CC3 to 7.40 in MS5 (Table 11). Only a few relatively sensitive taxa were collected, including *Acentria* (2A, CC1, CC2), *Dineutus* (CC1, CC2, CC3, CC4), *Gyrinus* (2B, MS7, CC1, CC3, CC4), *Epiaeschna* (2A, CC1), *Orconectes* (2B, CC2), *Palaemonetes* (all except CC4) *Stactobiella* (CC3), and Trichopteran larva (2A, CC2, CC4). At every site more than 50 percent of the taxa collected were surface air breathers and therefore not dependent on dissolved oxygen concentrations (Table 12). Metabolic conforming taxa which require high oxygen or high flow velocities made up less than 20 percent of the taxa collected. Similarly,

Ephermeroptera-Plecoptera-Trichoptera (EPT) taxa represented less than 20 percent of the taxa at each site. Isopods, leeches and Gastropods represented less than 30 percent of all taxa at each site; although Hirudinea were not identified past Order and gastropods were not identified past family, while most insect taxa were identified to genus. Predators were the dominant feeding guild at every site (Figure 11).

Two amphipod taxa were collected: *Hyalella* and *Gammarus*. *Hyalella* was collected at every site, while *Gammarus* (a typically riverine group) was found only in the creek sites (CC1, CC2, CC3, CC4) and the river-connected pool 2B. Isopods were only found in sites CC1 and CC2. *Caecidotea* was found at both of these sites while *Lirceus* was only found at CC1. Decapods were found at all sites except CC4, with *Palaemonetes* being found at all other sites and *Orconectes* at only 2B and CC2. Leeches were present at all sites, and Oligochaetes were found at all sites except 2B and MS6 but were almost certainly present everywhere.

Dreissena mussels were found only in the creek sites (CC1, CC2, CC3, and CC4) and in pool 2B. Lymnaeid and Physid snails were found at all sites, while Planorbid snails were found at all sites except CC4 and MS5. Limpets were found in CC1 and CC2 while Viviparids were found only in CC2 and CC3.

Hemipterans were present and common at all sites. *Notonecta* and *Trichocorixa* were found at all sites. *Belostoma* was found at all sites except MS5. *Ranatra* was found at all sites except CC3 and 2B. *Rheumatobates* was found in only the creek sites (CC1, CC2, CC3, and CC4), while *Metrobates* was only found in pool sites (2B, MS5, MS7). Corixids were common at all sites with three genera found at many sites (*Trichocorixa* was found at all sites, *Hesperocorixa* was found at all sites except CC2 and 2A, and

Palmacorixa was found at all sites except MS5).

Beetles were present at all sites, and they were abundant at many sites, especially pool 2B. *Peltodytes* was very common, being found at all sites except MS5. *Tropisternus* was also common being found at all sites except CC4, 2A, and MS7. *Dineutus* was found only in the creek sites, while *Gyrinus* was found in the creek but also in pools 2B and MS7. Dytiscid beetles were absent from creek sites (only found in pools 2B, MS6 and MS7).

Dipterans were present at all sites, with Chironomidae present at every site. *Anopheles* was also found in nearly all sites, only absent from CC1, CC4, and MS5.

Odonates were common, with *Ischnura* present at all sites. CC1, 2A and 2B seemed to have the greatest diversity of Odonates. *Anax*, *Erythemis*, *Lestes*, and *Pachydiplax* were all found at four or more sites. Two families (Corduliidae and Gomphidae) were rare and found in only one site (CC1).

Only three genera of mayfly were found: *Caenis*, *Hexagenia*, and *Paracloeodes*. *Caenis* and *Paracloeodes* were present at nearly all sites, while *Hexagenia* was found only in 2A, CC2, and CC3. Trichopterans were only found only in the creek (CC1, CC2, CC3, and CC4) and in pools 2A and 2B. No stoneflies were collected at any site.

Nine of the taxa collected were found at every site: *Caenis*, Chironomidae, Hirudinea, *Hyalella*, *Ischnura*, Lymnaeidae, *Notonecta*, Physidae, and *Trichocorixa*.

Sørensen similarity quotients calculated to compare each site to all other sites indicate that most sites share between fifty and sixty percent of taxa with other sites (Table 13). The creek sites (CC1, CC2, CC3, and CC4) all shared greater than sixty percent of taxa with one another while the diked units were more variable in composition

than were the Crane Creek sites. Overall, the average similarity was 0.631. Pools 2B and MS5 were the most unique compared to the other sites.

A PCA with invertebrate taxa also indicated a high degree of faunal variation among the units (Table 14). The first three components explained 26.3, 17.2, and 13.5 percent of the variation. In plots of component space (Figure 12) it can be seen that both 2B and CC1 stand out from all other sites. The remaining sites fall into two groups: a tight cluster of CC2, CC3, CC4, and MS5 and a somewhat looser cluster of 2A, MS6, and MS7.

Discussion

The goal of this study was to assess ecological similarities and differences between various habitats in a Lake Erie coastal and diked wetland complex and to document the extent of their seasonal and spatial variability in terms of basic chemistry, phytoplankton density, and the invertebrate community. ONWR was a fitting site for this study because of the current GLSC hydrologic reconnection project and the potential for future reconnection projects for other diked wetland pools in this complex and nearby lake marshes. Because this site is similar to other diked wetland complexes on the southwestern shore of Lake Erie (Herdendorf 1987), the results of this study may be applicable to other wetlands in this region.

Generally the range of phytoplankton densities observed were consistent with other estuaries in the region (Bridgeman et al. 2012). The parts of the creek closest to Lake Erie (CC2, CC3, and CC4) were generally more similar to one another than they were to other sites explored in this study. This is likely due to the heavy influence of inflowing lake water on these areas due to Lake Erie's regular and large seiches, which can have amplitudes from 0.7 m to 2 m over a 12 to 14 hour period (Herdendorf 1987). Previous studies found that water movement in the lower creek alternates between flowing into the lake and out of the lake, with velocities similar in both directions, approaching 1 m per second at the mouth of Crane Creek (Kowalski 2010). Though seiches stall creek flow and raise water levels higher up the creek as well, it appears that the upper reaches of the creek (e.g. CC1) are more heavily influenced by agricultural runoff from Crane Creek (Kasat 2006) and receive little actual dilution from Lake Erie seiches. Further, the high standard deviations of TDS and specific conductance recorded

for this site seem to indicate that it is a transitional area between the agricultural runoff dominated upstream and the lake water dominated downstream. Nutrient data from previous studies have shown that Crane Creek at Stange Road (meeting point of CC1 and CC2) has higher SRP concentrations than those downstream, though this pattern was not seen in the nitrate or ammonia concentrations (Kasat 2006). However, the pattern seen in SRP concentration should be explored further to determine its relationship with the patterns in primary production in this study.

In terms of water quality and primary production parameters, the upper portion of the creek (CC1) seems to have more in common with a number of the pools than it does with the lower creek sites. Creek site CC1 was most similar to pools MS6 and MS7. This could be due to the similar bathymetric structure at these sites (MS6 and MS7 both had deeper open water zones areas flanked with littoral vegetation, similar to the channel in CC1).

Of the remaining pools, MS5 was fairly different from all other sites in terms of abiotic and primary production parameters. This could be due to its large size or the unique management actions applied to this pool (e.g. water was removed from this pool; Huffman 2011).

The adjacent GLSC study pools 2A and 2B were similar to one another in terms of abiotic factors and levels of primary production. This could be due to the similarity of vegetative community and structure, as well as similarity in previous management history (e.g. previous to reconnection pools 2A and 2B both received water inputs passively via throughflow from the same adjacent pool). In previous studies these pools have been used as a control-reference pair due to this similarity (USGS 2012), however since the

reconnection of pool 2B in 2011 with Crane Creek, it is not surprising that they are not currently identical. It is also unsurprising that 2B was also similar to the lower creek sites in terms of phytoplankton density (especially site CC2), because the water intake structure for 2B is located at the boundary between CC2 and CC3. The GLSC project found that after the structure opened nutrients in 2B began to emulate those found in Crane Creek. This was most notable in the dramatic increase in nitrate + nitrite found in 2B after the structure opened (USGS 2012). The GLSC study also found an increase in orthophosphate, which is noteworthy because previous studies (Kasat 2006) found orthophosphate levels between 2A and 2B to be similar. The GLSC nutrient data is further evidence of the changes occurring in 2B that are shifting its water chemistry to appear less like the other pools and more like the creek.

All sites had very similar turbidity values, which made comparison of phytoplankton density based on this parameter uninformative. I expected there to be differences in turbidity between the pools and the creek, however there were no strong (i.e. statistically significant) differences between them. This could be due to the presence of greater and more consistent sources of turbidity in the pools than I had anticipated. Further research into the sources of turbidity and mixing in the pools may help to clarify this pattern.

