Overcoming Barriers to Coastal Wetland Ecosystem Rehabilitation:

Strategies for the Great Lakes

by

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True knowledge exists in knowing that you know nothing
- Socrates

The Laurentian Great Lakes

To my beloved wife Kim

Your steadfast love, friendship, and support are my foundation

We are blessed in so many ways

To Joshua and Andrea

Your patience and understanding were appreciated more than you know

I hope this dissertation serves as an example of what can be accomplished with hard work, self sacrifice, and the support of family

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List of Abbreviations

Abbreviation	**Definition**
ANOVA	ANalysis Of VAriance
CCCH	Chapter 4: Crane Creek CHannel site
CCLow	Chapter 3: Crane Creek Lower site
CCLW	Chapter 4: Crane Creek Lower Wetland site
CCUp	Chapter 3: Crane Creek Upper site
CCUW	Chapter 4: Crane Creek Upper Wetland site
CPUE	Catch-Per-Unit-Effort
DIDSON	Dual-frequency IDentification SONar
DIN	Dissolved Inorganic Nitrogen
DO	Dissolved Oxygen
ENS	Chapter 4: Erie Nearshore Site
hPa	HectoPascal
LO_5	Chapter 2: Low Open sampling group (2005)
Mg/l	Milligrams per liter
MHz	MegaHertz
NMDS	Non-metric MultiDimensional Scaling
NTU	Nephelometric Turbidity Units
ONWR	Ottawa National Wildlife Refuge
PC-ADP	Pulse Coherent Acoustic Dopplar Profiler
SAV	Submersed Aquatic Vegetation
SNRE	School of Natural Resources and Environment
SRP	Soluble Reactive Phosphorus
uS	MicroSiemens
YOY	Young-Of-Year
°C	Degrees Celsius

Abstract

Overcoming Barriers to Coastal Wetland Ecosystem Rehabilitation:

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Chair: Michael J. Wiley

Great Lakes coastal wetlands provide many important ecological functions and values, but most of these highly productive systems have been degraded or destroyed by anthropogenic stressors. The multidimensional nature of wetland degradation presents challenges for habitat rehabilitation, but rehabilitation efforts designed to mimic natural processes could yield positive results. In this dissertation, I explored two hydrology-related habitat rehabilitation strategies (i.e., short-term management-induced dewatering to mimic cyclic low water levels and reducing hydrologic isolation typically associated with diked wetland units) applied to the riverine and diked wetlands at Crane Creek, a small western Lake Erie tributary.

Initially, I studied the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural growth of emergent vegetation from the seed bank. A short dewatering stimulated a rapid seed-bank-driven response by 45 plant taxa, but submersed aquatic species reestablished after subsequent flooding. Although long-

term habitat rehabilitation using this technology may be difficult, it could be an important tool for resource managers.

Fishes, plants, and water quality in the wetland complex were sampled to describe spatial and seasonal patterns of fish assemblages and explore habitat rehabilitation through hydrologic reconnection of diked wetlands and Lake Erie. Pronounced differences were found in hydrology (water-level fluctuation), fish assemblages (composition and abundance), and wetland vegetation (composition) between the diked and coastal wetlands, suggesting that a fish-passage structure and periodic management actions could improve habitat and restore seasonal access to Lake Erie fishes.

Finally, I quantified wetland use (abundance and movement) by Lake Erie fishes using a high-resolution sonar (DIDSON). Despite very dynamic environmental conditions, the degraded Crane Creek wetlands supported an abundance of fishes that moved extensively through the channel connecting to Lake Erie. Longnose gar, shoals of small fish, and other unidentifiable large fish used the channel as a temporary habitat and to escape diurnally poor water quality.

Results of my research suggest that rehabilitation strategies that account for ecosystem complexity and mimic natural hydrologic processes (e.g., water-level variability, habitat connectivity) can benefit wetland ecosystems on multiple dimensions. Finally, numerous management objectives could be met through function-based rotation of wetlands in the landscape.

Chapter 1
Introduction

1.1 Overview

Although now considered highly valuable, many Great Lakes coastal wetlands have been destroyed since the late 1800s, and those that remain are often severely degraded by systematic dredging, diking, and/or draining since the late 1800s. There currently is significant national interest in restoring or rehabilitating these important habitats as part of the Great Lakes Restoration Initiative, a $475M (FY10) initiative targeting the most significant environmental problems in the region (EPA 2009). Objectives of the Initiative and components of the 2005 Great Lakes Regional Collaboration Strategy (GLRC 2009) focus on implementing on-the-ground rehabilitation activities, but it is not always clear how to attack the problems. Degradation of coastal wetlands involves a complex multidimensional set of problems, including altered hydrology, poor water quality, and invasive species. Interactions among the dimensions (e.g., altered hydrology can lead to poor water quality) complicate things further, so clear and universally applicable best management practices and rehabilitation techniques that can be applied routinely in degraded coastal wetlands of the Great Lakes do not exist. Therefore, the need for new and innovative approaches to habitat rehabilitation is large.

One potential approach to this challenge is to identify site-specific rehabilitation strategies that are based on the unique "ecological design space" of each individual wetland ecosystem. *Ecological design space* is one way to think about how all biotic and abiotic variables of an ecosystem interact with one another to produce the wetland unit we see and generate the wetland's relative value. It is loosely similar to the idea of the *adaptive landscape* that Wright (1932, 1988) introduced when he was describing how populations can change genetically over time. An adaptive landscape, as described by Wright, suggests that the relationship between genotypes and reproductive success could be visualized as a landscape with replication rates (reproductive success or fitness) defining the height of the landscape. Ecological design space is analogous to multidimensional *evolutionary design space*, as described by Dennett (1995). To Dennett (1995), evolutionary design space is a philosophical concept used to help describe his views of biology, natural selection, and a suite of other topics. Ecological design space, as I am using the term, is a concept of interest in the context of ecosystem rehabilitation. It differs from the concepts of Wright and Dennett in that 1) the focus is on the varying surface of ecological integrity and valued ecosystem endpoints (e.g., biological diversity, productivity, increasing water quality in some measurable way) for a specific ecosystem unit, rather than on the reproductive fitness of individual organisms or specific populations, and 2) for my purposes, its dimensionality must include anthropogenic stressors, design management actions (e.g., hydro-geomorphic manipulation such as diking and gated flow structures commonly used in wetland management), and naturally occurring ecological processes and variables. Within every design space, there are good areas (e.g., high value areas yielding essentially pristine or otherwise desirable ecosystem characteristics), bad areas (e.g., low value, highly degraded system composition or structure, often associated with many long-term stressors), and gradients of structural change in between.

From this perspective, the goal of any rehabilitation program should be to move the targeted system toward a better area (i.e., higher integrity and value) within its ecological design space. Management actions are the tools that can be used to force system moves within design space. Finding strategies for moving an ecosystem in a helpful direction that ascends (i.e., is a good move) and not descends (i.e., is a bad move) the design topography becomes a critical task in which ecologists can play an especially helpful role. In some cases, good moves might be accomplished rapidly across many design dimensions at once (e.g., a true ecosystem-wide rehabilitation event). More often, however, ecological knowledge, political, or logistical constraints limit the scope and frequency of possible rehabilitation actions. Given this context of constraint, it would be useful to identify not only potentially good moves, but also “smart” moves in design space. A *smart move* will both proceed in the right direction and help establish an efficient, sustainable trajectory of change that will facilitate future moves towards improved ecosystem condition. Likewise, it also is very desirable to avoid *poor* moves -- steps that might be in a good direction for a particular dimension but which set up difficulties with other dimensions (e.g., recreating natural hydrologic conditions that lead to invasion by exotic plant species) or begin trajectories that lead to long-term declines, make further good moves more difficult, or otherwise lead to design space dead ends.

In order to understand these issues more fully, I studied the Crane Creek drowned-river-mouth wetland complex in western Lake Erie. Crane Creek is an excellent example of a highly-stressed and degraded coastal wetland system. It drains an agriculture-dominated watershed and is influenced by point-source wastewater treatment effluent, both of which can contribute to poor water quality (e.g., high turbidity, high nutrient concentrations). Its hydrology has been altered by dredging and revetment-lined banks. Invasive fish and plant species are prevalent and often dominate the ecosystem.

This multidimensional set of stressors presents a great opportunity for research, especially because lower Crane Creek lies completely within the boundaries of the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge. I examined the pros and cons of two very practical rehabilitation activities relevant to the past and future operations of the refuge at Crane Creek. My goal was to identify smart moves within ecological design space related to 1) extended high water levels and wetland plant degradation and 2) hydrologic isolation typically associated with diked wetlands.

1.2 Background

The need for ecological research in coastal wetlands of the Great Lakes, especially in western Lake Erie, is well established (Ball 1985, Herdendorf 1987, Johnson 1989, Jude and Pappas 1992, Mitsch et al. 2001, Mayer et al. 2004). Over 96% of the original wetland habitats along the U.S. shoreline of western Lake Erie have been lost since the 1860s (Herdendorf 1987, Mitsch and Wang 2000), and most of the remaining coastal wetlands have been isolated by earthen dikes to protect them from wave attack and promote management as migratory waterbird habitat. Although these diked wetlands are adjacent to the Lake Erie shoreline, they no longer provide many of the functions of coastal wetlands (e.g., fish habitat, nutrient cycling) because they are hydrologically separated from the lake. Fishes, clams, and organisms with limited mobility are impacted the most because they are not able to use the diked wetland habitats similarly to hydrologically connected wetlands (e.g., for seasonal movement). Unfortunately, most of the few remaining undiked wetlands are severely degraded (Herdendorf 1987, Maynard and Wilcox 1997, Kowalski and Wilcox 1999). They remain hydrologically connected to the lake, but the water quality and wetland vegetation that provides vital fish habitat are sufficiently degraded to impact negatively the approximately 43 species of Great Lakes fishes that use wetland habitats as spawning and nursery locations (Jude and Pappas 1992). These degraded conditions favor very tolerant taxa, such as carp and bullheads, while less tolerant taxa can be outcompeted.

Many factors contribute to the long-term degradation of wetland habitats in Lake Erie, including damage done by carp (King and Hunt 1967), high turbidity (Barry et al. 2004), and wave action (Whillans 1996), but altered hydrology is often the driving force for degradation (Wilcox 1995a, Maynard and Wilcox 1997). Before early settlers began hardening the shoreline with dikes and rock revetments, emergent vegetation of coastal wetlands would occur in upslope (i.e., higher elevation) areas during periods of high lake levels and return to downslope areas (i.e., lower elevation) during subsequent lows (Wilcox 1995b). Now, the extensive armoring of the shoreline designed to protect municipal, agricultural, residential, and commercial properties, as well as diked wetlands, both prevents natural wetlands from existing upslope (Sherman et al. 1996) and alters the natural wetland hydrology. The result is flooding and destruction of emergent vegetation in hydrologically-connected coastal wetlands during extended periods of high water level. Water levels low enough to expose the wetland seedbank during the growing season, promote seed germination, and allow wetland plants to reestablish over broad areas have not occurred naturally in recent history (NOAA 2006), which has resulted in loss of critical fish and wildlife habitat.

1.3 Contents

In this dissertation, I explore aspects of two rehabilitation strategies for Great Lakes coastal wetland habitats: 1) short-term, management-induced dewatering to mimic cyclic low water levels, and 2) minimizing the hydrologic isolation typically associated with diked wetlands. Specifically, I focus on the complex of drowned-river-mouth and diked wetlands (i.e., Pool 2A and Pool 2B) at Crane Creek, a small stream flowing into western Lake Erie. I also use a high-resolution sonar to quantify the abundance and flux rates of Great Lakes fishes using these degraded wetland habitats.

My second chapter explores the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural growth of emergent vegetation from the seed bank in a 10-ha section of Crane Creek. The test area was dewatered to mimic a low-water year, and wetland seed bank response at differing elevations was characterized. This chapter was published as presented here in the Journal of Great Lakes Research (Kowalski, K. P., D. A. Wilcox, and M. J. Wiley. 2009. Stimulating a Great Lakes coastal wetland seed bank using portable cofferdams: implications for habitat rehabilitation. Journal of Great Lakes Research 35(2):206-214).

The third chapter begins an exploration of the ecological impacts of traditional diked wetland management strategies and the potential benefits of restoring hydrologic functionality. My research involves sampling fishes, plants, and water quality in the Crane Creek wetland complex to characterize spatial and seasonal patterns of fish assemblages. It also examines the implications of habitat rehabilitation by reestablishing a hydrologic connection between diked wetlands and Lake Erie. Alternatives to the current diked unit management strategies, including extended hydrologic reconnection to maximize habitat availability to fishes coupled with periodic dewaterings to restart plant succession, are discussed.

The fourth chapter takes a much more detailed look at the abundance and movement of Great Lakes fishes between Crane Creek and the Lake Erie nearshore. Fish data collected with a high-resolution sonar (DIDSON) are analyzed along with water-level, water-flow, and water-quality data to characterize how Great Lakes fishes respond to dynamic water quality in the wetland and how they use a connecting channel to Lake Erie for both wetland access and regress. I also discuss the implications of these results for habitat rehabilitation design including how the time and duration of hydrologic connectivity may impact fish passage.

I conclude my dissertation in Chapter 5 by integrating the major results from each component of the study and discussing how the multidimensional nature of coastal wetland ecosystem stressors require us to identify “smart” habitat rehabilitation actions. I explain how the two hydrology-related rehabilitation strategies examined in my work led to a multidimensional strategy that could be a smart move on multiple dimensions. I describe a unique management approach that maximizes the ecological benefits to Lake Erie by supplementing the long-term hydrologic reconnection of diked wetlands with occasional management actions that mimic the intermediate level of disturbance associated with low water levels. Finally, I describe the implications of applying this approach to Lake Erie wetlands on a regional scale and explain how these actions could be a smart move in the design space of Lake Erie coastal wetlands.

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Chapter 2

Stimulating a Great Lakes coastal wetland seed bank using portable cofferdams: implications for habitat rehabilitation

2.1 Abstract

Coastal wetland seed banks exposed by low lake levels or through management actions fuel the reestablishment of emergent plant assemblages (i.e., wetland habitat) critical to Great Lakes aquatic biota. This project explored the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural growth of emergent vegetation from the seed bank in a Lake Erie coastal wetland. A series of dams stretching approximately 450 m was installed temporarily to isolate hydrologically a 10-ha corner of the Crane Creek wetland complex from Lake Erie. The test area was dewatered in 2004 to mimic a low-water year, and vegetation sampling characterized the wetland seed bank response at low, middle, and high elevations in areas open to and protected from bird and mammal herbivory. The nearly two-month drawdown stimulated a rapid seed-bank-driven response by 45 plant taxa. Herbivory had little effect on plant species richness, regardless of the location along an elevation gradient. Inundation contributed to the replacement of immature emergent plant species with submersed aquatic species after the dams failed and were removed prematurely. This study revealed a number of important issues that must be considered for effective long-term implementation of portable cofferdam technology to stimulate wetland seed banks, including duration of dewatering, product size, source of clean water, replacement of damaged dams, and regular maintenance. This technology is a potentially important tool in the arsenal used by resource managers seeking to rehabilitate the functions and values of Great Lakes coastal wetland habitats.

2.2 Introduction

There is a complex but well-established cyclical relationship between the seed bank and emergent wetland vegetation in systems with fluctuating water levels like those in the Laurentian Great Lakes. Although confounded by other factors (e.g., changing species pool, shoreline structures, wave action), the pattern of water-level fluctuation is critical to development and renewal of shoreline wetland plant communities (Keddy and Reznicek 1985, Wilcox 2004, Wilcox and Nichols 2008). In fact, the extent and diversity of coastal wetlands is driven by changes in water levels (Keddy and Reznicek 1986).