The various diked pools and creek units at ONWR showed a great deal of variability in the makeup of the invertebrate community that they host. Overall, the taxa found are consistent with those found at other Lake Erie coastal marshes (Kulesza et al. 2008) as well as elsewhere in the Great Lakes region (Provence 2008). The low number of mayfly and caddisfly taxa found in this study is noteworthy because these groups tend

be sensitive to pollution and are usually not well represented in polluted areas, especially mayflies (MDEQ 1997). Likewise, the absence of stonefly taxa may indicate poor water quality since this group is very sensitive to water quality (MDEQ 1997). However these taxa are also often absent from low-gradient, warm water systems such as lower Crane Creek, regardless of water quality. High percent of individuals in the order Isopoda, class Gastropoda, or class Hirudinea are typical of degraded streams (MDEQ 1997), but often dominant in wetland settings. Similarly, surface dependent taxa are indicators of habitats in which oxygen stress is a persistent issue, and these taxa were very well represented at all sites (MDEQ 1997).

In my surveys the Crane Creek sites were all fairly similar to one another in terms of taxa, with higher similarity quotients and tight clustering in the invertebrate taxa PCA (especially the lower creek, sites CC2, CC3, and CC4). This is reasonable, since these sites are all part of the same channel system. All of the lower creek sites were fairly dissimilar to diked pools in terms of invertebrate community composition, including pool 2B. This is perhaps more surprising since pool 2B is connected to the creek where CC2 meets CC3. However, 2B is a wetland marsh unit and has a great deal of dense vegetation, as well as a greater variety of habitat areas and much less riprap edging compared to the creek. Also, pool 2B was more frequently visited, which may have contributed to the higher number of incidental observations of taxa there.

It has been found that variable water levels can increase the abundance and diversity of aquatic insects (Batzer and Wissinger 1996) and that water level can exert more influence than the vegetation type on the invertebrate community (Gatham and Burton 2011). Pool 2B had the highest numbers of invertebrate taxa collected which can

perhaps be explained by the naturally variable water levels that it experiences due to its reconnection with seiche driven lower Crane Creek. This is an interesting potential benefit of hydrologic reconnection at Crane Creek and deserves further exploration and monitoring.

Pool MS5 appeared to be unique in terms of most parameters explored. This is not surprising in that sampling occurred while this pool was drawn down, which left little water and very deep silty sediment. On the other hand it is unclear how different these results might have been if it had not been drawn down, since this pool has a distinct physical structure with very little cover and extensive bare riprap on all sides.

I expected that there would be clear differences between the diked pool sites and the creek sites in terms of invertebrate taxa richness since previous studies had found that taxa richness was significantly greater in diked than in undiked marshes (Provence 2008). However the data in this study does not support this generalization since some pools showed greater taxa richness and others showed less. A possible explanation for the observed variability in taxa richness could be the variability in the number and kinds of large fish in the sites, since it has been found that there is greater insect diversity in fishless ponds than in ponds with thriving fish populations (Batzer and Wissinger 1996). It has also been found that in fishless ponds, large invertebrates can take over the role as top predators and limit prey production (Batzer and Wissinger 1996). Though much is known about the fish population in pools 2A and 2B as well as in the creek (especially CC2, CC3, and CC4), comparatively little is known of the size and type of fish found in the other sites (USGS 2012). A better understanding of the fish populations in CC1, MS5, MS6, and MS7 may help to explain the patterns of variation in taxa richness and feeding

guilds that were found in this study.

Further, differences in the invertebrate community between sites could be due to differences in primary carbon source for the site, which in the pools can change as they age and as emergent vegetation spreads (Friday 1987). There is likely more autochthonous production in the pools than in the creek, both in the water column with high phytoplankton densities and in the form of submersed and emergent macrophytes. Crane Creek (especially CC2, CC3, and CC4) had much less submersed aquatic vegetation and lower phytoplankton densities, so may have received much of its carbon input from allochthonous sources. Further research into the primary carbon pathways in this system may help to explain the relative presence of invertebrate feeding guilds found at each site (Vannote et al. 1980).

Finally, invertebrate community differences among sites could be due to differences in the habitat available at each site. Relative amounts of open water and standing vegetation as well as vegetation community composition can have an impact on the invertebrate community (Batzer and Wissinger 1996), and detailed analysis of the makeup of the vegetation communities and seasonal changes in these communities at each site may help to explain some patterns seen here.

Overall there was a lot of variability in all the parameters measured from site to site in this study, however some patterns emerged. The lower creek sites (CC2, CC3, and CC4) were strongly similar to one another in all parameters, while CC1 was relatively distinct from the other creek sites. The pool sites showed much more variability than the creek sites, which is not surprising due to their relatively high degree of hydrologic isolation and the variety of goals set for the pools by the managers at ONWR. Despite

this, the pool sites are still more often similar to one another than they are to much of the creek. Pool MS5 seemed quite unique ecologically compared to the other pools, and further research could shed light on whether it would appear more like the other pools if water levels were kept higher or whether is indeed unique for reasons other than water level (e.g. larger size).

Management implications

Pool 2B has been experimentally reconnected with Crane Creek by a roughly 3 m water control structure since spring 2011 and so water flows into it and out of it with rising and falling levels of Crane Creek. In most respects it is still more pool-like (e.g. many abiotic parameters, observed standing vegetation); however differences between it and the other un-connected units seem to be emerging despite the high variability of these units. Longer term monitoring of trends in the similarity of this pool to other units will provide a useful evaluation of the results of reconnection. It now seems likely that pool 2A will also be reconnected to the creek in the near future (Kowalski 2012) which will provide an opportunity to see if the changes that have occurred in 2B over the past year are characteristic of this kind of connection.

Diet information for waterfowl and other avifauna that are of most interest at ONWR may be useful in interpreting the biological results of this study. With information about taxa preferences, the invertebrate taxa list developed in this study could help elucidate which units are most productive in terms of supporting invertivorous birds. This might be especially important in those pools where brood or nesting habitat is a primary goal (MS6 and MS7), since it is during nesting that invertebrates are more important in the diet of avifauna (Batzer and Wissinger 1996). Because fishless ponds

tend to have larger predatory insect taxa, it may be beneficial to limit fish production in pools where management goals are more directed at the avian community than the fish community in order to provide large invertebrate prey items for waterfowl (Batzer and Wissinger 1996).

Dreissenid mussels are another invasive species that Great Lakes managers are concerned about. These mussels were found throughout the length of the creek as well as in the connected pool 2B. Zebra mussel survivorship has been found to be better at depths greater than 18 cm because veliger colonization increases with depth (Bowers and Szalay 2005). Note that 2B was recently reconnected and was the only pool site with zebra mussels present. Periodically drawing down hydrologically reconnected units might be necessary to limit zebra mussel colonization. Further research should be conducted to determine the likelihood of Dreissenid mussel invasion causing serious ecological problems in the pools, as well as to determine whether quagga mussels have similar invasion potential as zebra mussels.

Conclusion

The goal of this study was to explore the ecological variability in a fragmented Lake Erie coastal and diked wetland system. This was achieved by exploring water quality, levels of primary production through measuring pigment concentrations, and invertebrate community composition in nine distinct wetland units (four sites in the estuary and five in managed diked wetland pools) at the Ottawa National Wildlife Refuge. Overall, it was found that those creek sites close to the lake were largely similar to one another, while the creek site farthest upstream from the lake was different from this group. It is likely that this is due to the strong influence that mixing with lake water has on the lower parts of the creek, an effect that diminishes upstream. The diked pool sites were more ecologically variable than the creek sites, with some being quite similar to and others being quite different from each other. It is likely that much of the variation in the pool sites was due both to variable management actions and differences in habitat structures.

This study arose largely due to the GLSC project currently taking place at ONWR and a need for a better understanding of how other pools may be different from those currently under investigation. Because of this, it is reasonable to discuss the implications of my findings on the GLSC project. The GLSC project to reconnect pool 2B initially used pool 2A as a reference site for invertebrate, avian, fish and vegetation communities as well as for water quality. Previous studies have indeed shown that these two pools are quite similar ecologically (Kasat 2006). Likewise, I found these two sites to be more similar to one another than the other sites in this study for most parameters, and so using them as a matched treatment and control set is appropriate for the GLSC study. However,

I have also shown that there is a good deal of ecological variability among the diked units at ONWR, so generalization of the results of the GLSC study beyond pools 2A and 2B should be done cautiously. The diked units at ONWR have all shown unique characteristics and should not necessarily be treated as replicates for one another. These unique characteristics make it difficult to apply predictions based on findings from one unit to another unit and argue for continued monitoring and evaluation studies in future re-connection projects. Similarly, I found that there is longitudinal variation present in even the relatively short portion of estuary investigated in this study, which will have implications for reconnection projects depending on where the connection with the estuary is made. A way in which these problems may be addressed is through longer term monitoring of these parameters across a wider array of ponds in order to create a bigger picture through time for each unit and its unique trajectory.

This study has highlighted the variability seen in a group of the diked pools at ONWR and offered some possible explanations for it. There are ten other managed wetland units at the refuge in addition to the five that were described in this study. At least seven of these pools are adjacent to the creek and could potentially be reconnected in the future. Further exploration of the physical conditions and faunal communities of these unexplored pools may help to reveal whether connections to the creek or other connected pools would be beneficial to management goals. I have also argued here that by reconnecting these pools the refuge might maximize production of the invertebrate taxa that are most beneficial to the avifauna as well as potentially allow other animals (especially fishes) access to more varied habitats in the numerous different pool types on the refuge. On the other hand, I have shown that hydrologic reconnection has

implications for the spread of invasive species and so action should be taken with caution.