Water levels in the unregulated Great Lakes fluctuate on many scales (e.g., hourly, seasonally, annually, multiple-years). Although short-term hourly changes (i.e., seiches) and seasonal variations can affect plant community distribution (Batterson et al. 1991), it is the annual and multiple-year water-level changes that influence wetland plant communities most (Maynard and Wilcox 1997). Shoreline and wetland plant assemblages have adapted to and thrive on cycling periods of low and high water levels. Each part of the cycle causes a moderate disturbance or stress to the ecosystem that plays a vital role in the long-term maintenance of diverse wetland plant communities.

As the water retreats during low-water periods and sediments are exposed, a number of physical and biological changes occur. Submersed aquatic and floating species are lost because there is no water to support them, but previously flooded mud flats, often containing very rich seed banks, oxygenate to some extent when exposed to air (Ponnamperuma 1972). If water levels recede during the growing season, buried seeds in the seed bank germinate and normally reestablish a high diversity of mudflat and emergent vegetation (Harris and Marshall 1963, van der Valk and Davis 1978, Smith and Kadlec 1983, Barry et al. 2004). Unless water levels rise again or the site is further disturbed by other forces (e.g., herbivory), some mudflat wetland plants are able to mature in one year and add their seeds to the seed bank. Many emergent species, however, need multiple growing seasons to mature enough to produce seeds (van der Valk and Davis 1978). Given enough time to grow, emergent species replenish the seed bank and prepare the mud flat for the next time it is exposed after flooding. Woody plants and shrubs requiring drier conditions are able to colonize and grow at lower elevations during longer low water periods. Over time, they often begin to dominate and out-compete the emergent vegetation (Maynard and Wilcox 1997). If water levels remain low over decades, then succession occurs until disrupted by the next series of high water levels (Wilcox 2004).

The alternate phase of the cycle begins when water levels rise. Flooding in lower elevations changes the sediments from oxic to anoxic (Ponnamperuma 1972), inundates mudflat species (van der Valk 1981), and stresses or kills trees and woody plants (Keddy and Reznicek 1986). Similarly, soils in higher elevation areas that do not flood may become much wetter, thereby creating a lethal environment for trees and woody plants. As these woody plants die off, the upper limit for herbaceous wetland species is moved upslope and the total area of herbaceous wetland can increase (Keddy and Reznicek 1982). With time, emergent plants respond to the new water levels and form new communities according to their preferred hydrologic conditions. However, this transformation may not occur if upper limits are determined by anthropogenic barriers (e.g., dikes) rather than woody species (Gottgens 2000).

These cycles of water-level changes and plant response are repeated over and over again unless the cycle is broken by anthropogenic factors (e.g., water-level stabilization through regulation; Wilcox 2004), invasive species able to survive a wide-range of hydrologic conditions (Saltonstall 2002), damage to the seed bank (e.g., burned, eroded), or extensive herbivory. Degradation or destruction of the wetland plant communities often occurs if the cycle is disrupted. For example, extended high water levels in Lake Erie and constructed earthen dikes on upslope edges have contributed to the degradation of coastal wetland plant assemblages (Sherman et al. 1996, Kowalski and Wilcox 1999, Gottgens 2000, Kowalski et al. 2006). These wetlands likely will remain in a degraded condition until water levels decrease or resource managers take action to promote plant reestablishment. Since the number of coastal wetlands providing critical ecological functions (e.g., fish and wildlife habitat, nutrient uptake, wave attenuation) has decreased in the Great Lakes (Herdendorf 1987, Mitsch and Wang 2000), those remaining are a high priority for most management agencies.

Many methods to reestablish emergent plant assemblages are available, including direct planting and vegetation mats and logs, but most are expensive, labor-intensive, and difficult to implement over large areas (Kadlec and Wentz 1974, Wilcox and Whillans 1999). Furthermore, the hydrologic conditions that contributed to the initial degradation of plant assemblages often continue to make large-scale reestablishment difficult. Thus, there is a need for a means to induce localized low-water conditions temporarily where continuous submergence suppresses normal seed bank germination and plant reestablishment. Permanent solutions, such as installing earthen dikes to isolate the wetlands hydrologically and gain water-level control, have a proven track record but are expensive, require regulatory approval, and can have significant negative impacts on the ecology of coastal wetlands (Johnson et al. 1997, Mitsch et al. 2001, Herrick et al. 2007). Although not without challenges of their own, temporary solutions (e.g., portable, water-filled cofferdams) can have many advantages over permanent solutions, including lower cost, reusable material, less adverse environmental impact, and removal after management objectives are met. Portable cofferdams are available commercially in many shapes and sizes and are capable of making a tight but temporary seal with whatever substrate they rest on and preventing water movement into or out of target areas. The dams are removed after project completion. Portable, water-filled cofferdams are commonly used for construction, river diversion, or flood protection purposes but also have application for ecological rehabilitation projects.

This project explored the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural growth of emergent vegetation from the seed bank in a Lake Erie coastal wetland. These types of cofferdams have rarely been used to restore wetland habitat. The objectives of this project, therefore, were to evaluate how well portable, water-filled cofferdams temporarily isolate a portion of a wetland and to characterize the wetland seed-bank response at low, middle, and high elevations in areas open to and protected from bird and mammal herbivory.

2.3 Methods

2.3.1 Study Area

This study focused on the approximately 345-ha Crane Creek drowned-river-mouth wetland located within the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge (ONWR; 41.628611, -83.207778) along the southern shore of western Lake Erie approximately 30 kilometers east of Toledo, Ohio, USA (Figure 2.1). Earthen dikes and rock revetment bound the wetland on all sides except where Crane Creek enters from the west and exits through a channel to Lake Erie on the eastern boundary. Water levels in the wetland are primarily determined by inter-annual and short-term fluctuations (seiches) in water levels of Lake Erie, but inputs from the approximately 146 km^2 Crane Creek watershed can magnify or reduce the effects of changes in Lake Erie water levels, especially after storm events (Kowalski et al. 2006). Open water less than 1 m deep covered much of the wetland in 2003, but short, periodic exposure of mudflats by extreme seiche events combined with high turbidity ensured submersed aquatic vegetation was sparse (Kowalski et al. 2006). Emergent wetland vegetation dominated by *Typha angustifolia* (Narrow Leaved Cattail) and *Phragmites australis* (Common Reed) was growing around the perimeter of the marsh, with floating-leaf assemblages of *Nelumbo lutea* (American Lotus) and *Potamogeton nodosus* (Longleaf Pondweed) extending further from shore. Surrounding earthen dikes and other upland areas supported woody plants, including *Salix* spp. (Willow) and *Populus deltoides* (Eastern Cottonwood). A rich seed bank existed in the approximately 30 cm of silty sediments that overlay hard pan clay (Barry et al. 2004). Very few logs, rocks, or other debris disrupted the nearly uniform sediment surface.

Historically part of the Great Black Swamp that extended from western Lake Erie southwestward to New Haven, Indiana (Kaatz 1955), most of the coastal marshes along this section of U.S. shore, including parts of the Crane Creek wetland complex, were isolated by earthen dikes in the early 1900s to protect them from Lake Erie’s wave energy (Herdendorf 1987) and promote their management as migratory waterfowl habitat (Campbell and Gavin 1995). High quality waterfowl habitat remains a priority focus for many managers, but managing coastal and diked wetland habitats for other waterbirds, fish, amphibians, reptiles, and other biota is especially important to the ONWR managers. Armored shoreline and other anthropogenic forces, coupled with frequent high Lake Erie water levels since the early 1970s, contributed to reduction in the area and diversity of coastal wetland vegetation (Kowalski and Wilcox 1999). These degraded conditions remain because water levels have not dropped low enough during the growing season to expose the seed bank and allow emergent plants to reestablish (NOAA 2006). Normally, the annual high water levels occur in June and the lowest levels occur in February (NOAA 2006), but short-term wind tides or seiche fluctuations of up to 3 m above low water datum are common throughout the year (Herdendorf 1987).

2.3.2 Portable Cofferdams

A series of AquaDams® (i.e., portable, water-filled cofferdams manufactured by Water Structures Unlimited in Carlotta, California, USA) approximately 450 m long was installed temporarily to isolate a 10-ha corner of the Crane Creek wetlands from Lake Erie (see Figure 2.1). Conducting this study on a small section of the whole wetland prior to a large-scale implementation of cofferdam technology maximized the likelihood of achieving research objectives and ensured efficient use of resources. Installation of the first set of 1.8-m high cofferdams began on 19 April 2004 and was completed on 21 April 2004. During the installation, damage to one of the dams resulted in the need for additional dam material to fill a gap between dam sections. New dams were added to the site periodically 8 June 2004 – 25 June 2004 to achieve hydrologic isolation of the test site. Dewatering of the site was achieved by the second full week of July and maintained until the test site was flooded when sections of the cofferdam were washed into the dewatered area on 17 September 2004. Cofferdam material was removed the week of 14 October 2004.

The elevation of the substrate where the dams would be installed was surveyed using laser-plane surveying equipment, and historical water levels in Lake Erie were used to estimate the maximum normal water depth during the study. AquaDams® can range in height from less than a meter to over 4.8 m and are designed to operate in areas where the depth of the water being contained or diverted is less than approximately 70% of the dam height. Per the manufacturer's recommendation, six approximately 70-m-long sections of 1.8-m-high and 4-m-wide cofferdam were filled with water and linked together end-to-end to isolate the test area. In response to problems during the manufacturer's installation and first weeks of operation, additional 1.8-m-high and smaller 76-cm-high auxiliary support dams were added parallel to and on the dewatered side of the larger dams to create the final dam configuration shown in Figure 2.1.

Using a diesel-powered pump with 30.5-cm-diameter hoses, the water behind the dams was drawn down to an elevation that fully exposed the majority of the marsh sediments, similar to a natural low-water year. Standard dam maintenance was performed and pumping occurred regularly to maintain moist-soil conditions in the test area from the initial drawdown in July 2004 through premature failure of the dams in

September 2004 (Kowalski et al. 2006). As a result of the failure, all of the dams and maintenance equipment were removed from the site in October 2004 rather than in the fall of 2005 as intended.

Sediment elevation measurements were made after the dewatering was complete to characterize the topography of the dewatered area. A total station, laser-plane surveying equipment, and standard land-surveying methods were used to collect and tie sediment surface elevation data to a first-order U.S. Geological Survey benchmark. Since there were small differences in sediment surface elevation in the dewatered area, the surveying equipment was used to identify the boundaries of three major elevation zones (i.e., low, mid, high). Measured elevations ranged from 173.70 m to 173.93 m. All vegetation sampling in the low zone occurred at elevations less than 173.78 m. Sampling in the mid zone occurred between 173.80 m and 173.86 m, and sampling in the high zone occurred at elevations greater than 173.88 m. All elevations are reported with reference to the International Great Lakes Datum 1985.

Since bird and mammal herbivory of young plants can significantly influence seed bank driven revegetation of a wetland (Lynch et al. 1947, Barry et al. 2004), thirty 2 m x 2 m herbivory exclosures were built and placed in the dewatered area behind the cofferdams after dam installation. The exclosures (i.e., poultry wire strung around and over four metal posts at least 1.5 m high) allowed analysis of the effects of herbivory in a recently dewatered area when compared to data collected inside the exclosures. Ten exclosures were placed randomly in each of the three elevation zones.

2.3.3 Sampling and Analysis

The vegetation in the 30 exclosures was sampled quantitatively using a 1 m x 1 m quadrat centered in each exclosure prior to flooding in September 2004 and again in August 2005, approximately 11 months after the site was hydrologically reconnected to Lake Erie. During the same time periods, 10 open (i.e., unprotected from herbivory) quadrats were placed randomly outside the exclosures but within each of the three elevation zones in the dewatered area. Therefore, a total of 60 quadrats were sampled in the dewatered area each year. For this analysis, quadrats in each combination of elevation (e.g., low, mid, high) and herbivory protection (e.g., exclosure, open) were considered a sampling group. Additional quadrats were sampled in nearby areas of Crane Creek that were at elevations similar to the dewatered area yet remained under the hydrologic influence of Lake Erie. These reference quadrats were considered a separate sampling group for each year. Plant species found in all quadrats were identified and assigned a percent cover value using visual estimation. Investigators regularly estimated percent cover values in test plots to minimize differences among sampling teams. No sampling was done prior to fall 2004 because air photo interpretation and site visits revealed very little wetland vegetation in the study site, excluding fringe stands of *T. angustifolia*, *P. australis*, and *N. lutea* (Kowalski et al. 2006). Herbaceous plant nomenclature followed Flora of North America (www.eFloras.org) and tree nomenclature followed Barnes and Wagner (2004).

Plant species richness (i.e., number of taxa) and importance values (i.e., sum of relative frequency and relative cover of each taxon in a sampling group; Curtis and McIntosh 1951) were calculated using data collected during quadrat sampling. The importance values were analyzed using non-metric multidimensional scaling (NMDS) to explore differences associated with herbivory (i.e., exclosure and open to herbivory) and low, mid, and high elevation zones (McCune and Grace 2002). The analysis was performed using the PC-ORD version 5.1 with the Bray-Curtis distance measure (Bray and Curtis 1957, McCune and Mefford 2006). Dimensionality of the data set was determined by using a random starting number, 250 runs with real data, 250 runs with randomized data, and 500 maximum iterations. The analysis was repeated with only the recommended number of dimensions (i.e., three) and without the Monte Carlo test.

2.4 Results

The nearly two-month drawdown maintained by the cofferdams produced a rapid and diverse response from the seed bank that was not observed in the reference plots. Thirty-nine of the forty-two plant and alga taxa found during the 2004 sampling were identifiable to species (Table 2.1). Thirty of those taxa were emergent herbaceous or woody species. Even though they were found at elevations similar to the plots in the dewatered area, all taxa sampled in the 2004 reference group were submersed aquatic or floating-leaf species except *Eleocharis acicularis* (Needle Spike Rush) and *N. lutea*.

Three of the six submersed aquatic taxa found in the reference group (i.e., *Ceratophyllum demersum* (Coontail), *Myriophyllum sibiricum* (American Watermilfoil), *Vallisneria americana* (Eel Grass)) were not found anywhere in the dewatered area. The alga taxa sampled in 2004 were not identifiable to species.

A different suite of eighteen taxa were sampled under the flooded conditions in 2005. All of the woody taxa found in 2004 were absent in 2005, and only three of the fifteen taxa identifiable to species (Table 2.1) were not submersed aquatic or floating-leaf species (i.e., *Butomus umbellatus* (Flowering Rush), *N. lutea*, *Pontederia cordata* (Pickerelweed)). Total species richness among the sampling years and groups ranged from the least (5 taxa) in the 2005 low-elevation exclosure and high-elevation open sites to the greatest (27 taxa) in the 2004 high open site (Table 2.2). The average species richness among the 2004 sampling groups (19.6 species) was more than double the 2005 sampling groups (7.1 species).