This study would have benefited from more sampling dates for collecting phytoplankton, abiotic, and invertebrate data and further taxa list development. Concurrent nutrient data for all of the sites would also have been helpful in explaining the trends in phytoplankton density. Larger scale invertebrate collection for quantitative, rather than simply qualitative, invertebrate data could also have been useful in shedding light on the diversity and evenness of invertebrate taxa present in the refuge; however the large size of the pools may make this sort of collection unrealistic. A more detailed description of vegetative cover in each site would also have been extremely beneficial in determining differences in habitat structure.

Figures and Tables

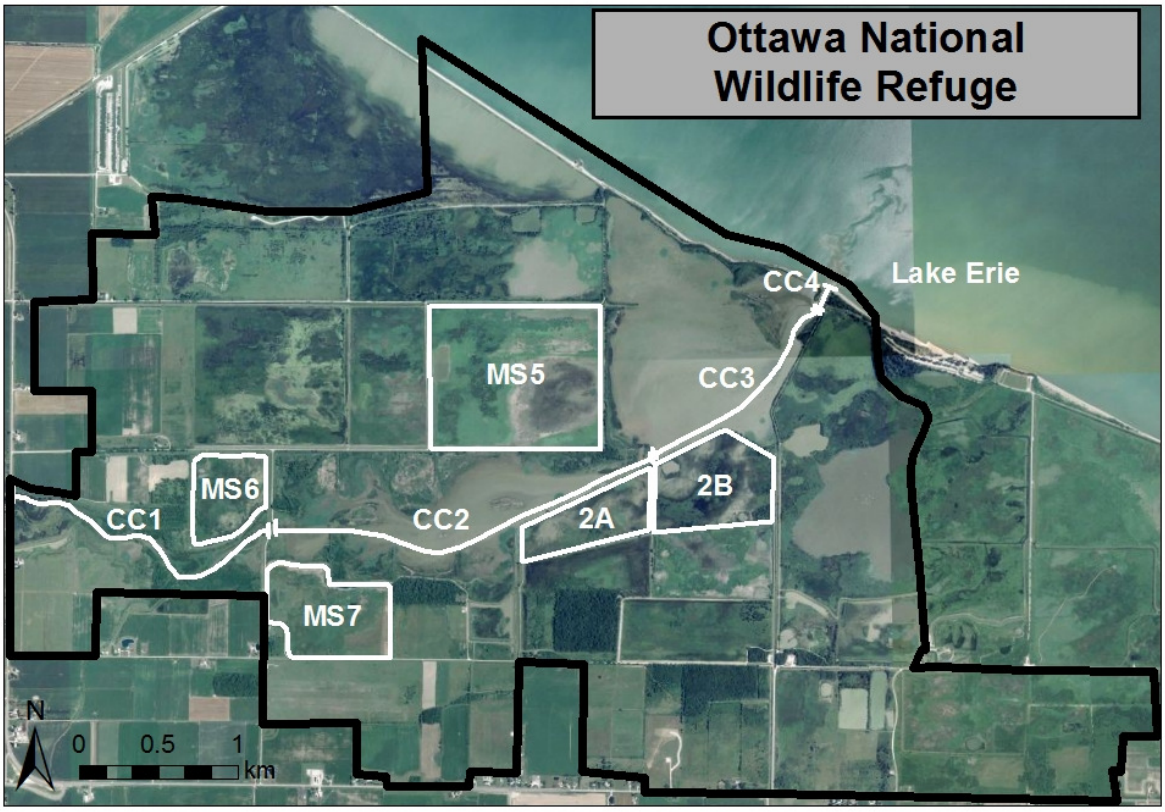

Figure 1. Outline of Ottawa National Wildlife Refuge (in black) with channel and pool sites outlined and labeled (in white). Bing Maps Aerial © 2010 Microsoft Corporation and its data suppliers.

Table 1. Description of sites (area from Huffman (2011) and lengths are rough measures using ArcMap 10 ©)

Site	Surface area (hectares)
2A	26.3
2B	38.4
MS5	103.6
MS6	28.3
MS7	38.0

Site	Length (km)
CC1	2.0
CC2	2.5
CC3	1.5
CC4	0.2

Table 2. Summary of all abiotic parameters collected during sampling events.

Site	Sampling Event	n	Specific Conductance (uS/cm)		Total Dissolved Solids (g/L)		Turbidity (NTU)		Chlorophyll concentration (RFU)		Blue-green Algae concentration (RFU)	
			Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
2A	Summer	24	279.917	14.120	0.182	0.009	49.300	27.784	2.590	1.729	2.511	3.926
2B	Summer	43	236.311	150.480	0.150	0.104	38.505	21.312	2.216	1.850	2.383	1.230
MS5	Summer	41	391.585	3.834	0.255	0.003	141.954	22.955	9.574	1.467	9.908	3.258
MS6	Summer	8	272.750	8.031	0.177	0.005	43.150	11.938	19.840	14.812	-0.204	0.978
MS7	Summer	17	298.882	25.448	0.194	0.016	48.606	30.268	14.841	13.444	7.772	5.402
CC1	Summer	53	613.981	56.010	0.399	0.036	52.821	19.783	5.805	3.082	0.742	0.728
CC2	Summer	42	0.354	0.070	-0.014	0.003	74.876	5.837	2.997	1.657	2.691	0.713
CC3	Summer	54	0.280	0.004	-0.014	0.003	65.856	9.366	1.345	0.159	4.835	1.269
CC4	Summer	11	0.274	0.001	-0.012	0.002	51.236	3.166	1.279	0.121	5.833	0.658
2A	Fall	25	274.360	5.816	0.178	0.004	15.920	5.312	0.797	0.566	0.451	0.128
2B	Fall	25	379.360	18.007	0.247	0.012	29.628	23.292	1.051	0.435	0.481	0.221
MS5	Fall	21	413.667	18.637	0.269	0.012	61.943	22.889	8.459	1.442	1.530	0.498
MS6	Fall	8	446.250	7.815	0.290	0.005	10.200	1.902	1.952	0.498	0.486	0.067
MS7	Fall	13	322.385	11.709	0.210	0.008	12.677	7.248	1.126	0.472	0.520	0.097
CC1	Fall	33	576.514	331.038	0.399	0.172	18.724	10.149	1.857	2.019	0.164	0.236
CC2	Fall	43	0.650	0.053	0.090	0.008	53.053	33.586	1.140	0.356	0.572	0.260
CC3	Fall	19	0.454	0.030	0.071	0.001	56.763	6.076	0.881	0.191	0.565	0.150
CC4	Fall	3	0.478	0.007	0.072	0.001	67.600	3.470	1.474	0.205	0.596	0.081

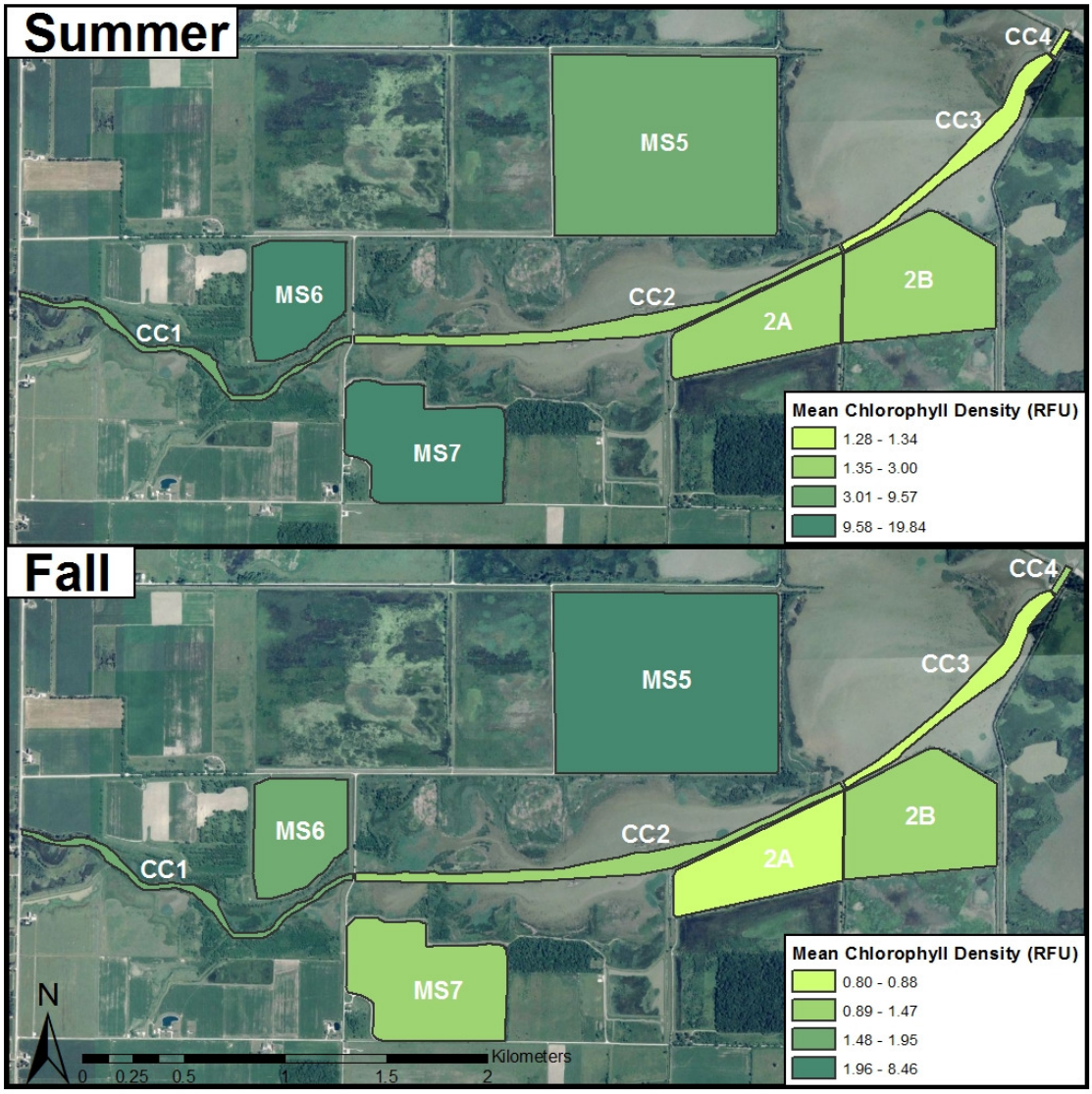

Figure 2. Mean values of chlorophyll concentration at each site for each sampling event. Note the different scales for each of the two season; categories were determined by the Jenks natural breaks method in ArcMap 10 ©.

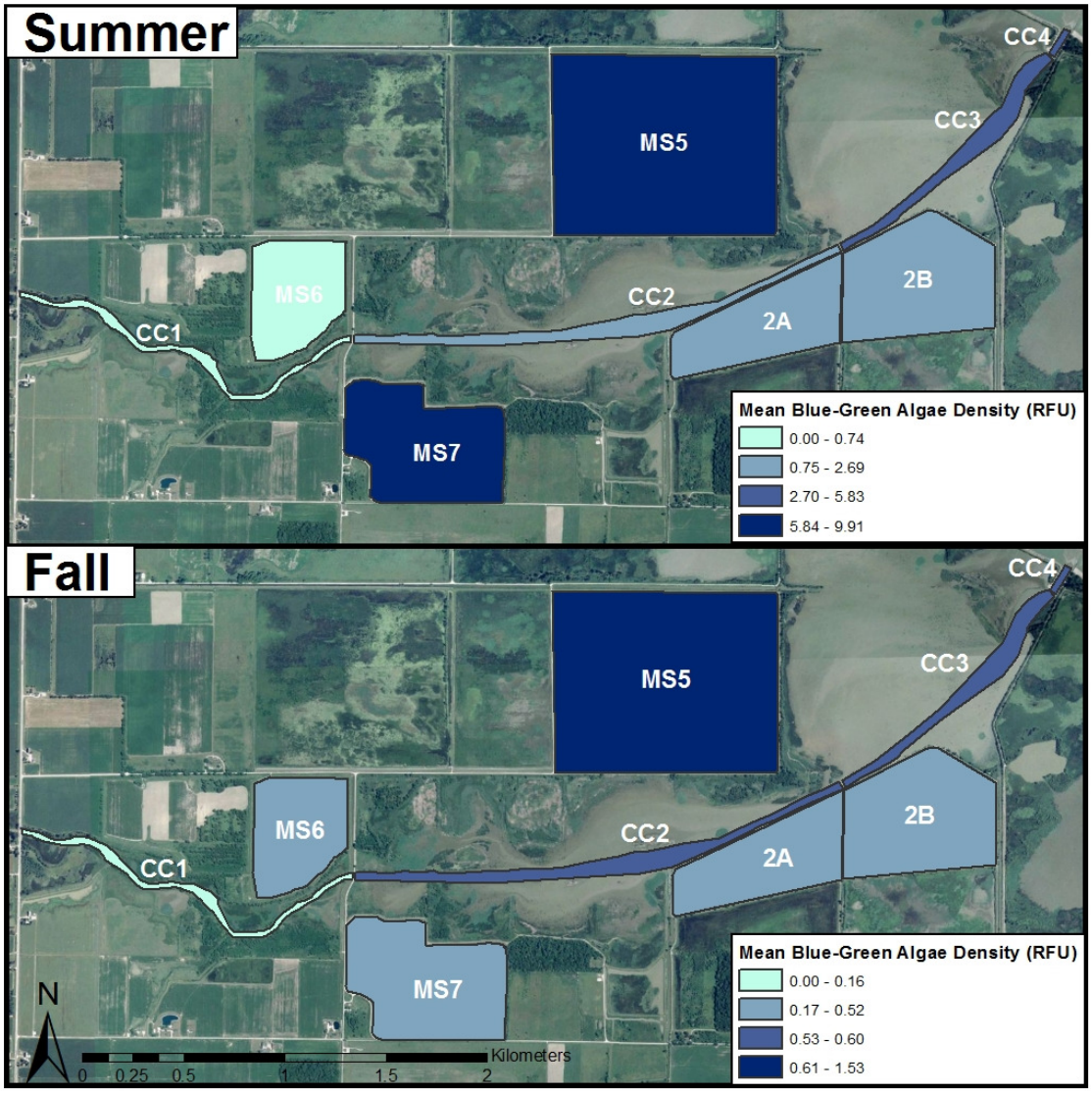

Figure 3. Mean values of blue-green algae concentration at each site for each sampling event. Note the different scales for each of the two seasons; categories were determined by the Jenks natural breaks method in ArcMap 10 ©.

Table 3. ANOVA results table for between site effects of the summer dataset. Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	SpCond_uS_per_cm	14587967.072[a]	8	1823495.884	458.223	.000
	TDS_g_per_L	6.655[b]	8	.832	442.323	.000
	Turbidity_NTU	298904.957[c]	8	37363.120	106.045	.000
	Chl_RFU	5814.326[d]	8	726.791	38.694	.000
	BGA_RFU	2689.116[e]	8	336.140	65.636	.000
Intercept	SpCond_uS_per_cm	10314714.587	1	10314714.587	2591.968	.000
	TDS_g_per_L	4.077	1	4.077	2167.881	.000
	Turbidity_NTU	754158.034	1	754158.034	2140.478	.000
	Chl_RFU	8603.545	1	8603.545	458.047	.000
	BGA_RFU	3127.842	1	3127.842	610.753	.000
Site	SpCond_uS_per_cm	14587967.072	8	1823495.884	458.223	.000
	TDS_g_per_L	6.655	8	.832	442.323	.000
	Turbidity_NTU	298904.957	8	37363.120	106.045	.000
	Chl_RFU	5814.326	8	726.791	38.694	.000
	BGA_RFU	2689.116	8	336.140	65.636	.000
Error	SpCond_uS_per_cm	1130175.785	284	3979.492		
	TDS_g_per_L	.534	284	.002		
	Turbidity_NTU	100062.193	284	352.332		
	Chl_RFU	5334.403	284	18.783		
	BGA_RFU	1454.447	284	5.121		
Total	SpCond_uS_per_cm	33792140.520	293			
	TDS_g_per_L	14.300	293			
	Turbidity_NTU	1749803.700	293			
	Chl_RFU	18636.958	293			
	BGA_RFU	8872.050	293			
Corrected Total	SpCond_uS_per_cm	15718142.857	292			
	TDS_g_per_L	7.189	292			
	Turbidity_NTU	398967.150	292			
	Chl_RFU	11148.728	292			
	BGA_RFU	4143.563	292			

a. R Squared = .928 (Adjusted R Squared = .926)
b. R Squared = .926 (Adjusted R Squared = .924)
c. R Squared = .749 (Adjusted R Squared = .742)
d. R Squared = .522 (Adjusted R Squared = .508)
e. R Squared = .649 (Adjusted R Squared = .639)

Table 4. ANOVA result table for between site effects of the fall dataset. Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	SpCond_uS_per_cm	8833855.705[a]	8	1104231.963	56.710	.000
	TDS_g_per_L	2.409[b]	8	.301	57.079	.000
	Turbidity_NTU	71162.875[c]	8	8895.359	21.131	.000
	Chl_RFU	1001.590[d]	8	125.199	116.363	.000
	BGA_RFU	25.445[e]	8	3.181	49.154	.000
Intercept	SpCond_uS_per_cm	7577978.003	1	7577978.003	389.181	.000
	TDS_g_per_L	4.329	1	4.329	820.576	.000
	Turbidity_NTU	138619.998	1	138619.998	329.294	.000
	Chl_RFU	456.467	1	456.467	424.251	.000
	BGA_RFU	37.427	1	37.427	578.407	.000
Site	SpCond_uS_per_cm	8833855.705	8	1104231.963	56.710	.000
	TDS_g_per_L	2.409	8	.301	57.079	.000
	Turbidity_NTU	71162.875	8	8895.359	21.131	.000
	Chl_RFU	1001.590	8	125.199	116.363	.000
	BGA_RFU	25.445	8	3.181	49.154	.000
Error	SpCond_uS_per_cm	3524363.942	181	19471.624		
	TDS_g_per_L	.955	181	.005		
	Turbidity_NTU	76193.997	181	420.961		
	Chl_RFU	194.744	181	1.076		
	BGA_RFU	11.712	181	.065		
Total	SpCond_uS_per_cm	26509981.349	190			
	TDS_g_per_L	11.742	190			
	Turbidity_NTU	395501.340	190			
	Chl_RFU	1978.653	190			
	BGA_RFU	99.236	190			
Corrected Total	SpCond_uS_per_cm	12358219.647	189			
	TDS_g_per_L	3.364	189			
	Turbidity_NTU	147356.872	189			
	Chl_RFU	1196.334	189			
	BGA_RFU	37.157	189			

a. R Squared = .715 (Adjusted R Squared = .702)
b. R Squared = .716 (Adjusted R Squared = .704)
c. R Squared = .483 (Adjusted R Squared = .460)
d. R Squared = .837 (Adjusted R Squared = .830)
e. R Squared = .685 (Adjusted R Squared = .671)

Summer

2A	2B	MS6	MS7	MS5	CC1	CC2	CC3	CC4
(279.9)	(236.3)	(272.8)	(298.9)	(391.6)	(614.0)	(0.4)	(0.3)	(0.3)

Fall

CC1	MS6	2B	MS5	2A	MS7	CC4	CC2	CC3
(576.5)	(446.3)	(379.4)	(413.7)	(274.4)	(322.4)	(0.5)	(0.6)	(0.5)

Figure 4. Post hoc contrasts for specific conductance from ANOVA of summer and fall data (Tables 3 and 4). Sites that are joined by a horizontal bar were not significantly different ($\alpha = 0.05$). Means are listed below each site (µS/cm).