Differences among sampled groups and years were apparent when NMDS was used to analyze the importance value data. The data best fit a 3-dimensional model, but only axis 1 and axis 3 are shown because they accounted for most of the variation (Figures 2.2a and 2.2b). There was a clear separation of groups based on the degree of flooding along axis one, which explained 57.4% of the variation. The mudflat assemblages found during the 2004 dewatered conditions were tightly grouped toward the left side of axis 1, while the submersed aquatic-dominated assemblages found in the 2004 reference plots and all of the 2005 plots were grouped toward the right side of axis 1 (Figure 2.2a). For both years, there was a pattern of separation among the low, mid, and high zones along axis 3 that explained 24.4% of the variation (see Figure 2.2a). The 2004 and 2005 reference data grouped with the 2005 high elevation data dominated by submersed aquatic species adapted to flooded conditions (Figure 2.2b). There was no discrimination between the open or exclosure groups among the zones, but the presence of the emergent invasive species *B. umbellatus* in the 2005 low elevation open group contributed to its separation from the 2005 low elevation exclosure group. An additional 10.0% of the variation was explained by the second axis (not shown), although no ecological groups or patterns were apparent along that axis.

The importance values for the individual taxa sampled in 2004 revealed few differences in the dominant species (i.e., those with the five highest importance values) among all of the elevation zones except the presence of *Schoenoplectus tabernaemontani* (Softstem Bulrush) in the low exclosure and *P. nodosus* in the high elevation open and exclosure quadrats (Table 2.2). No *Schoenoplectus* was found in the low open quadrats. The *Potamogeton* spp. were rooted prior to the drawdown and survived on the wet mudflats. *Cyperus erythrorhizos* (Red Rooted Flatsedge) and other classic mudflat taxa were common among all 2004 sampling groups, which contributed to the high (i.e., 17 – 27 taxa) species richness in 2004. The species richness dropped significantly to a range of 5 – 8 taxa per sampling group by 2005 after cofferdam failure. A suite of *Potamogeton* species replaced most of the emergent species, and *Najas minor* (Brittle Waternymph) became much more dominant. Except for the presence of *P. nodosus* in the high open group, there were no clear differences in the composition of samples taken inside and outside of the exclosures.

2.5 Discussion

The loss of emergent vegetation in Great Lakes coastal wetlands during high water levels is part of the cycle of destruction and renewal caused by naturally fluctuating water levels (Keddy and Reznicek 1985). Subsequent low water levels during the growing season expose the seed-rich sediments and promote the natural regeneration of wetland plants. If anthropogenic disturbance (e.g., altered hydrology) or extended high water levels coupled with upslope backstopping (Gottgens 2000) prevent exposure of the sediments, then the wetlands remain in a degraded state until water levels recede naturally or management actions are employed to restart the cycle. Water-filled, portable cofferdams are one of many technologies currently available to separate a section of river, lake, or wetland hydrologically from its parent waterbody. Unlike cofferdams with a rigid design made out of plastic or other materials, soft-bodied dams (i.e., geotextile material wrapped around a seamless liner) like the Aquadam® used in this study are flexible enough to mold around irregularities in sediments and make a water-tight seal with the bottom. This temporary seal allows managers to conduct a drawdown that mimics conditions found during a low water year. If a viable seed bank exists in the marsh, then simply exposing the sediments elicits a positive response from the seed bank. However, this response is short-lived and habitat is not reestablished if dewatered conditions are not maintained long enough to allow the plants to mature. Unlike earthen dikes, the footprints of these portable cofferdams have minimal ecological impact (e.g., sediment disruption) and can be removed from the marsh after plants reestablish or management objectives have been met. Experiences during this study, however, revealed that a significant amount of effort (e.g., planning, installation, maintenance) is required to maximize the likelihood of maintaining dewatered conditions long enough to meet project objectives and technological improvements are needed to make these dams viable for extensive habitat restoration projects.

2.5.1 Maintaining Dewatered Conditions

Although the portable, water-filled cofferdams used in this project only maintained dewatered conditions for a short time, lessons were learned that can be used to improve future deployments in Great Lakes coastal wetlands (see Appendix A for additional details). We found that selection and preparation of the study site is very important to establish dewatered conditions and maximize response from the seed bank. Optimal installation sites will have a reliable source of clean water to pump into the dams, easy access by people and heavy equipment, a limited amount of rocks, trees, or other debris in the sediments under the cofferdams, and a rich seed bank in the area to be dewatered. In addition to site selection, we found that using a product sized appropriately for the application is critical for maintaining dewatered conditions long enough to allow seedlings to reach maturity. Undersized cofferdams are vulnerable to being overtopped by high water levels or undermined by erosion, water seepage, or wildlife activities while oversized dams are more expensive and may be more difficult to install and maintain. Although water depth is the most important factor to consider when selecting dam size (Water Structures Unlimited 2004), there are many other factors that can influence cofferdam performance including installation and maintenance.

It is essential to have the proper equipment (see Appendix B) on site during installation and maintenance of the dams to prevent delays and additional expenditures. In addition, problems encountered during installation must be fixed immediately, and any damaged dams must be replaced rather than repaired to minimize the chance of later problems. Once the cofferdams are installed and filled with water, regular and often labor-intensive maintenance activities are required to keep the dams full and to maintain dewatered conditions at the study site.

2.5.2 Wetland Plant Growth From the Seed Bank

Moist-soil conditions were maintained in our study site for about two months. During these two months, the cofferdams effectively created conditions for seed-bank derived growth of emergent wetland vegetation. Shortly after the seed bank was exposed in July 2004, seeds from over 40 different taxa began to germinate, as they likely would have during a low-water year (Keddy and Reznicek 1985; see Table 2.1). Previous studies found an extensive seed bank in Crane Creek and neighboring coastal marshes (Wilcox and Kowalski 1995, Davis and Welch 2000, Barry et al. 2004), but areas that have not been vegetated for a long time or have been eroded by waves may have a severely diminished seed bank.

Most of the plants growing in the dewatered area of Crane Creek were mudflat wetland species with seeds that remain dormant but viable in the seed bank for a long time. However, there were some plants that likely came from seeds transported to the recently dewatered sediments via wind or other vectors. *Salix* spp. and *P. deltoides*, for example, are woody taxa that often become densely established in wetlands when sediments are exposed. If sufficient sources are available, wind-dispersed seeds land in fertile wetland sediment and quickly germinate. Unlike in a neighboring coastal marsh (Kowalski and Wilcox 1999), these woody species were not a large component of the plant assemblages growing among the elevation zones within the dewatered area (see Table 2.2) because the marsh was not fully dewatered until July. Most *Salix* and *Populus* species flower and produce seeds in late spring or early summer (June for western Lake Erie). The drawdown occurred after most of these woody species should have reproduced (Chadde 2002), so their seeds likely had already been distributed by the wind. The woody seedlings that did grow during the drawdown were not able to survive the flooding after the cofferdams failed and were removed, so the timing of the drawdown and subsequent flooding were important in preventing invasion by woody species. The absence of woody species growing at the reference sites both during and after the management drawdown suggests that the dewatering action allowed the temporary growth of woody species but flooded conditions were not conducive to their establishment or growth. Management-driven drawdowns often are conducted later in the growing season to minimize the establishment of woody species and promote a greater diversity of wetland species (Fredrickson and Taylor 1982). Late season drawdowns also can be used to target the growth of certain emergent and submersed aquatic plant species for waterbirds (Keith 1961, Payne 1992), although certain plants established late in the season can become management problems in subsequent years (Meeks 1969).

In addition to the timing of a management drawdown, small but ecologically important differences in elevation of the marsh sediments can influence species richness and composition (see Figure 2.2b). Ordinations revealed similarities among taxa collected at each elevation zone as well as dissimilarity among the elevation zones. The NMDS-derived groupings (i.e., circles in Figure 2.2a) show a pattern among the sampling groups along axis 3, with low elevation sampling groups having the largest axis 3 values and the high sampling groups with the smallest values. This pattern is apparent in both the 2004 data and the 2005 data, with the exception of the 2005 low open sampling group (LO_5). This group is an outlier because the invasive species *B. umbellatus* was present. *Butomus umbellatus* is an aggressive perennial herb that establishes quickly and can persist in flooded conditions (Hroudova et al. 1996). The LO_5 sampling group was the only one where *B. umbellatus* had a high importance value, so it plotted closer to the groups composed of emergent taxa. The reference sampling groups for both 2004 and 2005 grouped close to the 2005 high elevation data in the NMDS because they were located at similar elevations and there was a strong presence of submersed aquatic species. The reference groups did not receive the dewatering treatment, and their species composition did not differ much between the two years, so we are confident that the significant differences observed in the dewatered area were the result of the hydrologic changes associated with the 2004 drawdown treatment and subsequent reflooding in 2005. The observed differences in plant assemblages associated with each elevation in 2004 likely are tied to differences in soil moisture during germination suggesting that even small elevation differences in dewatered sediments can affect the seed bank germination success and ultimately the composition of plant assemblages. In contrast, the 2005 data suggest that, when flooded, only relatively large differences in water depth (and therefore light availability) associated with each elevation zone influence species presence.

Although grazing of wetland plant seedlings can be a management problem, this study did not detect a strong overall effect of herbivory on the species richness of wetland plants growing in the dewatered area. However, some plant species only occurred in the plots protected from herbivory, while others only grew in plots open to the full effects of herbivory. For example, *S. tabernaemontani* had the greatest importance value for the 2004 low exclosure data but unexpected did not appear at any of the low elevation areas not protected by exclosures. Five other species (*E. acicularis, Polygonum lapathifolium* (Nodding Smartweed), *P. cordata, Salix exigua* (Sandbar Willow), *T. angustifolia*) also had high importance values only in the protected sample sites. Conversely, only two species (*Eleocharis obtusa* (Blunt Spike Rush), *Najas marina* (Spiny Naiad)) had high importance values in the open sites. These results could be in response to many factors (e.g., synchronicity between waterbird migrations and seedling growth, herbivore disturbance by the presence of the cofferdams, a seed bank with high diversity and variation in density), but the absence of a strong pattern suggests that plant herbivory may be present at a site without impacting the composition of developing plant assemblages.

Regardless of protection from herbivory, the species richness was high during the 2004 drawdown in the low, mid, and high elevation zones. The low elevation zone had fewer taxa than the other zones, likely because the sediments in much of this zone remained saturated or in some places were covered by very shallow water. This zone was dry immediately after the drawdown began, but water channeling under a dam flowed over this zone throughout the project and likely prevented some emergent plants from germinating. Where present, the shallow surface water supported submersed aquatic taxa (e.g., *Potamogeton* spp.) common in the reference sampling group but generally absent from the higher elevation zones of the dewatered area. Similarly, the 2004 reference sampling group and all of the 2005 sampling groups remained inundated and, as a result, had many fewer species.

The plant assemblage changed dramatically after the cofferdams failed and the hydrologic connection to Lake Erie was restored to the site in late 2004, when much of the test area was covered by over 71 cm of water (see axis 1 values in Figure 2.2). Although off to a good start, most emergent species had not grown tall enough during the brief drawdown to survive inundation by the late-summer high water levels. These emergent plants were replaced in 2005 by a suite of submersed aquatic species that tend to thrive in deeper water. A similar suite of species was found in other parts of Crane Creek that did not receive the dewatering treatment, so it appears that the post-cofferdam reflooding promoted the quick return of pre-drawdown submersed aquatic plant assemblages. If the sediments had been exposed during a time of low water-levels in Lake Erie, emergent plants likely would have had one or more growing seasons to reach maturity. The height advantage achieve by many plants at maturity would allow them to survive higher water-levels, as aerenchyma tissue could reach atmospheric oxygen, and the benefits of increased wetland habitat would last longer. Not surprisingly, the length of time that the marsh seed bank is exposed is critical to the longevity of seed-bank-driven plant growth in Great Lakes coastal marshes.

2.5.3 Implications for Large-Scale Habitat Rehabilitation

The intent of this study was to test a novel technology that created temporarily dewatered conditions in a section of coastal marsh to allow wetland plants to grow from the seed bank. The study revealed both the potential benefits of applying this management tool in coastal wetlands and a number of challenges that must be addressed prior to large-scale implementation. Understanding the operation and technical details of the cofferdam technology is critical in determining how to maximize the response from the seed bank and promote the long-term survival of emergent plants (i.e., habitat rehabilitation). Many significant problems were identified during tests of early designs during the studies performed in Lake Ontario coastal wetlands (i.e., Cootes Paradise) in the early 1990s (Wilcox and Whillans 1999). Vandalism and product design issues proved to be the biggest challenges that prevented large-scale implementation in Cootes

Paradise and in Crane Creek (see Appendix A). Although a different suite of challenges arose during the test at Crane Creek, our limited results show that this tool can be used to isolate portions of a coastal marsh temporarily and promote plant growth. However, the extent and longevity of that growth depends on the length of time that dewatered conditions are maintained and the hydrologic conditions present once the dams are removed. A tool like this is of particular interest to managers of highly-degraded coastal wetland habitats because it has the potential to provide the benefits of hydrologic isolation without causing long-term damage to wetland sediments or permanently altering the hydrology.

Advancements in the technology and the implementation process will continue to improve the odds of successfully achieving research and management objectives in similar wetland habitat rehabilitation projects throughout the Great Lakes. Although whole wetland complexes may not be able to be rehabilitated at once, these relatively small-scale habitat rehabilitation projects can provide localized benefit to the system and, in aggregate, improve the habitat available to Great Lakes biota. The temporary and highly customizable (e.g., height, length) design of portable cofferdams also supports their repeated use in one area over time or in multiple areas within a wetland. This technology, therefore, can be a potentially important tool in the arsenal used by Great Lakes resource managers.

2.6 Acknowledgments

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Table 2.1. List of plant species collected in Crane Creek in 2004 and 2005. Code lists the abbreviations used in Figure 2b. Form is designated as emergent (E), submersed aquatic (S), or other (O). "X" indicates present. "*" indicates found in reference plots and "" indicates only found in reference plots. Table only includes taxa identifiable to species.**

Species	Code	Form	2004	2005
Abutilon theophrasti Medikus (Velvetleaf)	ABUTHE	E	X	
Ammannia robusta Heer & Regel (Grand Redstem)	AMMROB	E	X	
Butomus umbellatus L. (Flowering Rush)	BUTUMB	E		X
Ceratophyllum demersum L. (Coontail)	CERDEM	S	X**	X*
Cyperus bipartitus Torr. (Shining Flatsedge)	CYPBIP	E	X	
Cyperus diandrus Torr. (Umbrella Flatsedge)	CYPDIA	E	X	
Cyperus erythrorhizos Muhl. (Red Rooted Flatsedge)	CYPERY	E	X	
Cyperus odoratus L. (Rusty Flatsedge)	CYPODO	E	X	
Eleocharis acicularis (L.) R. & S. (Needle Spike Rush)	ELEACI	E	X*	
Eleocharis obtusa (Willd.) Schultes (Blunt Spike Rush)	ELEOBT	E	X	
Eragrostis hypnoides (Lam.) BSP (Creeping Lovegrass)	ERAHYP	E	X	
Heteranthera dubia (Jacq.) MacM. (Grassleaf Mudplantain)	HETDUB	S		X
Hibiscus trionum L. (Rosemallow)	HIBTRI	E	X	
Lactuca serriola L. (Prickly Lettuce)	LACSER	E	X	
Lemna minor L. (Common Duckweed)	LEMMIN	O	X*	X*
Myriophyllum sibiricum Komarov (American Watermilfoil)	MYRSIB	S	X**	
Myriophyllum spicatum L. (Eurasian Watermilfoil)	MYRSPI	S		X
Najas marina L. (Spiny Naiad)	NAJMAR	S	X	X
Najas minor Allioni. (Brittle Waternymph)	NAJMIN	S		X*
Nelumbo lutea Wildenow (American Lotus)	NELLUT	E	X*	X
Penthorum sedoides L. (Ditch Stonecrop)	PENSED	E	X	
Phalaris arundinacea L. (Reed Canarygrass)	PHAARU	E	X	
Phragmites australis (Cav.) Steudel (Common Reed)	PHRAUS	E	X	
Polygonum lapathifolium L. (Nodding Smartweed)	POLLAP	E	X	
Polygonum pensylvanicum L. (Pennsylvania Smartweed)	POLPEN	E	X	
Pontederia cordata L. (Pickerelweed)	PONCOR	E	X	X
Populus deltoides Marshall (Eastern Cottonwood)	POPDEL	O	X	
Potamogeton crispus L. (Curled Pondweed)	POTCRI	S		X*
Potamogeton foliosus Raf. (Leafy Pondweed)	POTFOL	S	X*	X*
Potamogeton nodosus Poiret. (Longleaf Pondweed)	POTNOD	S	X*	X*
Potamogeton pectinatus L. (Sago Pondweed)	POTPEC	S	X*	X*
Potamogeton richardsonii (Benn.) Rydb. (Redhead Pondweed)	POTRIC	S		X
Rhus hirta (L.) Sudworth (Staghorn Sumac)	RHUHIR	O	X	
Riccia fluitans L. (Crystalwort)	RICFLU	S	X	
Rorippa palustris (L.) Besser (Common Yellowcress)	RORPAL	E	X	
Rumex crispus L. (Curly Dock)	RUMCRI	E	X	
Sagittaria latifolia Willd. (Duck Potato)	SAGLAT	E	X	
Salix cordata Michx. (Heartleaf Willow)	SALCOR	O	X	
Salix eriocephala Michx. (Missouri Willow)	SALERI	O	X	
Salix exigua Nutt. (Sandbar Willow)	SALEXI	O	X	
Salix fragilis L. (Crack Willow)	SALFRA	O	X	
Schoenoplectus tabernaemontani (C.C. Gmelin) Palla (Softstem Bulrush)	SCHTAB	E	X	
Scirpus fluviatilis (Torr.) A. Gray (River Bulrush)	SCIFLU	E	X	
Typha angustifolia L. (Narrow Leaved Cattail)	TYPANG	E	X	
Vallisneria americana L. (Eel Grass)	VALAME	S	X**	X**

Table 2.2. List of the plant and alga taxa with the top five importance values collected in the drawdown area behind the cofferdam in Crane Creek in 2004 and 2005. Missing values do not necessarily indicate the absence of taxa, because taxa might have importance values below the five highest values. Species richness of each sampling group is noted.

	Importance Value													
	Low				Mid				High				Reference	
	Open		Excl		Open		Excl		Open		Excl			
Taxa	**2004**	**2005**	**2004**	**2005**	**2004**	**2005**	**2004**	**2005**	**2004**	**2005**	**2004**	**2005**	**2004**	**2005**
Butomus umbellatus		20.5												
Ceratophyllum demersum												7.9		
Cyperus erythrorhizos	44.0		29.6		82.1		42.4		28.6		21.4			
Eleocharis acicularis							14.5		31.2		60.3		8.9	
Eleocharis obtusa	17.1													
Heteranthera dubia		30.7		19.2										
Lemna minor													33.6	25.7
Myriophyllum spicatum		23.3												
Najas marina	9.7			25.8				9.6						
Najas minor						103.7		86.6		47.2		119.4		22.9
Nelumbo lutea										7.6				
Nitella sp.										15.7		15.7		
Polygonum lapathifolium							18.0							
Pontederia cordata			23.3			15.9								
Potamogeton crispus						16.7								
Potamogeton foliosus				90.4										
Potamogeton nodosus		14.6				19.0		36.5	39.4	116.0	25.2	33.8	105.6	92.2
Potamogeton pectinatus		90.5		64.6		24.7		24.5		13.6		7.7	21.1	18.2
Potamogeton richardsonii				19.2				22.6						
Rumex crispus					28.4				12.5		12.9			
Sagittaria latifolia	41.7		29.1		8.8		14.9							
Salix cordata					7.3									
Salix eriocephala					7.3				9.3					
Salix exigua											10.8			
Schoenoplectus tabernaemontani			42.2											
Typha angustifolia	16.0		23.9				18.3							
Vallisneria americana													14.0	15.1
SPECIES RICHNESS	**19**	**7**	**17**	**5**	**21**	**7**	**23**	**8**	**27**	**5**	**22**	**8**	**9**	**8**

Figure 2.1 Maps and 2004 digital orthorectified photograph of Crane Creek study site. Thick white dashed lines indicate boundaries of Crane Creek. Approximate boundaries of elevation zones are noted with black dashed lines.

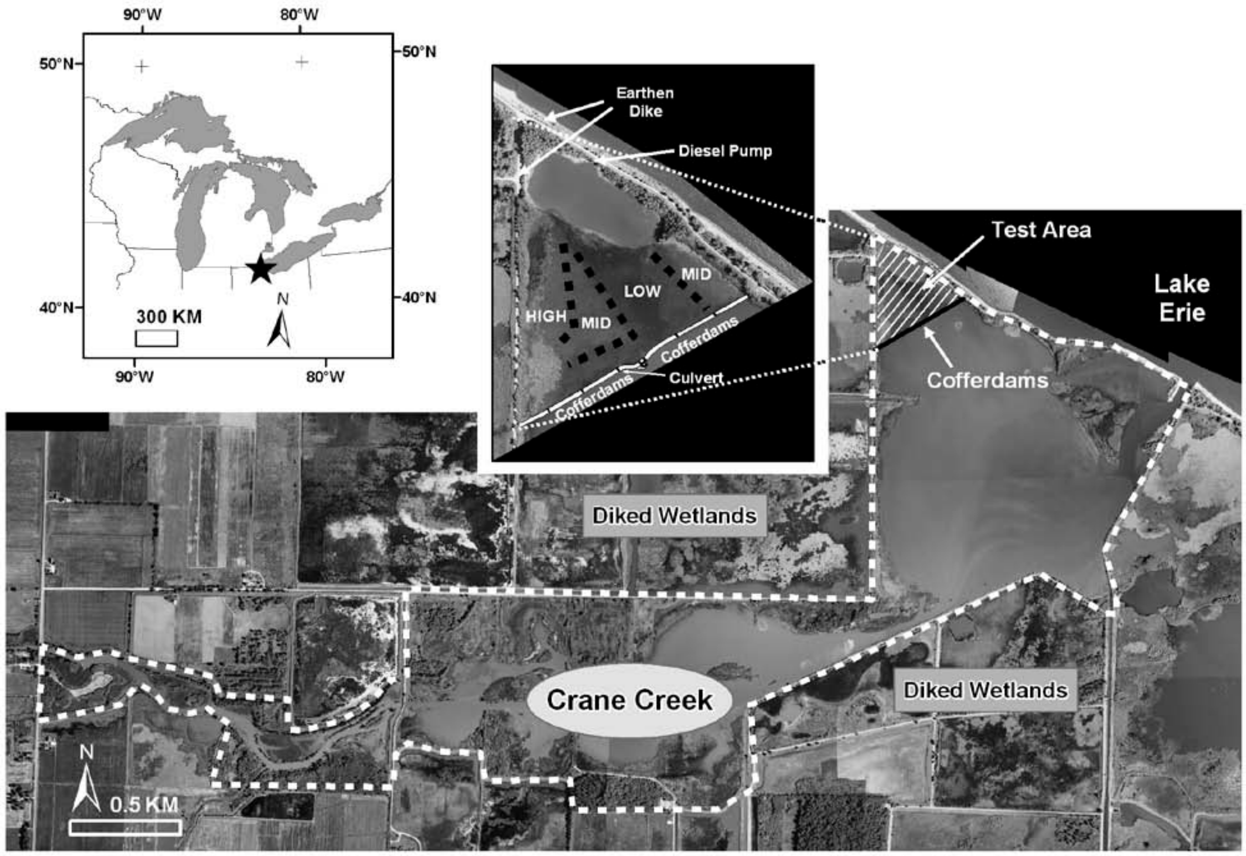
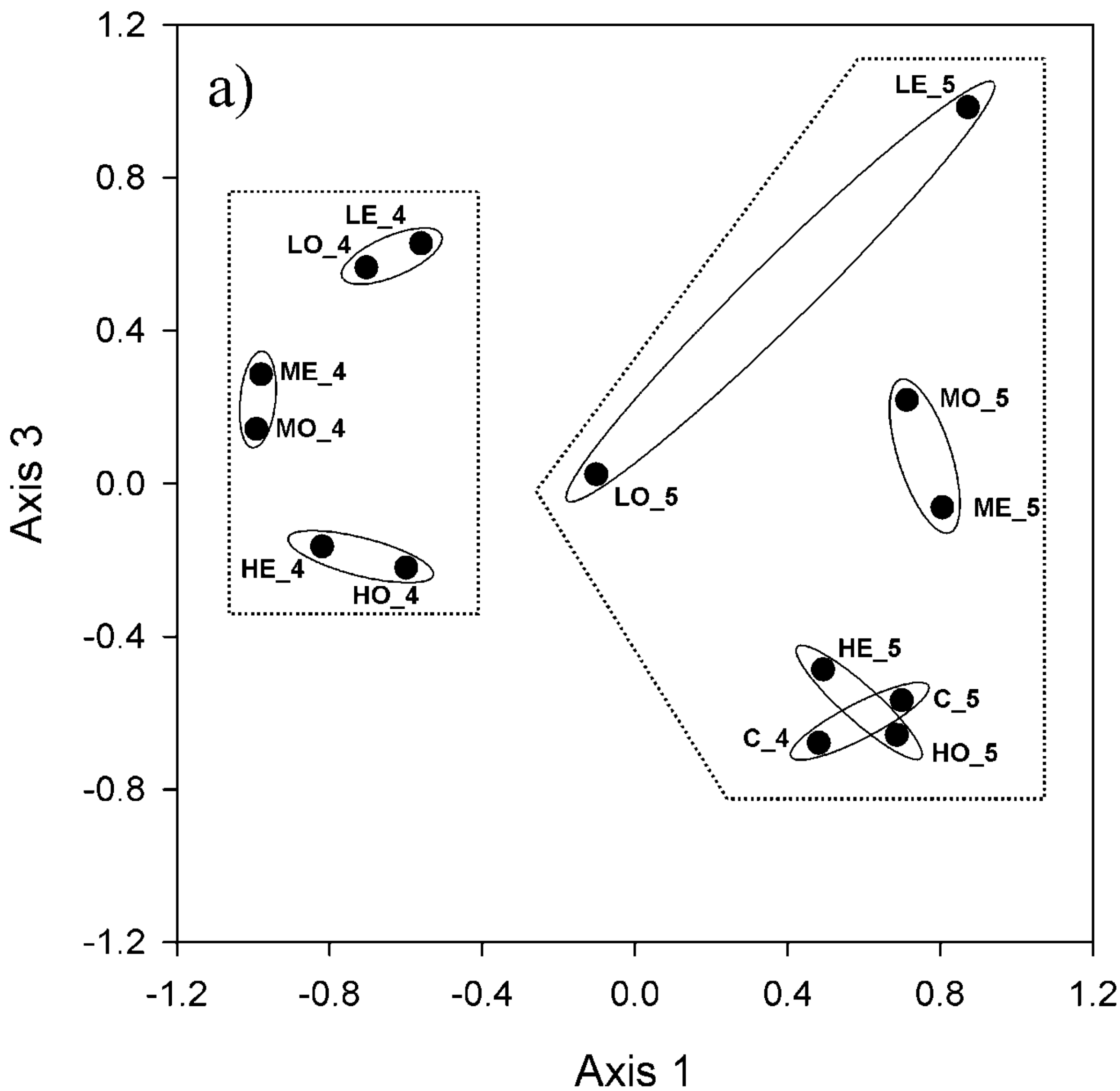
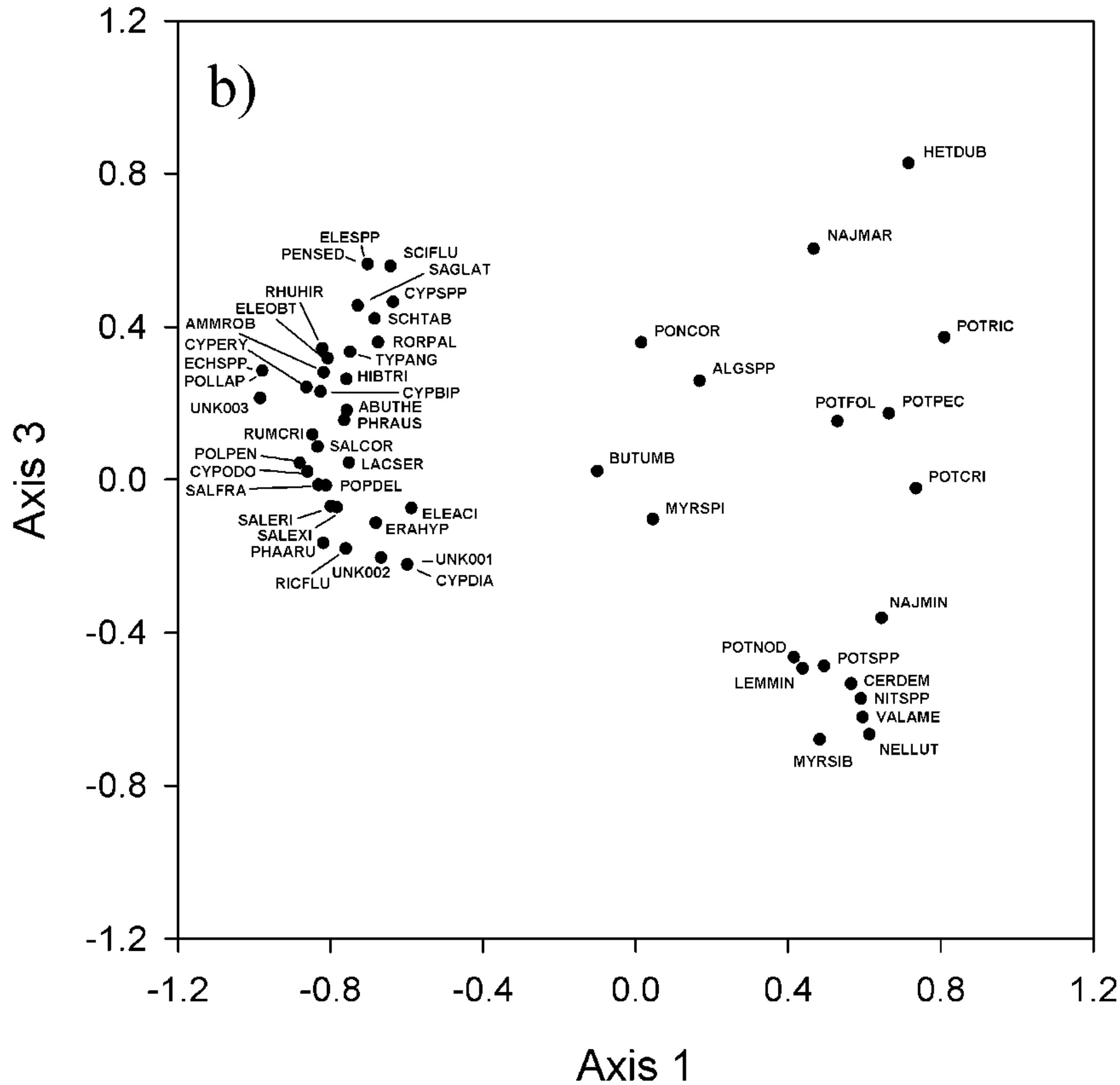