Summer

2A	2B	MS6	MS7	MS5	CC1	CC2	CC3	CC4
(0.18)	(0.15)	(0.18)	(0.19)	(0.25)	(0.40)	(-0.01)	(-0.01)	(-0.01)

Fall

CC1	MS6*	MS5	MS6*	MS7	2B	2A	CC4	CC2	CC3
(0.40)	(0.29)	(0.27)	(0.29)	(0.21)	(0.25)	(0.18)	(0.07)	(0.09)	(0.07)

Figure 5. Post hoc contrasts for TDS from ANOVA for summer and fall (Tables 3 and 4). Sites that are joined by a horizontal bar were not significantly different ($\alpha = 0.05$). Means are listed below each site (g/L). *Note that MS6 appears twice.

Summer

2B	2A	MS6	MS7	CC1	CC4	CC3	CC2	MS5
(38.5)	(49.3)	(43.2)	(48.6)	(52.8)	(51.2)	(65.9)	(74.9)	(142.0)

Fall

2A	MS6	MS7	2B	CC1	CC4	CC2	CC3	MS5
(15.9)	(10.2)	(12.7)	(29.6)	(18.7)	(67.6)	(53.1)	(56.8)	(61.9)

Figure 6. Post hoc contrasts for turbidity from ANOVA of summer and fall data (Tables 3 and 4). Sites that are joined by a horizontal bar were not significantly different ($\alpha = 0.05$). Means are listed below each site (NTU).

Summer

CC1	CC2	CC4	2A	2B	CC3	MS6	MS7	MS5
(5.8)	(3.0)	(1.3)	(2.6)	(2.2)	(1.3)	(19.8)	(14.8)	(9.6)

Fall

2A	2B	MS6	MS7	CC1	CC2	CC3	CC4	MS5
(0.8)	(1.1)	(2.0)	(1.1)	(1.9)	(1.1)	(0.9)	(1.5)	(8.5)

Figure 7. Post hoc contrasts for chlorophyll concentration from ANOVA of summer and fall data (Tables 3 and 4). Sites that are joined by a horizontal bar were not significantly different ($\alpha = 0.05$). Means are listed below each site (RFU).

Summer

CC1	2A	2B	MS6	CC2	CC3	CC4	MS7	MS5
(0.7)	(2.5)	(2.4)	(-0.2)	(2.7)	(4.8)	(5.8)	(7.8)	(9.9)

Fall

2A	2B	CC2	CC3	MS7	MS6	CC4	CC1	MS5
(0.5)	(0.5)	(0.6)	(0.6)	(0.5)	(0.5)	(0.6)	(0.2)	(1.5)

Figure 8. Post hoc contrasts for BGA concentration from ANOVA of summer and fall data (Tables 3 and 4). Sites that are joined by a horizontal bar were not significantly different ($\alpha = 0.05$). Means are listed below each site (RFU).

Table 5. Factor loading table for principal components analysis for the summer water quality dataset. Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance.

Total Variance Explained

Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
1	2.275	45.500	45.500	2.275	45.500	45.500
2	1.660	33.199	78.699	1.660	33.199	78.699
3	.758	15.161	93.860	.758	15.161	93.860
4	.307	6.137	99.997	.307	6.137	99.997
5	.000	.003	100.000	.000	.003	100.000

Extraction Method: Principal Component Analysis.

Table 6. Component matrix for principal components analysis for the summer water quality dataset. Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance.

Component Matrix[a]

	Component				
	1	2	3	4	5
MeanSpCond	.968	.000	-.244	.060	.009
MeanTDS_g_per_L	.971	.000	-.230	.065	-.009
MeanTurbidity_NTU	.106	.911	-.108	-.384	.000
Mean_Chlor_RFU	.609	.016	.791	-.060	.000
MeanBGA_RFU	-.116	.911	.094	.385	.000

Extraction Method: Principal Component Analysis.
a. 5 components extracted.

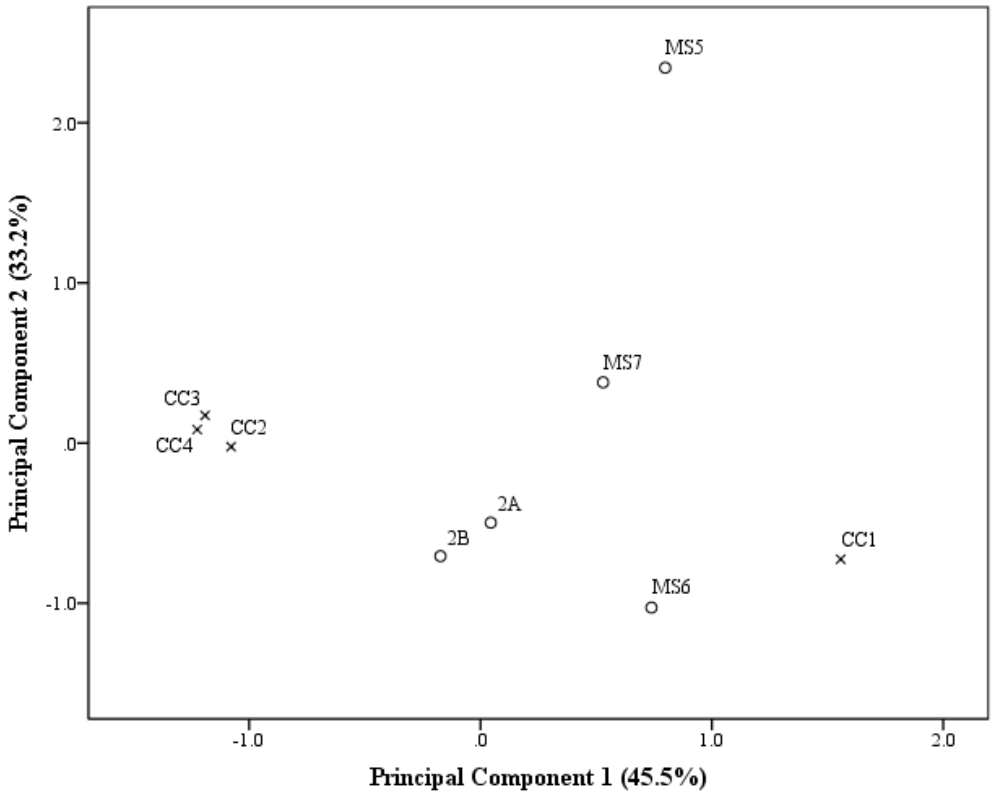

Figure 9. Plot of sample sites on first and second principal component axes for the summer water quality dataset (Table 5). Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance. Pool sites are represented as circles, while creek sites are represented as x's.

Table 7. Factor loading table for principal components analysis for the fall water quality dataset. Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance.

Total Variance Explained

Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
1	2.599	51.977	51.977	2.599	51.977	51.977
2	2.166	43.328	95.305	2.166	43.328	95.305
3	.218	4.360	99.665	.218	4.360	99.665
4	.014	.285	99.950	.014	.285	99.950
5	.003	.050	100.000	.003	.050	100.000

Extraction Method: Principal Component Analysis.

Table 8. Component matrix for principal components analysis for the summer water quality dataset. Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance.

Component Matrix[a]

	Component				
	1	2	3	4	5
MeanSpCond	.914	.396	.041	.068	-.026
MeanTDS_g_per_L	.905	.376	.197	-.018	.032
MeanTurbidity_NTU	-.880	.317	.353	.035	-.002
Mean_Chlor_RFU	-.064	.995	.001	-.075	-.019
Mean_BGA_RFU	-.408	.882	-.231	.050	.020

Extraction Method: Principal Component Analysis.
a. 5 components extracted.