Figure 2.2 The first and third axes of the non-metric multidimensional scaling ordination, based on Importance Values calculated on fifty-four wetland plant and alga taxa collected in the sampling groups in 2004 and 2005. a) Ordination of sites identified by location in the elevation gradient (high (H), middle (M), low (L)); exclosure (E) or open (O); and year (2004, 2005). b) Ordination showing taxa with high importance values, labeled using the first three letters of the genus and first three letters of the species. Final 3-dimensional solution stress was 4.98820 after 99 iterations.




b)
Axis 3
Axis 1
1.2
0.8
0.4
0.0
-0.4
-0.8
-1.2
-1.2
-0.8
-0.4
0.0
0.4
0.8
1.2
HETDUB
NAJMAR
POTRIC
PONCOR
ALGSPP
POTFOL
POTPEC
BUTUMB
POTCRI
MYRSPI
NAJMIN
POTNOD
POTSPP
LEMMIN
CERDEM
NITSPP
VALAME
NELLUT
MYRSIB
ELESPP
PENSED
SCIFLU
SAGLAT
RHUHIR
CYPSPP
ELEOBT
SCHTAB
AMMROB
RORPAL
CYPERY
TYPANG
ECHSPP
HIBTRI
POLLAP
CYPBIP
UNK003
ABUTHE
PHRAUS
RUMCRI
SALCOR
POLPEN
LACSER
CYPODO
SALFRA
POPDEL
ELEACI
SALERI
ERAHYP
SALEXI
PHAARU
UNK001
UNK002
RICFLU
CYPDIA

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Chapter 3
Variations in fish populations related to hydrologic connectivity in a diked coastal wetland: implications for habitat rehabilitation

3.1 Abstract

Many Great Lakes fishes use coastal wetlands for feeding, spawning, or nursery habitat, however, the condition of and access to coastal wetlands have declined significantly. Fish and plant assemblages in the Crane Creek coastal and diked wetland complex (Lake Erie) were sampled quantitatively to characterize spatial and seasonal patterns of fish assemblages and examine the implications of habitat rehabilitation by hydrologically reconnecting diked wetlands to Lake Erie. Fyke netting captured fifty-three species and a large abundance of fishes in Crane Creek, but fewer than half of those species and a much smaller number of fish were captured in the adjacent diked wetlands. Although located adjacent to Lake Erie, there were pronounced differences in hydrology, fish assemblages, and wetland vegetation between the diked and coastal wetlands. Therefore, establishing a long-term hydrologic connection between diked and coastal wetlands in Lake Erie would allow fishes to use vegetated habitats seasonally. Periodic management actions involving hydrologic isolation of the diked wetlands could be used to mimic intermediate levels of disturbance and maintain wetland vegetation.

3.2 Introduction

Great Lakes coastal wetlands are believed to provide valuable habitat for a large variety of wetland-, river-, and lake-associated species of fish (Jude and Pappas 1992, Wei et al. 2004). In particular, many economically valuable fishes and forage fish species depend on these productive habitats to feed, spawn, or provide protection to young-of-year (YOY; Herdendorf 1987). However, the ecological condition of coastal wetlands in the Great Lakes region has declined significantly since intensive human development in the coastal zone and upland landscapes began accelerating over a century ago (Campbell and Gavin 1995). Today, both changes in water quality and extensive hydrologic modifications have negatively affected the majority of coastal, wetland, and ultimately the nearshore lake environs they border.

The glacial lake plain of northwestern Ohio once supported a large area of coastal wetlands which comprised the eastern edge of the Great Black Swamp (Kaatz 1955). However, greater than 96% of those wetland habitats along the U.S. shoreline of Lake Erie have been lost since the 1860s (Herdendorf 1987, Mitsch and Wang 2000), and most of the remaining wetlands along the shore have been hydrologically isolated by earthen dikes to protect them from wave attack and to promote intensive management as migratory bird habitat. Water levels in diked units generally are controlled directly by managers, so conditions can be maximized to promote growth of wetland plants, inhibit the growth of invasive species, minimize high turbidity, and provide optimal habitat structure for waterfowl, shorebirds, and muskrats. Although adjacent to the Lake Erie shoreline, these diked wetlands cannot provide many of the ecological functions of typical of coastal wetlands (e.g., migratory fish habitat, fluvial nutrient processing) and often are not even classified as coastal wetlands due to this hydrologic segregation from the lake (Keough et al. 1999, Albert et al. 2005, Simon and Stewart 2006).

On the other hand, hydrologic connection between Lake Erie and wetland habitats alone does not ensure that coastal wetlands will provide quality habitat for aquatic biota. Intensive land development for both urban and agricultural use has severely impacted the ecological condition of most Lake Erie tributary systems (Herdendorf 1987, Kowalski and Wilcox 1999, Kasat 2006), and the resulting quality of water delivered to most receiving wetlands is poor and heavily influenced by landscape export of nutrients and contaminants. These and other factors, including invasive species, watershed drainage, and armoring of adjacent Lake Erie shoreline, have contributed to the severe degradation of the few remaining undiked wetlands in this region (Herdendorf 1987, Maynard and Wilcox 1997, Kowalski and Wilcox 1999). While the undiked wetlands remain hydrologically connected to the lake, their water quality and wetland vegetation are often significantly degraded. Diked units today comprise a majority of the remaining Lake Erie wetland habitat, and likewise constitute the majority of holdings managed by state and federal refuge systems (e.g. diked units comprise approximately 80% of the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge).

At a time when efforts to rehabilitate the degraded coastal habitats of the Great Lakes are attracting unprecedented national investment (Great Lakes Restoration Initiative 2009), the need for careful, science-based evaluation and prioritization of restoration activities has never been greater. Habitat restoration activities often focus on planting native vegetation, controlling invasive species, improving water quality, removing contaminated sediments, or correcting altered hydrology. Lake Erie coastal wetlands undoubtedly require both extensive water quality and hydrologic rehabilitation. However, there is an interesting contradiction between potential gains in biological function from restored hydrologic connectivity and potential losses from reconnection to degraded river water quality, expanded exposure to invasive species, and loss of submersed aquatic vegetation. It is not clear how current patterns of biological use and productivity can guide decisions about the relative priority of hydrologic reconnection to rehabilitate coastal wetland habitats.

Our study examined current plant and fish assemblages in river-mouth and adjacent diked wetland units of a Lake Erie drowned-river-mouth wetland complex. Our goal was to compare spatial and seasonal patterns of biological composition. We were interested in the benefits and risks of restoring habitat for Great Lakes fish assemblages by reestablishing the surface connection between diked wetlands and Lake Erie.

3.3 Materials and Methods

3.3.1 Study Area

We studied four sites within the Crane Creek drowned-river-mouth wetland complex managed by the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge (ONWR; 41.628611° Latitude, -83.207778° Longitude) along the southern shore of western Lake Erie approximately 48 kilometers southeast of Toledo, Ohio, USA (Figure 3.1). Crane Creek flows slowly into the >370-ha wetland complex from the west and exits through a channel to Lake Erie on the eastern boundary. The permanently open channel between the wetland and Lake Erie is approximately 100 m long and 50 m wide, with a variable depth that can exceed 4 m in localized areas. Bounding earthen dikes built in the early 1900s constrict the channel approximately 1.7 km upstream from the junction with Lake Erie. We considered this constriction as the boundary between the upper (CCUp; 210 ha) and lower (CCLow; 160 ha) Crane Creek study sites.

As with other drowned-river mouth wetlands, water-level fluctuations in Lake Erie drive the water levels in the adjoining coastal wetlands (Keough et al. 1999). Annual water levels in Lake Erie fluctuate greatly depending on water supply and climate, but short-term, wind-driven water-level oscillations (i.e., seiches) also occur, most often have an amplitude between 0.7 m and 2 m, and can exceed 3 m during storm events (Herdendorf 1987). The average gradient through the approximately 146 km^2 Crane Creek watershed is 0.359 m/km (Ohio Department of Transportation 1987), and water velocities normally are low except in the channel to Lake Erie, where the velocity of water moving from Crane Creek into Lake Erie can exceed 1 m/s during large seiche events (K. Kowalski, unpublished data, 2006). Large nutrient loads from agricultural and point-source discharges in the watershed contribute to poor water quality (e.g., high concentrations of nitrate, ammonia, and soluble reactive phosphorus) in Crane Creek as it reaches the influence of Lake Erie (Kasat 2006). Water depth in most of the wetland was less than 1 m deep during this study, but turbidity was high and submersed aquatic vegetation was sparse (K. Kowalski, unpublished data, 2006).

Earthen dikes and rock revetment comprise most of the wetland boundaries, but robust exotic emergent wetland plants (e.g., *Typha angustifolia*, *Phragmites australis*) populate the perimeter of the marsh, and floating-leaf assemblages of *Nelumbo lutea* and *Potamogeton nodosus* extend further from shore. Earthen dikes and other upland areas adjacent to the study sites support woody species, including *Salix* spp. and *Populus deltoides*. Deep silty sediments, often with abundant seed banks (Barry et al. 2004, Kowalski et al. 2009), cover most of the wetland except in a few areas near Lake Erie, where greater water velocities expose sand and a hard pan bottom (Bowers 2003).

In addition to the current riverine wetlands in the upper and lower Crane Creek study sites, this project focused on two diked wetland units adjacent to Crane Creek (see Figure 3.1). Pool 2A (28 ha) and Pool 2B (40 ha) are diked units that have remained hydrologically isolated from Crane Creek since the 1940s, except during flood events (e.g., 1973). Diesel pumps are used to move water into or out of the pools to achieve specific management objectives (e.g., provide shallow water shorebird habitat), but precipitation, evapotranspiration, and groundwater also affect water-level. A 60.96-cm diameter culvert and water-control structure allow periodic exchange of water between the pools, but we considered the two pools individual diked wetland sites. During this study, water depths generally were less than 1 m, except in a few deeper former borrow pits where water depths exceeded 3 m. Water quality in the diked pools was similar to adjacent coastal wetlands (Kasat 2006), but submersed aquatic plant assemblages were, in contrast, prevalent.

3.3.2 Field Sampling

We used fyke nets to sample fish assemblages quantitatively. Nets in small frames (45 cm x 45 cm) and large frames (91 cm x 91 cm) and each of two knotted mesh sizes (small: 0.48 cm, large: 1.27 cm) were haphazardly set in a variety of water depths to capture both large and small fishes. For each sample, we installed eight fyke nets, placed in the morning, and began retrieving them the following morning. The data were combined to obtain a 24-hr set. The nets were fished for two consecutive days at each of the three sites within the Crane Creek complex. Each site was sampled in the spring, summer, and fall of 2004 and 2005. The four large-frame nets were set facing the shore in water 1 m deep or greater, with 6-m to 15-m long leads perpendicular to and reaching shore and 3-m long wings extending to each side. The four small-frame nets were set similarly in water less than 1 m deep. Where submersed aquatic vegetation was sparse, nets were set randomly throughout each site in areas with appropriate water depth with leads extending into the edge of dominant emergent vegetation (e.g., *Typha*, *Nelumbo*).

Fish caught in each net were identified according to American Fisheries Society (2004), measured for total length, enumerated, and released. Fish specimens serving as representative samples or requiring further taxonomic work were stored in containers containing an approximately 10% concentration formalin solution. After two weeks in formalin, preserved specimens were transferred to a 95% ethyl alcohol solution for additional analysis and long-term storage. All captured fishes were measured for length unless more than 100 individuals of a species were found. Species with more than 100 individuals captured in a net were enumerated, but only a 100-fish subsample was selected for measurement of total length. Biomass by species was calculated using formulas published in Schneider et al. (2000).

To characterize wetland vegetation, we interpreted aerial photographs and quantitatively sampled major vegetation associations. Color-infrared aerial photographs at a nominal scale of 1:8000 and 1:24000 were collected in July 2004 and July 2005. These images were taken to the field for ground truthing to identify the major vegetation types clearly definable on the photographs, including submersed aquatic plant assemblages. To prepare for stereo interpretation with a mirror stereoscope, preparation of aerial photos was completed following procedures outlined in Owens and Hop (1995). We identified, delineated, digitized, and georeferenced the boundaries of major wetland vegetation associations in the study areas. All geospatially-referenced data were maintained in Universal Transverse Mercator Zone 17 projection and North American Datum 1983.

Wetland plant assemblages were sampled quantitatively in August 2004 and 2005 using up to twenty 1-m x 1-m quadrats placed haphazardly in each dominant wetland vegetation association found at each study site and identifiable in aerial photographs. Visual estimation was used to assign a percent cover value (1% intervals from 1 – 10%; 5% intervals from 15 – 100%) to all identifiable plant species found in the quadrats. To minimize differences among sampling teams, the field crews regularly estimated percent cover values in test plots and calibrated their estimates appropriately.

A YSI model 6920 automated data recorder (YSI Incorporated, Yellow Springs, OH), stationed in the lower water column of Crane Creek near the northeast corner of Pool 2B, measured dissolved oxygen (DO; mg/l), temperature (°C), turbidity (NTU), pH, and water levels (m) in 10-minute intervals from May 5 through October 24, 2005. Similarly, a YSI model 6920V2 automated data recorder stationed in the lower water column of Pool 2B measured dissolved oxygen (DO; mg/l), temperature (°C), turbidity (NTU), and pH hourly from June 23, 2009 – September 15, 2009 (J. Eash, unpublished data, 2009). Simultaneously in 2009, a Pressure Systems KPSI vented submersible pressure transducer collected stage data (m) in 15 minute intervals. The interquartile range method was used to remove outliers (i.e., greater than three times the interquartile range above the third or below the first quartile) for all water quality parameters measured.

Monthly water samples collected in CCUp, Pool 2A, and Pool 2B from May to November 2004 and April to June 2005 were analyzed for soluble reactive phosphorus, ammonia nitrogen, and nitrite-nitrate nitrogen using standard methods (Kasat 2006). Annual loading estimates for dissolved inorganic nitrogen and soluble reactive phosphorus were calculated using these same data (Kasat 2006). Land-surveying techniques were used to determine the elevation of the gages and convert water-depth data to elevations reported in International Great Lakes Datum 1985.

3.3.3 Statistical Analyses

To facilitate data analyses, fish data were entered into an Oracle-driven relational database created by the U. S. Geological Survey – Great Lakes Science Center. Data from the Crane Creek Lower (CCLow), Crane Creek Upper (CCUp), diked Pool 2A (Pool 2A), and diked Pool 2B (Pool 2B) sites were analyzed individually. The fish catch data from all nets at each site were combined, averaged over the number of nets providing data (e.g., 16 nets), and expressed as catch per unit effort (CPUE = fish / net day).

General linear models (ANOVA) were used to identify differences in abundance, biomass, and species richness among year, season, and site. Year, season, site, and season*site interaction factors were included in full model runs. Factors with insignificant p-values (i.e., >0.05) were removed following a backwards stepwise selection process, leaving only significant factors ($p<0.05$) in the model. The Tukey multiple comparison test was applied after a significant ANOVA result (Tukey 1951). To prepare for multivariate analysis and account for the high variance of species in each sample, the fish abundance data (i.e., CPUE) were log transformed (McCune and Grace 2002). Species that were found in three or fewer of the sites were not included in the multivariate analyses (McCune and Grace 2002). To reduce data dimensionality, PC-ORD v. 5.27 was used to perform a non-metric multidimensional scaling (NMS) of the abundance data (autopilot mode set to “slow and thorough”, Euclidean distance measure, random starting number, 500 runs with real data, 500 runs with randomized data, 500 maximum iterations). The Bray-Curtis dissimilarity statistic (values bounded by 0 and 1) was used to calculate how dissimilar the sites were to one another (Bray and Curtis

1957). Smaller values indicate greater similarity in fish species composition, abundance, and biomass between sites.