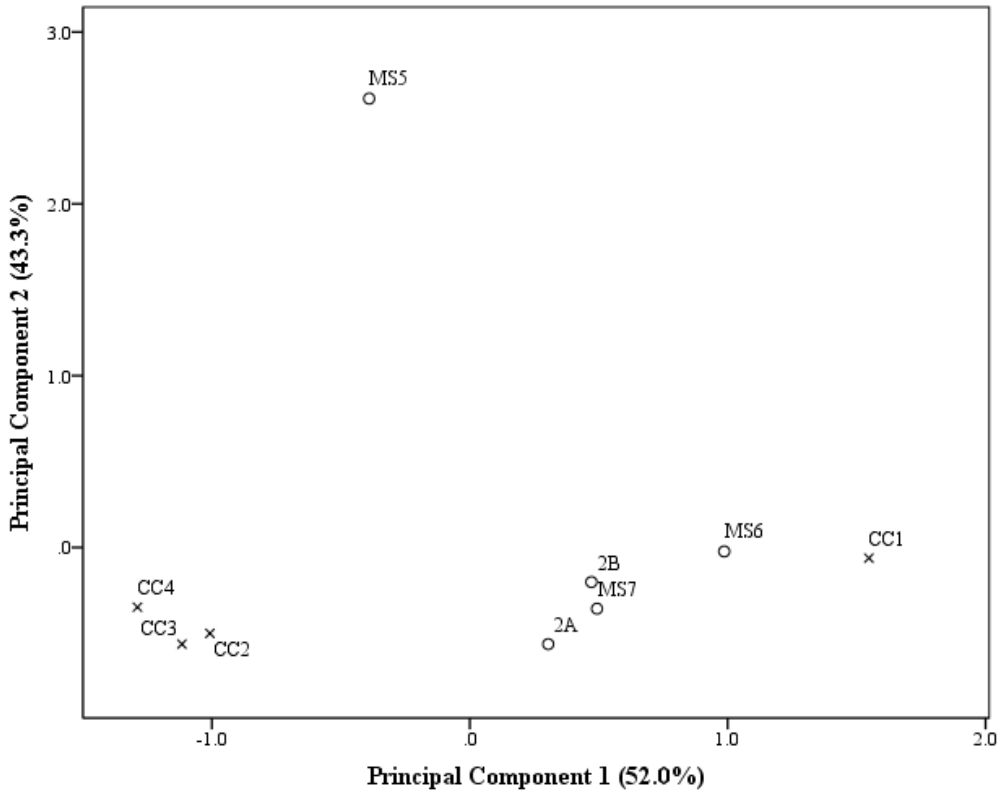

Figure 10. Plot of the first and second principal components of the fall water quality dataset (Table 6). Variables include: mean blue-green algae concentration, mean chlorophyll concentration, mean TDS, mean turbidity, and mean specific conductance. Pool sites are represented as circles, while creek sites are represented as x's.

Table 9. Complete list of taxa collected or observed incidentally at all sites. Listed tolerance values and feeding guilds are from the Wiley lab database. Feeding guilds: P = predator, Sc = scraper, F = filter feeder, C = collector/gatherer, and Sh = shredder. Collection: I = incidental, S = summer sampling event, and F = fall sampling event.

		Tolerance	Feeding Guild	2A	2B	MS5	MS6	MS7	CC1	CC2	CC3	CC4
Oligochaeta	Oligochaeta	9	C	F		S		F	SF	SF	S	SF
Hirudinea	Hirudinea	7.4	P	SF	SF	S	S	S	S	SF	SF	F
Veneroida												
Dreissenidae	*Dreissena*	10	F						S			
Sphaeriidae	Sphaeriidae	8	F	F				SF				
Gastropoda												
Ancylidae	Ancylidae	6	Sc	SF	SF		F	SF	S	S	SF	
Lymnaeidae	Lymnaeidae	6.9	Sc	I						S		S
Physidae	Physidae	8	Sc		SF							
Planorbidae	Planorbidae	7	Sc	SF	SF		SF	SF	SF	SF	SF	SF
Viviparidae	Viviparidae	6	Sc	I								
Isopoda												
Asellidae	*Caecidotea*	6	Sh					F			S	
	Lirceus	7.7	Sh		S							
Amphipoda												
Gammaridae	*Gammarus*	4	Sh	S	S		F		S		S	
Talitridae	*Hyalella*	8	Sh						S	S	S	S
Decapoda												
Cambaridae	Cambaridae*	7.2	Sh	SF	F	SF	SF	SF	SF	SF	SF	
	Orconectes	2.7	Sh		I			F				
Palaemonidae	*Palaemonetes*	4	C		S							
Hydrachnida	Hydrachnida	9	P			S		S	S			
Collembola	Collembola	10	C	I						I	I	
Ephemeroptera												
Baetidae	*Paracloeodes*	8.7	Sc		F							
Caenidae	*Caenis*	7	Sc/C		S					S		
Ephemeridae	*Hexagenia*	6	C					SF		S		
Odonata												
Aeshnidae	*Anax*	8	P	F	SF	SF	SF	SF	SF	SF	SF	S
	Epiaeschna	3	P		S							
	Nasiaeschna	8	P								I	
Coenagrionidae	*Amphiagrion*	9	P						S	SF		
	Ischnura	9	P	F		S	F	SF	SF	SF		F
Corduliidae	*Epitheca*	7	P	SF	SF		S	SF	S			
Gomphidae	*Arigomphus*	4	P						SF	F		
Lestidae	*Lestes*	9	P	SF	SF	SF	SF	SF	SF	SF	SF	S
Libellulidae	Libellulidae	9	P		S	S	F	SF	S			
	Erythemis	9	P	S								
	Leucorrhinia	9	P				S					
	Pachydiplax	9.6	P					S				
	Tramea	9	P	S	S		S	S				
Hemiptera												
Belostomatidae	*Belostoma*	9.8	P						F			
Corixidae	Corixidae	9	P		S			S		S	S	
	Hesperocorixa	7	C		S	S		S				
	Palmacorixa	5	P		F		F	F				
	Trichocorixa	5	P	F	S		SF	S		S	S	
Gerridae	*Gerris*	5	P	S	S							
	Metrobates	8	P		I							
	Neogerris	8	P	S					S	S		
	Rheumatobates	8	P		S			S	S		S	
	Trepobates	10	P	SF	SF	SF	SF	SF	SF	SF	SF	S
Hebridae	*Hebrus*	8	P	I	I		S	S	S	S	S	S
	Merragata	8	P						S			
Hydrometridae	*Hydrometra*	8	P	SF	S	S		SF				
Mesoveliidae	*Mesovelia*	8	P								S	
Naucoridae	*Pelocoris*	7	P				S					
Nepidae	*Ranatra*	7.5	P			S						

		Tolerance	Feeding Guild	2A	2B	MS5	MS6	MS7	CC1	CC2	CC3	CC4
Notonectidae	*Buenoa*	8	P	SF	SF	SF	SF	SF	SF	SF	S	F
	Notonecta	8	P		I							
Pleidae	*Neoplea*	8	P		I							
Veliidae	*Microvelia*	6	P		I							
Megaloptera												
Corydalidae	*Chauliodes*	4	P	F			F	F	S			S
Trichoptera												
Trichoptera	Trichoptera	3	Sc/C	S				S				
Hydroptilidae	*Orthotrichia*	6	Sh				F	S				
	Stactobiella	2	Sh				S					
Leptoceridae	*Oecetis*	8	P/Sh		S							
	Ylodes	4	C/Sh	F				SF				
Phryganeidae	Phryganeidae	4	Sh		SF			F	SF		SF	SF
Lepidoptera												
Pyralidae	*Acentria*	1	Sh		I				S	SF	SF	SF
	Nymphula	7	Sh		I							
Coleoptera												
Coleoptera	Coleoptera	5	P	SF	SF	S	SF	SF	SF	SF	S	S
Chrysomelidae	*Donacia*	8	Sh	SF	SF	SF	SF	SF	SF	SF	SF	SF
Curculionidae	*Notiodes*	5	P					S				F
Dytiscidae	*Agabetes*	5	P		F					SF		
	Cybister	5	P	S					S			
	Graphoderus	5	P	SF	SF		SF	S	SF	SF	SF	SF
	Hydrovatus	6	P		S			S				
	Hygrotus	6	P	SF	S	SF	SF	SF	SF	S	S	
	Ilybius	5	P	F	S		SF	SF	S	S		
	Laccophilus	7.9	P	S	S			S	F		S	
Elmidae	*Dubiraphia*	6	C						S			
Gyrinidae	*Dineutus*	3.7	P						S			
	Gyrinus	3.6	P		SF				F	S	S	SF
Haliplidae	*Haliplus*	7	Sh		F	F	SF	SF	F		F	F
	Peltodytes	7	P/Sh						SF	S	S	SF
Hydrophilidae	*Anacaena*	5	C	S	S	S	S	SF	SF	SF	SF	S
	Berosus	6.7	C/Sh		F				S	SF	SF	SF
	Enochrus	8.5	C	SF	F			F	S			
	Helocombus	5	C		S		S		S			
	Tropisternus	10	C		I							
Noteridae	*Hydrocanthus*	6.9	P	S	S			S		S		
Staphylinidae	Staphylinidae	8	P		SF	S	SF		F	SF	F	
Diptera												
Ceratopogonidae	Ceratopogonidae	5.7	P/C		I							S
Chironomidae	Chironomidae	7	F	SF		S	SF	SF	S	S	S	
Culicidae	*Anopheles*	6	F							SF	SF	
	Uranotaenia	10	F		SF							
Sciomyzidae	*Dictya*	6	P						F			
Stratiomyidae	*Stratiomys*	7	F	SF	SF		F	SF				S
Tabanidae	*Tabanus* or *Atylotus*	5	P		F				S			
Tipulidae	*Helius*	4	C/Sh	F	SF	SF	SF	SF	F	SF	SF	SF

*Only females were observed and in the absence of males they cannot be identified past family (Pennak 1989).