Plant species richness (i.e., number of taxa) and importance values (i.e., relative frequency and dominance of each taxon in a site; Curtis and McIntosh 1951) were calculated for each site. The Bray-Curtis dissimilarity statistic was used to calculate how dissimilar the sites were to one another based on plant species composition and cover. Because wetland plant assemblages were very similar in 2004 and 2005, only the 2005 vegetation data were used during our analysis. Herbaceous plant nomenclature followed eFloras (2009), and tree nomenclature followed Gleason and Cronquist (1991).

3.4 Results

3.4.1 Fishes

Fyke net sampling collected a total catch of 126,381 fishes (53 species; 18% exotic) in 267 net-days of effort (Table 3.1). Overall, both Crane Creek sites contained many more fishes than the two pools. Analysis-of-variance indicated no significant differences in fish species richness, mean abundance, or mean biomass between 2004 and 2005 (Table 3.2). However, there were significant site differences in species richness, mean abundance, and mean biomass. Season was a significant factor for catch biomass only. Both diked pools were significantly lower than CCUp and CCLow in terms of species richness and mean abundance, and biomass (except between the CCLow and Pool 2B sites; Table 3.3). Similarly, the Bray-Curtis values calculated using fish abundance data revealed great dissimilarity between both pools and both CCUp and CCLow sites (Table 3.4a). The CCUp and CCLow sites were more similar to each other than to either Pool 2A or Pool 2B, and Pool 2A and Pool 2B were more similar to each other than to either CCUp or CCLow sites.

Overall, emerald shiner was the most abundant species followed by gizzard shad, bluegill, and tadpole madtom. Most of the biomass sampled at each site was attributed to bowfin, carp, and gizzard shad. All species found in Pool 2A and Pool 2B also were found in CCLow. Smallmouth bass was the only species found in Pool 2A or Pool 2B but not in CCUp. Eight species captured in CCLow were not captured in CCUp, but only the silverjaw minnow was unique to the CCUp site. Only 48% of the species captured in

CCUp or CCLow were found in either the Pool 2A or Pool 2B wetlands. Some taxa were only found at one site (e.g., silver chub, golden shiner, silverjaw minnow), but most were found in more than one site. Fifty-two species of fish were found in CCLow but only 44 were found in CCUp. Pool 2B produced the fewest species (15). When broken out by site and season, species richness ranged from 11 species (Pool 2B Spring and Fall) to 42 species (CCLow Summer), and mean abundance (CPUE) ranged from 12.3 (Pool 2A Spring) to 1,348.4 (CCLow Fall; Table 3.5). In CCUp, mean abundance ranged from 254.8 to 534.6 (mean = 420.9). Similarly, mean abundance from the CCLow ranged from 703.9 to 1,348.4 (mean = 959.0) while abundance from Pool 2A and Pool 2B were much lower, ranging from 12.3 to 133.6 and 14.7 to 35.7, respectively.

NMS ordination of the species data showed a strong gradient from centrarchid-dominated assemblages (i.e., dominated by bluegill, green sunfish, and largemouth bass) found mostly in the two pools and upstream waters to cyprinid and other lake-associated species (e.g., alewife, spotfin shiner, round goby, freshwater drum) found in greater abundance closer to Lake Erie (Figure 3.2). Axis 1 of the NMS ordination of fish abundance data clearly separated the diked pools from both CCUp and CCLow sites (Figure 3.3).

Seasonal differences in mean length of some species of fish were observed but were not statistically significant. The mean length of gizzard shad, for example, was greater in the spring when spawning was taking place (Table 3.6, Figure 3.4). Although few gizzard shad were captured in the Crane Creek sites each spring, over 67% of them were longer than the minimum length (30.5 cm) identified as adults by Trautman (1981). The mean length was less in the summer and then was greater again in the fall as the YOY fish matured. Similar statistically significant seasonal patterns in mean gizzard shad biomass were observed (Table 3.6b). The total abundance of gizzard shad reflected the annual recruitment pattern with the greatest in summer and fall, a pattern not observed in the total abundance data (Table 3.5). Very few gizzard shad were sampled in Pool 2A and Pool 2B. Similar length and abundance patterns were observed in approximately 45% of the fish species analyzed.

Some species like emerald shiner, however, showed a slightly different pattern in length among seasons because many appeared to reach adult length (6.4 cm; Trautman

1981) by the end of the first growing season (Figure 3.5). The mean length of emerald shiners was less in summer when the YOY were present but was greatest in the fall (Table 3.7). Site and season were significant factors during our comparison of abundance, biomass, and length (Table 3.7).

Excluding the many small carp captured in the fall at the CCUp site, CPUE of carp did not exceed 4.7 fish (Figure 3.6). The greatest mean lengths of carp were observed in the spring, with the smallest lengths observed in the fall at all sites (Table 3.8). Mature common carp were in lower abundance during the warm summer months than during the spring spawning season. In fact, less than 10% of the carp captured in either Crane Creek site during the fall were adult length (>= 30.5 cm; Trautman 1981), but most of the carp trapped in Pool 2A and Pool 2B were adults at this time of the year. Finally, the predator longnose gar had the greatest abundance, biomass, and length in the spring at the CCLow site (Table 3.9, Figure 3.7) but were not present in either Pool 2A or Pool 2B.

3.4.2 Plants

Emergent wetland vegetation and submersed aquatic vegetation were common at all sites studied (see Figure 3.1), but the composition of the plant assemblages varied among the sites. The Bray-Curtis analysis revealed the greatest dissimilarity between the plants in the Pool 2B site and the CCLow site (0.69; Table 3.3b). The CCUp site was moderately dissimilar to the Pool 2A site (0.64), with the lowest Bray-Curtis statistic calculated for the CCUp and CCLow sites (i.e., these two sites were the most similar).

In Crane Creek, 209.6 ha or 54.8% of the total area was vegetated, with most (176.7 ha; 46.2%) located in the CCUp site. Forty-nine plant taxa were identified in the CCUp site (Table 3.10), with the greatest importance values calculated for *Sagittaria latifolia* (31.79), *P. australis* (31.23), *T. angustifolia* (22.90), *N. lutea* (18.25), and *Eleocharis acicularis* (14.83). Forty-seven percent of the taxa were forbs and all four invasive taxa (i.e., *Butomus umbellatus, Phalaris arundinacea, P. australis, T. angustifolia*) were present.

The CCLow site supported 32.9 ha of emergent and submersed aquatic vegetation located adjacent to the shore, on islands, or in small isolated patches. Eight (44%) of the 18 taxa found in the CCLow site were submersed aquatic species with only five taxa

(28%) classified as forbs. *Nelumbo lutea* and *P. australis* had the greatest importance values (41.54 and 41.53, respectively) and the invasive *T. angustifolia* had the next largest importance value (29.67). Submersed aquatic species and *S. latifolia* also had high importance values.

Based on aerial photograph interpretation, 25.7 ha (84.4%) of the Pool 2A area were covered by trees, shrubs, or herbaceous and submersed aquatic vegetation. Pool 2A had the greatest plant species richness (50) among all of the sites (Table 3.10). Thirty-six (72%) of the species in Pool 2A were classified as forbs, grasses, sedges, or rushes, including those considered invasive (e.g., *Butomus umbellatus, Typha angustifolia*). Fewer plants were classified as submersed aquatic vegetation (SAV; 12 taxa; 24%), but *Ceratophyllum demersum* (26.20), *Potamogeton nodosus* (24.08), and *Elodea canadensis* (23.85) had the largest importance values.

In Pool 2B, 37.7 ha (88.8%) of the area were classified as trees, shrubs, or herbaceous and submersed aquatic vegetation. The plant species richness (48) was nearly as large as in the adjacent Pool 2A, with 24 (50%) of the species classified as native and invasive forbs (Table 3.10). Fourteen taxa of submersed aquatic vegetation sampled, but only *Myriophyllum spicatum* had a large importance value (22.51). The forb *Polygonum amphibium* had the largest importance value (39.58), followed by *Leersia oryzoides* (17.47) and the tree *Salix cordata* (14.41).

3.4.3 Water Quality

The CCLow site had a daily mean water temperature of 22.1 °C with a maximum daily range of 11.7 °C during the May 5, 2005 – October 24, 2005 collection period (Table 3.11). The pH levels in the slightly alkaline water varied daily (Max Daily Range = 2.4), with the maximum range occurring on October 18, 2005. Turbidity averaged 59.4 NTU during the 2005 study period and was moderately variable (22.3 Min; 127.5 Max) compared to the wide ranging DO values. DO ranged from 5.6 mg/l to 15.2 mg/l, with a maximum daily range of 16.6 mg/l observed on October 4, 2005. Hypoxic conditions (i.e., < 3 mg/l) were observed in 9% of the CCLow sampled days, and extremely low DO levels < 4 mg/l were observed in 21.2% of the days. Mean water elevation in Crane Creek during the study was 174.2 m with a range of 60 cm.

Measurements in Pool 2B from June 23 – September 15, 2009 revealed conditions similar to those observed at the CCLow site. The daily mean water temperature in Pool 2B (23.9 °C) was similar to the CCLow site, except the maximum daily range was only 6.7 °C. The mean pH was 8.5, and the daily mean turbidity was 19.4 NTU, lower than the levels observed at the CCLow site. The DO ranged from 0.5 to 12.5 (mg/l) with a daily mean of 5.8. Hypoxic conditions were recorded in Pool 2B during over 48% of the days, and DO levels less than 4 mg/l occurred during 74.1% of the days.

Crane Creek water flowing into the refuge had elevated nutrient concentrations, especially compared to water flowing from the refuge to Lake Erie and water in the diked pools. Nitrate concentrations in CCUp (mean = 0.18 mg/l) were higher than in CCLow (mean = 0.09 mg/l), Pool 2A (mean = 0.04 mg/l), and Pool 2B (mean = 0.02 mg/l). Similarly, soluble reactive phosphorus (SRP) concentrations in CCUp (mean = 0.05 mg/l) were elevated compared to CCLow (mean = 0.02 mg/l), Pool 2A (mean = 0.02 mg/l), and Pool 2B (mean = 0.01 mg/l). Ammonia concentrations were slightly higher in Pool 2A (mean = 0.09 mg/l) than in CCUp (mean = 0.08 mg/l) and much higher than in Pool 2B (mean = 0.02 mg/l). Concentrations of nitrate, ammonia, and phosphorus in the water entering the refuge showed a seasonal pattern, with a peak occurring in mid- to late summer. An estimated 2,094 kg/year of dissolved inorganic nitrogen (DIN) and 498 kg/year soluble reactive phosphorus entered the refuge during the study period, and approximately 1,270 kg/year of DIN and 100 kg/year of SRP exited the refuge into Lake Erie.

3.5 Discussion

3.5.1 Coastal Wetland Variability and Fish Use

Coastal wetlands are nutrient-rich areas and can support a diversity of emergent and submerged aquatic vegetation, which in turn provide substrate for fish eggs, protection for young fish, expanded surface areas supporting increased primary and invertebrate production, and feeding grounds for predatory fishes (Chubb and Liston 1986, Wiley et al. 1984). Water quality in these highly productive systems can vary greatly, depending on the source and extent of surface-water inputs (e.g., nutrient-enriched creeks draining agricultural watersheds), rates of photosynthesis and respiration, sediment delivery, and many other variables. The concentration of dissolved oxygen, for example, frequently has a large diurnal range, with wetland plants driving supersaturated peaks followed by very low concentrations after extensive respiration at night (Mitsch and Reeder 1989). Dissolved oxygen rates also vary seasonally, with the lowest concentrations occurring during the warm summer months. Similarly, the amount of suspended sediments and turbidity can vary in response to land use and flooding in the watershed, wind-driven local turbulence, and seasonal spawning activity by carp that churns up sediments (Cooper 1987). The result is a highly dynamic physical system that often produces harsh short-term conditions for fishes and other aquatic biota.

Coastal wetland areas also are very shallow and warm up quickly in the spring. Many spring-spawning fish species (e.g., carp, northern pike, emerald shiners, channel catfish) reproduce in these warmer waters but do not require their shallow habitats throughout the year. In fact, many fish will leave these areas after they spawn and when high water temperature, low dissolved oxygen concentrations, and other habitat characteristics create harsh conditions in the middle of summer, unless they are able to survive in such conditions (e.g., longnose gar, emerald shiners). This seasonal movement and intermittent use of the wetland resources allows fish both to take advantage of favorable habitat conditions (e.g., wetland vegetation, tree branches, and other debris in the water that provide protection for spawning fish and their eggs) and to avoid extended exposure to harsh conditions. While it is widely believed that many species of Great Lakes fishes use coastal wetland habitats at some stage of their life cycle (Trautman 1981, Jude and Pappas 1992), temporal variation in species composition and density is understudied and poorly understood. This is especially so among the large number of significantly degraded wetland habitats that are the focus of many rehabilitation efforts.

3.5.1.1 Effects of Season and Habitat Condition on Fish Abundance

At least 53 species of fish were using the two sites connected to Lake Erie (CCUP and CCLow), even though water quality conditions were relatively poor (Table 3.11). Furthermore, while only 55% of Crane Creek (21% of CCLow site) was vegetated by a small suite of plant species in 2005 (Table 3.10), fish species richness and abundance were rather high. This richness exceeded previous findings of 42 species by Johnson

(1989) and 46 species by Jude and Pappas (1992) in degraded Lake Erie coastal marshes and suggests that these habitats can be highly productive and valuable even in their currently degraded condition. Many of the species we found are commercially, recreationally, or ecologically valued. White perch, white bass, channel catfish, trout-perch, yellow perch, freshwater drum, smallmouth bass, and silver chub are all found in the open waters of Lake Erie and are recreationally or commercially harvested (Herdendorf 1987, Nepszy 1999). Other species, including emerald shiner, gizzard shad, spottail shiner, alewife, and rainbow smelt are important prey fish (Trautman 1981). Several others, such as the sand shiner, orange spotted sunfish, bigmouth buffalo, black buffalo, silver shiner, and western banded killifish, are given a protected status (e.g., species of concern) in Michigan, Ohio, and/or Ontario, Canada. The widespread presence of gizzard shad, emerald shiners, and other forage fish important in the Great Lakes food webs is consistent with earlier observations by Mansfield (1984), Chubb and Liston (1986), Lapointe (1986), Stephenson (1990), Jude and Pappas (1992), Wei et al. (2004), Bouvier (2006), and Bouvier et al. (2009), suggesting that even degraded coastal marshes provide important habitat for large numbers of forage fishes. These species, in addition to YOY of all species, provide food for larger local predatory species (e.g., longnose gar, northern pike, largemouth bass) and piscivorous water birds, which are found in Crane Creek throughout the ice-free season.

The observed seasonal variability in fish assemblages is related to many factors, including changing water levels, species' reproductive strategies and other life history traits, and likely also unquantified flow-related sampling biases. For example, summer was generally the time when the fewest fish were caught during our study, although a few species were most abundant during the summer months (e.g., channel catfish, gizzard shad, spottail shiner, white bass, white crappie, white perch). The low overall summer abundance observed at the CCUp and CCLow sites is likely related to harsh environmental conditions in the shallow wetland habitats (Table 3.11) that repel all but the most tolerant fishes. The air and water temperatures were at their annual maximums, and frequently the dissolved oxygen was extremely low. Most large fishes likely moved to cooler water near or in Lake Erie to find refuge during this period. Young-of-year fishes hatched in the wetland may have remained there during the summer months, possibly in great abundance, but our sampling gear mesh size would not have captured them effectively.

Across all seasons, the greatest abundance of fish occurred in the CCLow site that was located close to the connection with Lake Erie, where schools of gizzard shad and emerald shiners were captured (Table 3.1, Table 3.5). These two Lake Erie forage fish species comprised 90.4% of the total abundance sampled at the CCLow site. The schooling behavior of these species accounts for the large variances in catch by fyke nets on certain sampling days (i.e., they are present only in large numbers) and a lower abundance and therefore biomass on other days when the schools avoid or bypass nets. However, it is unclear whether nets with few schooling fish indicate that those species are not using the wetland habitats during the sampling timeframe or if the patchy nature of their distribution minimized their probability of capture. Increased sampling effort (e.g., more nets deployed) or use of other sampling strategies (e.g., a high resolution acoustic camera) could provide a clearer characterization of the temporal variability of species richness and abundance in a wetland and would help clarify the potential benefits of habitat rehabilitation (e.g., improve access to spawning habitat during spawning season).

3.5.1.2 Relationships Between Fish Abundance, Biomass, and Length

Reproductively mature fishes appeared to be using the marsh habitats mostly in the spring, likely to find suitable conditions for spawning (e.g., warmer water, egg attachment sites, protection from predators). The marsh habitats then acted as a nursery ground for smaller fishes resulting from spring spawning runs. For example, 100% of the gizzard shad captured during the spring at the CCUp site and nearly 67% captured during the spring at the CCLow site were long enough to be considered adults (Trautman 1981, Minns et al. 1993; Figure 3.4c). It is likely that the few gizzard shad observed in the spring were there to spawn or were feeding in the productive shallows before spawning in the Lake Erie nearshore. By summer, the abundance of gizzard shad increased tremendously (Figure 3.4a), and the mean length decreased. Schools of young shad were using the wetland as a nursery even though the water was quite warm and there were large diurnal swings in dissolved oxygen levels (Table 3.11; K. Kowalski, unpublished data, 2006). Nutrient-rich river wetland habitats, including the CCUP site, helped the young fishes grow in length and biomass by the fall (Figure 3.4b), but the overall abundance of shad was lower in the fall likely due to mortality or emigration to Lake Erie.

Some fish species appeared to use wetland habitats well into the fall, especially locales closer to Lake Erie. Emerald shiners, for example, had a large abundance in the spring and fall at the CCLow site (Figure 3.5a). The mean length of the fish present in the fall was much greater than during the summer and slightly exceeded the mean spring length. This suggests that the emerald shiners spawned in marsh habitats during the spring, grew quickly over summer, stayed in the wetland after maturing, and were again efficiently captured in our fall sample. We hypothesize that the very low abundance and biomass observed in our study was because either the YOY emerald shiners were too small to be captured with fyke nets (i.e., they were smaller than our 0.48-cm net mesh) during our summer sampling or that they are moving out of the marsh temporarily.

Other species like common carp revealed a unique pattern (Figure 3.6) in that the mean length and biomass was lowest in the fall (Figure 3.6b, Figure 3.6c). Spawning adults observed in the spring tended to leave the marsh after spring spawning and were not frequently caught into the fall. This is a useful observation for resource managers because it suggests that the wetland damage (e.g., uprooted vegetation, disturbed sediments) caused by mature carp feeding on benthic invertebrates and submersed aquatic plants should decrease after spawning occurs if the mature carp are allowed to leave the wetland. The abundance of carp also generally decreased with time, except for some large fall samples of small fishes at the CCUp site. Length of fishes captured in the fall suggest that all were YOY fishes using the marsh, in particular the deeper upstream pools found in the CCUp site, when conditions in shallower lower marsh were decreasing in quality for the season.

Finally, some fish species only seemed to access the marsh habitats during certain times of the year, likely associated with spawning. Longnose gar, for example, were only found in the spring and summer, with larger fishes dominating the catches (Figure 3.7c). Gar are one of the few fish species able to thrive in low oxygen environments by breathing air (Scott and Crossman 1998), so they are able to hunt summer nursery grounds and feed on the numerous YOY of prey species even though DO levels are low and water temperature is warm. Therefore, benefits of coastal wetland habitat to longnose gar depend on access to the habitats during the spring and summer seasons and the ability to retreat to Lake Erie at other times of the year.

3.5.2 Ecological Differences in Diked Pools

3.5.2.1 Habitat Characteristics

The composition and extent of habitat in diked units (i.e., plant assemblages) often are quite different than in adjacent Lake Erie coastal wetlands. Management actions (e.g., exotic species removal, periodic dewatering) are commonly used to promote the growth of emergent vegetation and maximize habitat for shorebirds, ducks, and other migratory water birds with little consideration for potential benefits to adjacent Lake Erie fish populations. Although not always supporting greater plant species richness, the plant assemblages resulting from these management actions are often reported to be robust and well-established compared to nearby coastal wetlands degraded by poor water quality, extended periods of high water levels, or shoreline armoring (Sherman et al. 1996, Gottgens et al. 1998, Thiet 2002). However, the isolation from fluctuating water levels in Lake Erie prevents these diked habitats from maintaining diverse plant assemblages without regular management draw-down actions that mimic lake-driven events.

Management of water levels and control of invasive plant species (e.g., *P. australis*, *Lythrum salicaria*) by refuge managers over the previous 35 years has undoubtedly contributed to woody, herbaceous, and aquatic vegetation covering over 84% of Pool 2A and 89% of Pool 2B. These plant assemblages were quite similar to each other (Table 3.4b) yet very different from the degraded CCLow site, which supported a much lower species richness. Surprisingly, the CCUp site was similar to both Pool 2A and Pool 2B even though it was exposed to many of the stressors (e.g., carp access, wave attack, shoreline armoring) that degraded the CCLow site. The greater plant species richness in CCUp likely was associated with the suite of species growing in the higher elevation wet meadows and transitional mudflats, but these species rich areas do not translate into increased fish habitat unless water levels are high enough to inundate them. Similarly, the species-rich habitats of the diked pools do not translate into increased fish habitat for Great Lakes fishes unless they are inundated and freely accessible by fishes.

Only small fishes are able to enter and exit the diked wetlands when pumps were used to manage water levels by exchanging water with Crane Creek. Presumably, large fishes are excluded completely. These conditions supported the development of fish assemblages in the diked pools that were distinct but not unique compared to the fish assemblages found in the CCUp and CCLow sites (Table 3.4a). All 25 of the species found in Pool 2A and Pool 2B were also found in the Crane Creek sites. This indicates, as expected, that the fish species in the diked wetlands are just a subset of the greater source population in Crane Creek and Lake Erie. It is likely that the fishes that were most abundant in the Crane Creek assemblages (e.g., gizzard shad, emerald shiner) were a larger component of the diked wetland assemblage immediately after the pools were first isolated or after the last major breach of the earthen dikes but were not able to survive long term because of harsh environmental conditions, predation, or other factors. Night time dissolved oxygen minima, for example, were generally even lower in the diked units than observed in Crane Creek proper. Dissolved oxygen dropped low enough to create hypoxic conditions in Pool 2B over 48% of the days we collected samples (Table 3.11b). Many species may have been extirpated by the low DO conditions, but it also is possible that they were outcompeted by those better adapted to the shallow lentic habitat or able to seek refuge in deeper portions of the pools created when sediments were excavated to create the surrounding dikes (e.g., common carp, bullhead, bluegill, largemouth bass). The large diversity of plant species and structural forms (i.e., habitat complexity) provided extensive habitat for Centrarchids and may help promote increased fish diversity (Emery 1978) even without the presence of lake-associated species. Johnson et al. (1997) observed similar conditions in other Lake Erie wetlands and concluded that the diked wetland fish communities appear isolated from other nearby populations, a conclusion supported by the results of our study. More specifically, an analysis of the size and age of white crappies also suggested that diked wetland populations were functionally isolated from those in coastal wetlands despite occasional water exchange (Markham et al. 1997).

3.5.3 Implications for Habitat Rehabilitation

Despite the generally poor water quality delivered by Crane Creek to its river-mouth wetlands, utilization by fishes was high and productivity in these remaining free-flowing units appeared to exceed greatly that of adjacent diked wetlands. Given the difficulties in reducing nutrient and sediment losses from Lake Erie watersheds, it seems that hydrologic restoration of diked wetland units would be a relatively easy way to bolster Lake Erie fish populations.

Differences in fish assemblages in coastal wetlands and diked wetlands have been studied before (Johnson 1989, Johnson et al. 1997, Markham et al. 1997, Bouvier 2006). My results are similar and demonstrate larger differences, but data from this study also demonstrate that there are large seasonal variations in the abundance and species richness of fish accessing degraded coastal wetland habitats from Lake Erie and its tributaries. The inaccessibility of adjacent diked wetland habitats prevents seasonal migration by fishes, even though they can provide habitat with a greater species richness and abundance of wetland plants. Spawning fishes are prevented from using the floristically and structurally diverse macrophyte and emergent wetland plant assemblages common to managed wetland units. Likewise, young fishes are not able to use the protected wetlands as nursery areas, often even if a limited hydrologic connection exists (Johnson 1989). Because the diked wetlands are essentially closed systems, fish and other biota are not able to emigrate or seek better habitat when dissolved oxygen levels are low during warm summer months or water levels drop. In addition, any YOY production that might occur in the diked wetland does not contribute to Lake Erie productivity.

Maintaining and enhancing hydrologic connectivity is one of the most important challenges facing the rehabilitation of fish habitat in Lake Erie coastal wetlands. Permanent hydrologic reconnection of the diked wetlands in this unit to Lake Erie could restore the water-level variability associated with high quality wetland-plant assemblages (Burton 1985, Keddy and Reznicek 1985, Wilcox 2004, Herrick and Wolf 2005), but most diked wetlands have extensive shoreline armoring around their perimeter or do not have topographic relief sufficient to allow wetland plants to respond to long-term water-level fluctuations like less-developed coastal wetlands. Therefore, permanent hydrologic reconnection could reduce the diversity and abundance of wetland plants if water levels were much higher or lower than the sediment surface in the diked wetland (e.g., lower water levels associated with global climate change; Doka et al. 2006). Plant diversity also could be reduced if enough subsidence or accretion has occurred in the diked-wetland sediments to alter their elevation relative to the main channel. For example, plants growing at lower elevations (i.e., subsided from historic elevations) may be more vulnerable flooding associated with high water levels in Lake Erie than plants growing at higher elevations. During periods of low water levels in Lake Erie, the low elevations may be the only places suitable for wetland plant growth as upland plants, shrubs, and trees invade higher elevations. The diked units we sampled generally had elevations below those observed in broad expanses of Crane Creek, so they will remain wetter or be even more flooded if hydrologically reconnected. In addition to very high or low water levels, the connection could introduce a suite of stressors that degrade coastal wetland habitats (e.g., high turbidity, phosphorus loads, more access by common carp). These stressors might ultimately degrade the newly reconnected coastal wetland habitat and, in the long run, eliminate many of the ecological benefits of reconnection.

If permanent hydrologic reconnection will restore select functions and values (e.g., vegetated fish habitat, flood retention) for a while but ultimately contribute to wetland degradation, then what options are there for long-term habitat rehabilitation? Wilcox and Whillans (1999) suggested that mimicry of natural processes (e.g., hydrology) is a good rehabilitation philosophy, and we argue that management interventions can be used to mimic more natural hydrologic patterns and maximize the seasonal use of wetlands by Lake Erie fishes. Rogers et al. (1994) suggested that managing fish-passage or water-control structures at certain times of the year can reduce the negative impacts of the impoundment, but we suggest that anything less than full hydrologic connection throughout the entire year will impact the Lake Erie fish assemblages negatively. Our results support the idea that many different fish use coastal wetland habitats at different times and for different purposes throughout the year. Therefore, access to valuable coastal wetland habitat could be restored by using an appropriately-designed fish-passage structure that allows fish of all shapes and sizes (excluding invasive common carp) to pass through without harm (French et al. 1999). The operation of fish-passage structures located in the Great Lakes (e.g., Cootes Paradise in Lake Ontario, Metzger Marsh in Lake

Erie) has shown that structures can be designed both to promote fish passage and to provide a hydrologic connection of similar size to historical channels.

Hydrologic connection could be maintained year-round until conditions in the rehabilitated wetland become degraded enough that management objectives can no longer be met. At this point, the fish-passage structure could be closed to allow dewatering and other management actions to “reset” the wetland similar to what might occur naturally during a couple seasons of low water levels. Depending on the topography of the wetland, borrow pits or other low areas could serve as refugia for trapped fishes. Using an adaptive management strategy and giving consideration for Lake Erie water-level patterns, an optimal frequency and duration for temporarily isolating the wetlands could be determined. It would be critical to isolate the wetland only long enough to reestablish perennial emergent wetland plants (e.g., two years) and address any invasive species problems. Fishes would not be able to access the wetland habitats while these actions were taking place, similar to when low water levels limit access to upslope habitats, but higher quality habitats would be made available each time the diked wetlands are reconnected.

Once the perennial vegetation has reestablished sufficiently (i.e., grown tall enough) to survive natural water levels, the water-control structure(s) could be reopened to start the cycle again (Ball 1985). In essence, this cycle would use periodic management actions to provide intermediate disturbances (i.e., low water levels) that reset the system during times of extended high water levels in Lake Erie, similar to the efforts to isolate coastal marsh temporarily using portable water-filled cofferdams (Kowalski et al. 2009). Cyclic isolation would both provide additional coastal wetland habitat to Lake Erie fish assemblages throughout the year and allow resource managers enough control to maintain high quality wetland habitat and sustainably achieve management objectives.

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Table 3.1. Abundance (CPUE = fish/net day; bold top row) and biomass (g/net day) sampled in each site during 2004 and 2005. Taxa are identified as native (N) or exotic (E) to the Great Lakes and classified as forage (F) or game (G) species.