Table 10. Number of taxa collected at each site broken down by collection time.

Site	Summer	Fall	Incidental	Total unique taxa
2A	26	25	4	39
2B	39	26	10	57
MS5	20	9	0	21
MS6	25	24	0	33
MS7	39	30	0	46
CC1	40	23	0	47
CC2	35	21	1	37
CC3	31	17	2	35
CC4	21	15	0	26

Table 11. Average tolerance of the unique taxa collected at each site, and number of sensitive taxa collected at each site (sensitive taxa have a tolerance less than 4).

Site	Average Tolerance	Sensitive Taxa
2A	7.30	4
2B	7.18	4
MS5	7.40	1
MS6	7.28	1
MS7	7.24	2
CC1	7.03	8
CC2	6.97	6
CC3	6.93	5
CC4	7.05	5

Table 12. Summary of amounts of surface dependent, metabolic conforming, EPT, or Isopoda-Hirudinea-Gastropoda taxa found for each site.

Site	Taxa	Surface Dependent		Metabolic Conformers		EPT Taxa		Isopoda-Hirudinea-Gastropoda	
		n	%	n	%	n	%	n	%
2A	39	24	62	3	8	6	15	6	15
2B	57	40	70	3	5	4	7	7	12
MS5	21	14	67	1	5	2	10	4	19
MS6	33	29	88	2	6	4	12	5	15
MS7	45	43	96	2	4	4	9	7	16
CC1	47	27	57	3	6	4	9	10	21
CC2	37	21	57	6	16	5	14	12	32
CC3	35	20	57	4	11	5	14	10	29
CC4	26	20	77	3	12	2	8	3	12

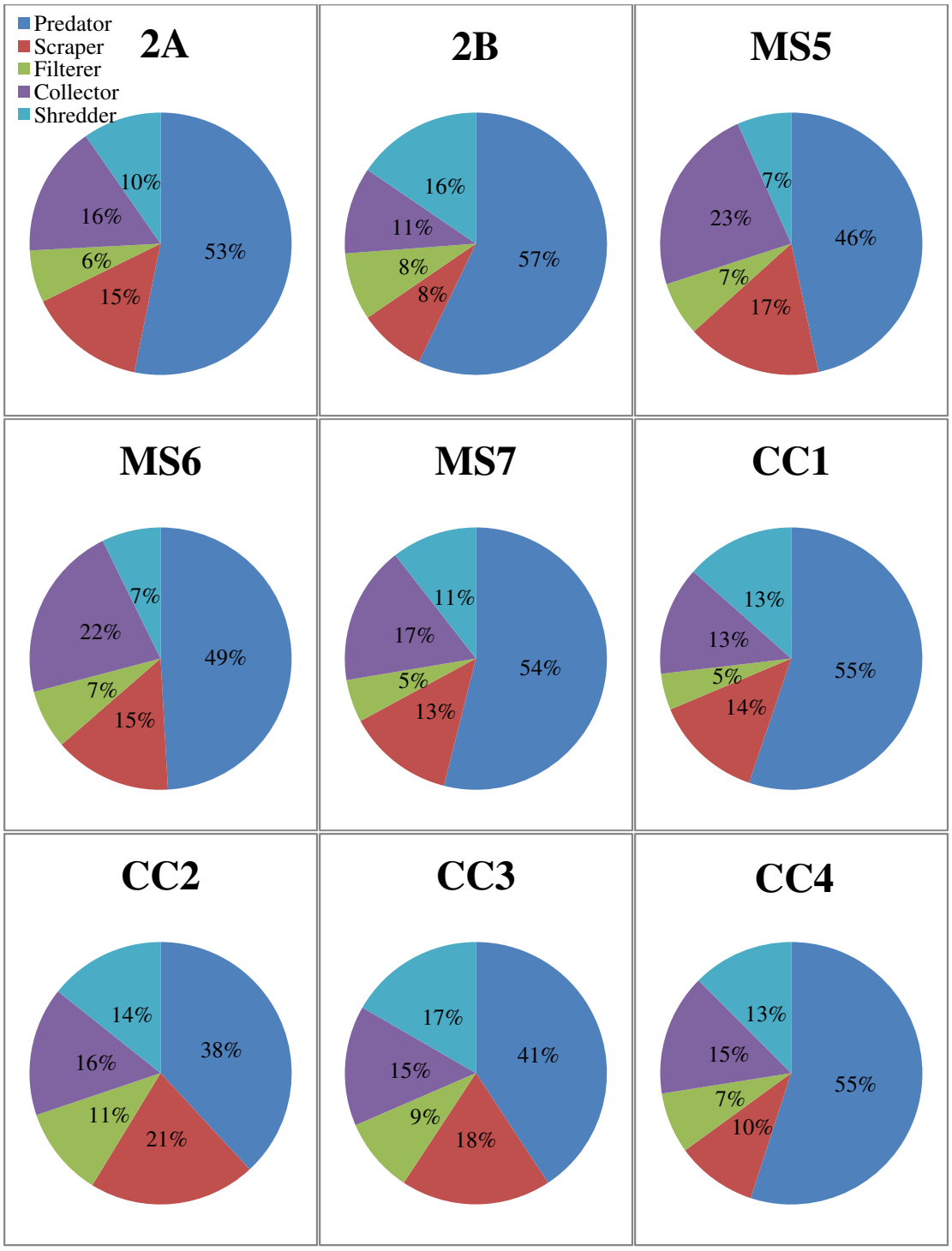

Figure 11. Percent of taxa collected in each feeding guild at each site.

Table 13. Sørensen similarity quotients for each site compared to each other site. Perfectly similar sites would generate a value of 1.000, while perfectly dissimilar sites would generate a value of 0.000. Values between 0.400 and 0.499 are colored red, between 0.500 and 0.599 are orange, between 0.600 and 0.699 are light green, and between 0.700 and 0.799 are dark green. Values of 1.000 are colored grey.

	2A	2B	MS5	MS6	MS7	CC1	CC2	CC3	CC4
2A	1.000								
2B	0.542	1.000							
MS5	0.500	0.410	1.000						
MS6	0.667	0.578	0.593	1.000					
MS7	0.729	0.641	0.567	0.684	1.000				
CC1	0.605	0.577	0.529	0.625	0.602	1.000			
CC2	0.632	0.532	0.517	0.571	0.554	0.667	1.000		
CC3	0.568	0.565	0.536	0.588	0.593	0.683	0.750	1.000	
CC4	0.523	0.458	0.511	0.542	0.528	0.603	0.635	0.656	1.000

Table 14. Factor loading table for the principal components analysis of the invertebrate community composition dataset. Variables used were all taxa found with the exception of those located at all sites: *Caenis*, Chironomidae, Hirudinea, *Hyalella*, *Ischnura*, Lymnaeidae, *Notonecta*, Physidae, and *Trichocorixa*.

Total Variance Explained

Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
1	22.051	26.251	26.251	22.051	26.251	26.251
2	14.434	17.184	43.435	14.434	17.184	43.435
3	11.375	13.542	56.976	11.375	13.542	56.976
4	9.767	11.628	68.604	9.767	11.628	68.604
5	7.439	8.856	77.460	7.439	8.856	77.460
6	7.076	8.424	85.885	7.076	8.424	85.885
7	6.870	8.179	94.063	6.870	8.179	94.063
8	4.987	5.937	100.000	4.987	5.937	100.000

Extraction Method: Principal Component Analysis.

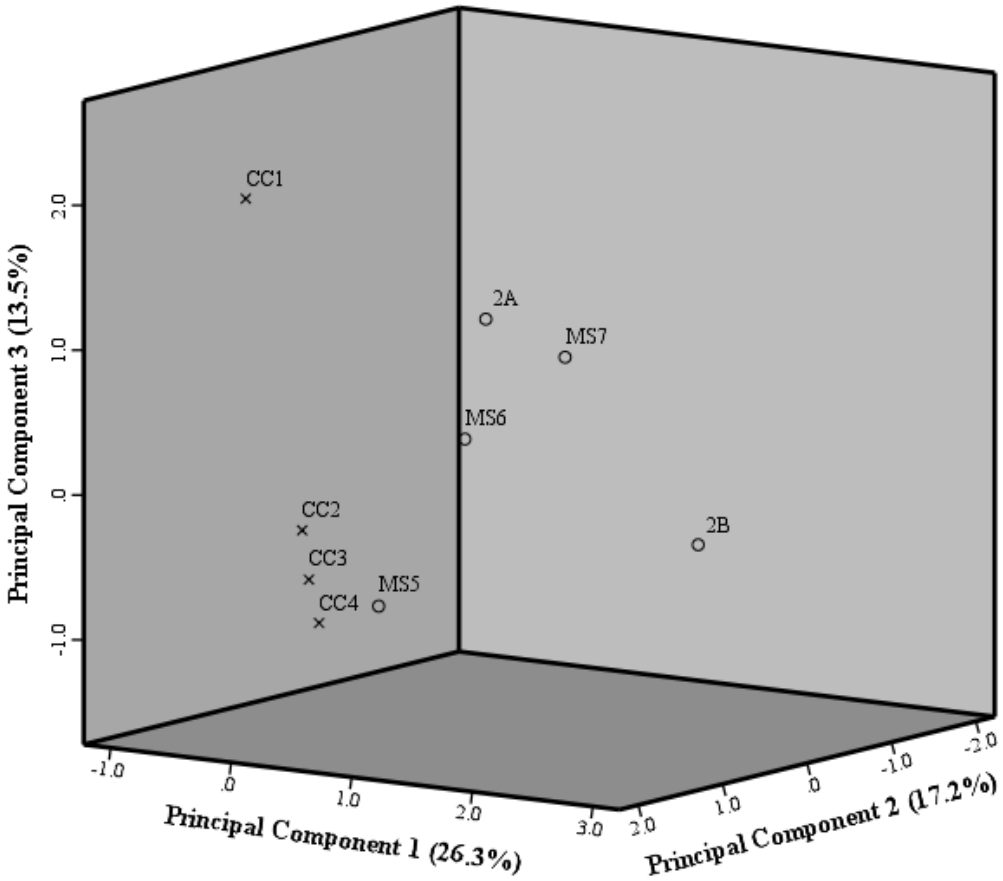

Figure 12. Plot of the first, second, and third principal components of the invertebrate community composition (Table 12). Pool sites are represented as circles, while creek sites are represented as x's. The following taxa were removed due to their presence at every site: *Caenis*, Chironomidae, Hirudinea, *Hyalella*, *Ischnura*, Lymnaeidae, *Notonecta*, Physidae, and *Trichocorixa*.

Works Cited

Albert, D.A., D.A. Wilcox, J.W. Ingram, and T.A. Thompson. 2005. Hydrogeomorphic classification for Great Lakes coastal wetlands. *International Association of Great Lakes Research* 31: 129-146.

Batzer, D.P. and S.A. Wissinger. 1996. Ecology of insect communities in nontidal wetlands. *Annual Review of Entomology* 41: 75-100.

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

Bowers, R. and F.A. de Szalay. 2005. Effects of water level fluctuations on zebra mussel distribution in a Lake Erie coastal wetland. *Journal of Freshwater Ecology* 20: 85-92.

Bridgeman, T.B., J.D. Chaffin, D.D. Kane, J.D. Conroy, S.E. Panek, and P.M.Armenio. 2012. From river to lake: phosphorus partitioning and algal community compositional changes in Western Lake Erie. *Journal of Great Lakes Research* 38: 90-97.

Brookhout, T.A., K.E. Bednarik, and R.W. Kroll. 1989. The Great Lakes' marshes. *In*: Habitat management for migrating and wintering waterfowl in North America. Texas Tech University Press, Lubbock, Texas. pp. 131-156

Cloern, J.E. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. *Continental Shelf Research* 7: 1367-1381.

Cooper, M.J., D.G. Uzarski, and T.M. Burton. 2007. Macroinvertebrate community composition in relation to anthropogenic disturbance, vegetation, and organic sediment depth in four Lake Michigan drowned river-mouth wetlands. *Wetlands* 27:894-903.

Cvetkovic, M. and P. Chow-Fraser. 2011. Use of ecological indicators to assess the quality of Great Lakes coastal wetlands. *Ecological Indicators* 11:1609-1622.

Friday, L.E. 1987. The diversity of macroinvertebrate and macrophyte communities in ponds. *Freshwater Biology* 18: 87-104.

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

Herdendorf, C.E. 1987. *The ecology of the coastal marshes of western Lake Erie: a community profile*. U.S. Fish Wildl. Serv. Biol. Rep. 85(7.9). 171 pp.

Herdendorf, C.E. 1992. Lake Erie coastal wetlands: an overview. *Journal of Great Lakes Resarch* 18: 533-551.

Herrick, B.M. and A.T. Wolf. 2005. Invasive plant species in diked vs. undiked Great Lakes wetlands. Journal of Great Lakes Research 31: 277-287.

Hilsenhoff, W.L. 1995. *Aquatic Insects of Wisconsin: Keys to Wisconsin Genera and Notes on Biology, Habitat, Distribution, and Species.* Natural History Museums Council University of Wisconsin-Madison.

Huffman, R. 2011. Personal communication regarding management goals and strategies for wetland pools at Ottawa National Wildlife Refuge.

Johnson, D.L., W.E. Lynch, and T.W. Morrison. 1997. Fish communities in a diked Lake Erie wetland and an adjacent undiked area. *Wetlands* 17: 43-54.

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

Kasat, R. 2006. Nutrient dynamics in a small Lake Erie tributary. Master of Science Thesis, University of Michigan, School of Natural Resources and Environment. Ann Arbor, Michigan, USA.

Keough, J.R., T.A. Thompson, G.R. Guntenspergen, and D.A. Wilcox. 1999. Hydrogeomorphic factors and ecosystem responses in coastal wetlands of the Great Lakes. *Wetlands* Vol 19 4: 821-834.

Kowalski, K. 2010. Overcoming barriers to coastal wetland ecosystem rehabilitation: strategies for the Great Lakes. PhD dissertation, University of Michigan, School of Natural Resources and Environment, Ann Arbor, MI, USA.

Kowalski, K. 2012. Personal communication regarding the future of research projects at Ottawa National Wildlife Refuge.

Kowalski, K. and D.A. Wilcox. 1999. Use of historical and geospatial data to guide the restoration of a Lake Erie coastal marsh. *Wetlands* 19: 858-868.

Kreiger, K.A. 1992. The ecology of invertebrates in the Great Lakes coastal wetlands: current knowledge and research needs. *Journal of Great Lakes Research* 18:634-650.

Kulesza, A.E., J.R. Holomuzki, and D.M. Klarer. 2008. Benthic community structure in stands of *Typha angustifolia* and herbicide-treated and untreated *Phragmites australis*. *Wetlands* 28: 40-56.

MDEQ (Michigan Department of Environmental Quality). 1997. GLEAS Procedure #51 Survey Protocols for Wadable Rivers. Chapter 25A *in* Schneider, James C. (ed.) 2000. Manual of fisheries survey methods II: with periodic updates. Michigan Department of Natural Resources, Fisheries Special Report 25, Ann Arbor, MI, USA.

Merritt, R.W., K.W. Cummins, and M.B. Berg. 2008. *An Introduction to the Aquatic Insects of North America*. Kendall/Hunt Publishing Company. Dubuque, Iowa.

Mitsch, W.J. and N. Wang. 2000. Large-scale coastal wetland restoration on the Laurentian Great Lakes: determining the potential for water quality improvement. *Ecological Engineering* 15:267-282.

National Weather Service Forecast Office. 2012. Toledo, OH Climate. http://www.erh.noaa.gov/cle/climate/tol/climatetol.html. Accessed 15 June 2012.

NOAA (National Oceanic and Atmospheric Administration). 25 August 2011. Experimental Lake Erie Harmful Algal Bloom Bulletin. 2011-012. http://www.glerl.noaa.gov/res/Centers/HABS/lake_erie_hab/lake_erie_hab.html. Accessed 11 June 2012.

NOAA (National Oceanic and Atmospheric Administration). 2012. Harmful Algal Blooms in Lake Erie – Experimental HAB Bulletin. http://www.glerl.noaa.gov/res/Centers/HABS/lake_erie_hab/lake_erie_hab.html. Accessed 28 July 2012.

Pennak, R.W. 1989. *Fresh-water Invertebrates of the United States: Protozoa to Mollusca*. 3rd ed. Wiley. New York, New York.

Provence, C.D. 2008. Effects of diking and plant zonation on invertebrate communities of Lake St. Clair coastal marshes. Master of Science Thesis, Michigan State University. East Lansing, Michigan, USA.

Richards, R.P., and D.B. Baker. 2002. Trends in water quality in LEASEQ rivers and streams (Northwestern Ohio), 1975-1995. *Journal of Environmental Quality* 31: 90-96.

Thiet, R.K. 2002. Diversity comparisons between diked and undiked coastal freshwater marshes on Lake Erie during a high-water year. *Journal of Great Lakes Research* 28: 285-298.

USEPA (U.S. Environmental Protection Agency). Maumee River Area of Concern. http://www.epa.gov/greatlakes/aoc/maumee.html. Accessed 12 June 2012.

USGS (U.S. Geological Survey). 2012. New Strategies for Restoring Coastal Wetland Function: Summary Annual Report – 2011. Annual report to U.S. Fish and

Wildlife Service Ottawa National Wildlife Refuge, Oak Harbor, Ohio. 58 p.

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

Wei, A., P. Chow-Fraser, and D. Albert. 2004. Influence of shoreline features on fish distribution in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 61: 1113-1123.

Wiley, M.J., M.E. Baker, and P.W. Seelbach. 1988. Summer field sampling and preliminary analysis of Michigan stream assemblages. Final Report to the Nature Conservancy. University of Michigan. Ann Arbor.

YSI Incorporated. 6-Series Multiparameter Water Quality Sondes User Manual. http://www.ysi.com/media/pdfs/069300-YSI-6-Series-Manual-RevF.pdf. Downloaded 11 November 2011.