				CCUp			CCLow			Pool 2A			Pool 2B		
Scientific Name	**Com. Name**	**Nat**	**Class**	**Spring**	**Summer**	**Fall**	**Spring**	**Summer**	**Fall**	**Spring**	**Summer**	**Fall**	**Spring**	**Summer**	**Fall**
TOTAL				**534.6** *25,447.7*	**254.8** *17,188.5*	**432.5** *4,763.2*	**824.8** *20,865.9*	**703.9** *8,325.8*	**1,348.4** *4,289.6*	**12.3** *1,939.7*	**133.6** *2,547.8*	**33.5** *2,446.6*	**35.7** *3,773.4*	**20.3** *1,740.1*	**14.7** *4,185.3*
Alosa pseudoharengus Wilson	Alewife	E	F	**0.0** *0.0*	**0.1** *<0.1*	**<0.1** *0.3*	**0.1** *11.4*	**22.6** *16.8*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Ameiurus melas Rafinesque	Black bullhead	N	G	**0.3** *97.0*	**0.5** *29.4*	**4.9** *58.5*	**0.0** *0.0*	**<0.1** *13.0*	**0.0** *0.0*	**0.4** *52.3*	**0.6** *31.8*	**1.1** *129.0*	**0.2** *88.0*	**0.0** *0.0*	**0.1** *80.1*
Ameiurus natalis Lesueur	Yellow bullhead	N	G	**1.8** *509.1*	**4.3** *273.2*	**23.7** *575.6*	**0.6** *118.6*	**0.4** *102.3*	**<0.1** *0.4*	**0.5** *109.4*	**1.1** *167.8*	**2.1** *282.2*	**0.6** *113.2*	**0.4** *80.4*	**4.9** *873.1*
Ameiurus nebulosus Lesueur	Brown bullhead	N	G	**0.2** *95.2*	**0.2** *39.3*	**5.4** *64.7*	**0.2** *71.8*	**0.1** *15.1*	**0.3** *1.5*	**0.0** *0.0*	**0.8** *99.8*	**0.9** *104.4*	**0.0** *0.0*	**0.1** *11.2*	**0.3** *27.2*
Amia calva Linnaeus	Bowfin	N	G	**1.4** *2,353.4*	**2.3** *4,242.9*	**0.1** *222.5*	**1.8** *3,422.6*	**1.0** *1,948.4*	**0.0** *0.0*	**0.3** *482.8*	**0.6** *836.4*	**1.1** *1,502.9*	**0.6** *1,349.2*	**0.4** *821.3*	**0.4** *711.3*
Aplodinotus grunniens Rafinesque	Freshwater drum	N	G	**0.1** *193.8*	**0.5** *278.8*	**0.0** *0.0*	**0.4** *149.5*	**0.8** *41.9*	**0.1** *0.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Carassius auratus Linnaeus	Goldfish	E	F	**1.2** *806.7*	**6.2** *50.1*	**1.5** *30.1*	**2.5** *684.9*	**1.7** *137.1*	**0.1** *0.6*	**0.1** *52.5*	**0.1** *20.4*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Carpiodes cyprinus Lesueur	Quillback	N	G	**0.7** *854.7*	**<0.1** *0.1*	**<0.1** *1.0*	**0.3** *264.4*	**0.1** *0.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Catostomus commersoni Lacepede	White sucker	N	G	**0.0** *0.0*	**0.0** *0.0*	**0.1** *18.8*	**0.6** *567.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Cyprinella Spiloptera Cope	Spotfin shiner	N	F	**0.1** *0.3*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *0.2*	**0.5** *1.1*	**0.2** *0.6*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Cyprinus carpio Linnaeus	Carp	E	G	**4.7** *14,606.2*	**4.5** *10,862.2*	**30.4** *1,740.4*	**2.2** *7,503.8*	**2.0** *3,751.6*	**0.4** *121.4*	**0.3** *587.0*	**2.9** *364.8*	**0.1** *2.5*	**0.5** *1,178.4*	**0.2** *78.6*	**0.3** *482.1*
Dorosoma cepedianum Lesueur	Gizzard shad	E	F	**<0.1** *12.2*	**154.6** *281.5*	**129.8** *1,116.7*	**0.7** *332.2*	**448.7** *906.2*	**204.5** *2,106.8*	**0.1** *25.4*	**0.3** *29.0*	**0.5** *6.9*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*

Esox lucius Linnaeus	Northern pike	N	G	**<0.1** *48.2*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *266.5*	**0.0** *0.0*	**<0.1** *142.4*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.4** *474.5*	**0.4** *578.0*	**1.1** *1,460.7*
Fundulus diaphanus menona Lesueur	Western banded killifish	N	F	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Hybopsis amblops Rafinesque	Bigeye chub	N	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *0.2*	**0.2** *0.9*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Hypentelium nigricans Lesueur	Northern hogsucker	N	G	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *2.0*	**<0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Ictalurus punctatus Rafinesque	Channel catfish	N	G	**0.2** *230.2*	**1.5** *29.5*	**17.1** *39.2*	**0.1** *113.8*	**4.7** *198.1*	**1.0** *1.2*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Ictiobus bubalus Rafinesque	Smallmout h buffalo	N	G	**1.2** *3662.6*	**0.0** *0.0*	**0.0** *0.0*	**2.2** *3,964.4*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Ictiobus cyprinellus Valenciennes	Bigmouth buffalo	N	G	**0.5** *726.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *<0.1*	**<0.1** *5.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *161.1*	**0.0** *0.0*	**0.0** *0.0*
Ictiobus niger Rafinesque	Black buffalo	N	G	**<0.1** *35.7*	**0.0** *0.0*	**<0.1** *0.1*	**0.0** *0.0*	**<0.1** *<0.1*	**<0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Labidesthes sicculus Cope	Brook silverside	N	F	**0.0** *0.0*	**0.1** *<0.1*	**<0.1** *0.1*	**0.0** *0.0*	***1.4*** *1.1*	**0.4** *1.3*	**0.0** *0.0*	**0.3** *0.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Lepisosteus osseus Linnaeus	Longnose gar	N	G	**0.6** *332.8*	**0.6** *494.1*	**0.0** *0.0*	**1.4** *1,257.8*	**0.9** *514.6*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Lepomis cyanellus Rafinesque	Green sunfish	N	G	**<0.1** *0.3*	**0.2** *0.1*	**0.7** *2.4*	**0.0** *0.0*	**<0.1** *0.1*	**0.8** *5.8*	**0.1** *2.4*	**18.4** *18.0*	**10.3** *29.9*	**0.0** *0.0*	**6.6** *20.2*	**0.1** *1.0*
Lepomis gibbosus Linnaeus	Pumpkin-seed	N	G	**0.4** *17.7*	**0.3** *5.9*	**5.3** *40.7*	**1.7** *66.3*	**1.4** *24.8*	**0.7** *3.0*	**2.3** *30.7*	**0.6** *19.1*	**0.5** *6.4*	**2.6** *19.0*	**0.3** *3.6*	**0.0** *0.0*
Lepomis humilis Girard	Orange spotted sunfish	N	G	**0.0** *0.0*	**6.5** *7.3*	**8.0** *7.0*	**0.1** *0.1*	**3.2** *1.9*	**22.5** *13.7*	**0.1** *0.2*	**1.4** *3.2*	**1.2** *1.9*	**0.0** *0.0*	**0.1** *0.1*	**0.0** *0.0*
Lepomis macrochirus Rafinesque	Bluegill	N	G	**0.4** *17.4*	**17.5** *97.7*	**31.1** *390.2*	**0.7** *13.3*	**4.5** *27.1*	**1.8** *1.5*	**6.7** *145.2*	**36.9** *109.5*	**11.3** *72.9*	**29.2** *207.1*	**5.3** *49.0*	**5.1** *239.3*
Luxilus chrysocephalus Rafinesque	Striped shiner	N	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *8.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Macrhybopsis storeriana Kirtland	Silver chub	N	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *1.8*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*

Micropterus dolomieui Lacepede	Small-mouth bass	N	G	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *11.3*	**0.3** *2.7*	**<0.1** *2.1*	**0.1** *95.4*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Micropterus salmoides Lacepede	Large-mouth bass	N	G	**0.1** *36.5*	**0.3** *47.4*	**0.9** *63.9*	**0.9** *157.5*	**0.4** *22.0*	**0.1** *0.4*	**0.7** *343.1*	**65.4** *719.5*	**0.7** *147.6*	**0.1** *43.8*	**3.5** *9.4*	**0.2** *129.8*
Minytrema melanops Rafinesque	Spotted sucker	N	G	**<0.1** *10.2*	**0.6** *209.4*	**0.1** *6.4*	**0.1** *47.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Morone americana Gmelin	White perch	E	G	**0.6** *48.9*	**7.4** *35.7*	**1.5** *11.0*	**0.6** *98.8*	**48.2** *175.7*	**4.8** *23.4*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *9.8*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Morone chrysops Rafinesque	White bass	N	G	**0.3** *77.9*	**16.4** *33.8*	**<0.1** *0.4*	**2.7** *552.5*	**38.8** *322.8*	**<0.1** *0.2*	**0.1** *9.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Neogobius melanostomus Pallas	Round goby	E	F	**0.0** *0.0*	**0.1** *<0.1*	**0.1** *0.1*	**0.1** *1.3*	**2.5** *2.2*	**10.8** *11.6*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Notemigonus crysoleucas Mitchell	Golden shiner	N	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *1.7*	**<0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Notropis atherinoides Rafinesque	Emerald shiner	Y	F	**488.9** *556.6*	**19.9** *8.7*	**88.1** *126.8*	**781.4** *943.5*	**93.1** *47.0*	**1,071.3** *1,758.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Notropis buccatus Cope	Silverjaw minnow	N	F	**0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Notropis hudsonius Clinton	Spottail shiner	N	F	**1.3** *14.9*	**1.8** *1.6*	**0.1** *0.2*	**11.0** *122.7*	**19.3** *18.4*	**6.6** *22.8*	**0.1** *0.2*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Notropis photogenis Cope	Silver shiner	N	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.2** *1.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Notropis stramineus Cope	Sand shiner	N	F	**27.0** *17.3*	**0.0** *0.0*	**0.3** *0.4*	**5.1** *33.9*	**<0.1** *<0.1*	**7.3** *33.2*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Noturus gyrinus Mitchell	Tadpole madtom	N	F	**0.8** *1.8*	**3.3** *9.1*	**80.6** *212.8*	**1.0** *2.1*	**0.4** *1.4*	**8.4** *14.7*	**0.1** *0.3*	**0.1** *0.4*	**1.1** *3.1*	**0.0** *0.0*	**0.1** *0.6*	**0.0** *0.0*
Osmerus mordax Mitchell	Rainbow smelt	E	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.1** *0.8*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Perca flavescens Mitchell	Yellow perch	N	G	**0.0** *0.0*	**0.3** *5.3*	**1.0** *28.5*	**0.2** *8.7*	**0.6** *9.0*	**<0.1** *1.5*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Percina caprodes Rafinesque	Logperch darter	N	F	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *0.2*	**0.0** *0.0*	**0.8** *0.8*	**<0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Percopsis omiscomaycus Walbaum	Trout-perch	N	G	**<0.1** *0.5*	**<0.1** *0.1*	**0.0** *0.0*	**4.3** *25.5*	**0.1** *0.2*	**0.1** *0.5*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*

Petromyzon marinus Linnaeus	Sea lamprey	E	F	**0.0** *0.0*	**0.1** *1.9*	**0.0** *0.0*	**0.1** *4.1*	**0.0** *0.0*	**0.1** *4.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Phenacobius mirabilis Girard	Sucker-mouth minnow	N	F	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *<0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Pimephales notatus Rafinesque	Bluntnose minnow	N	F	**0.4** *1.1*	**0.2** *0.5*	**0.3** *0.3*	**0.4** *1.6*	**0.4** *0.6*	**3.8** *5.3*	**0.1** *0.3*	**0.2** *0.6*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Pimephales promelas Rafinesque	Fathead minnow	N	F	**0.8** *0.8*	**0.1** *0.1*	**0.2** *0.4*	**0.3** *0.6*	**0.4** *0.3*	**2.0** *2.9*	**0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Pomoxis annularis Rafinesque	White crappie	N	G	**0.0** *0.0*	**2.8** *5.3*	**0.6** *8.2*	**0.0** *0.0*	**2.9** *8.6*	**0.0** *0.0*	**0.0** *0.0*	**0.8** *15.6*	**0.3** *21.6*	**0.6** *69.7*	**1.3** *31.9*	**0.7** *47.9*
Pomoxis nigromaculatus Lesueur	Black crappie	N	G	**0.4** *81.7*	**1.6** *82.5*	**0.4** *6.3*	**0.2** *30.1*	**1.3** *10.7*	**<0.1** *0.2*	**0.1** *2.7*	**3.1** *107.3*	**2.3** *119.0*	**0.8** *69.4*	**1.6** *47.4*	**1.7** *132.9*
Semotilus atromaculatus Mitchell	Creek chub	N	F	**<0.1** *0.1*	**0.0** *0.0*	**0.0** *0.0*	**<0.1** *0.2*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*
Umbra limi Kirtland	Central mud-minnow	N	F	**0.0** *0.0*	**0.1** *0.4*	**0.1** *0.2*	**0.1** *0.4*	**<0.1** *0.0*	**<0.1** *0.1*	**0.1** *0.7*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*	**0.0** *0.0*

Table 3.2. Three-way ANOVA results (Type III Sum of Squares) of fish species richness, abundance per unit effort, and biomass per unit effort from 2004 and 2005 sampling in Crane Creek. Significant (p<0.05) values are in bold and a star (*) indicates that a factor became significant after other factors were removed following a backwards stepwise selection process.

a) Species Richness

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.69984592	0.69984592	1.09	0.3187
Season	2	0.77402437	0.38701218	0.60	0.5642
Site	3	57.42336504	19.14112168	29.84	**<0.0001**
Site*Season	6	1.56916595	0.26152766	0.41	0.8591

b) Abundance

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	26.0907177	26.0907177	3.22	0.1004
Season	2	2.7680013	1.3840006	0.17	0.8454
Site	3	617.7050704	205.9016901	25.38	**<0.0001**
Site*Season	6	47.8401584	7.9733597	0.98	0.4809

c) Biomass

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.5084237	0.5084237	0.09	0.7664
Season	2	34.3692303	17.1846152	3.13	0.0837*
Site	3	143.7225254	47.9075085	8.74	**0.0030**
Site*Season	6	40.0442187	6.6740365	1.22	0.3672

Table 3.3. P-values resulting from ANOVA using fish species richness, abundance per unit effort, and biomass per unit effort data from 2004 and 2005 sampling in Crane Creek. Data were analyzed using the least squares means adjustment for multiple comparisons (Tukey method). Significant ($p<0.05$) values in bold.

a) Species Richness

	CCUp	**CCLow**	**Pool 2A**	**Pool 2B**
CCUp		0.5384	**0.0002**	**<0.0001**
CCLow			**<0.0001**	**<0.0001**
Pool 2A				0.0892
Pool 2B				

b) Abundance

	CCUp	**CCLow**	**Pool 2A**	**Pool 2B**
CCUp		0.2634	**0.0005**	**0.0001**
CCLow			**<0.0001**	**<0.0001**
Pool 2A				0.0892
Pool 2B				

c) Biomass

	CCUp	**CCLow**	**Pool 2A**	**Pool 2B**
CCUp		0.9285	**0.0160**	**0.0361**
CCLow			**0.0445**	0.0945
Pool 2A				0.9892
Pool 2B				

Table 3.4. Results of Bray-Curtis dissimilarity analysis by site conducted using fish abundance data and plant importance values. A value of 1 means the sites do not share any common species and a value of 0 means that the sites have the same density and composition.

a) Fish

	CCUp	**CCLow**	**Pool 2A**	**Pool 2B**
CCUp		0.5045	0.9145	0.8960
CCLow			0.9862	0.9814
Pool 2A				0.4962
Pool 2B				

b) Plants

	CCUp	**CCLow**	**Pool 2A**	**Pool 2B**
CCUp		0.3995	0.6456	0.7899
CCLow			0.7121	0.8483
Pool 2A				0.6063
Pool 2B				

Table 3.5. Fish species richness at each site during each season and mean abundance (CPUE = fish/net day) from 2004 and 2005 sampling in Crane Creek.

Site	Spring (April/June)	Summer (July/August)	Fall (November)
CCUp	33 (534.6)	33 (254.8)	34 (432.5)
CCLow	41 (824.7)	42 (703.9)	35 (1,348.4)
Pool 2A	19 (12.3)	17 (133.6)	15 (33.5)
Pool 2B	11 (35.7)	13 (20.25)	11 (14.7)

Table 3.6. Three-way ANOVA results (Type III Sum of Squares) of gizzard shad abundance per unit effort, biomass per unit effort, and length from 2004 and 2005 sampling in Crane Creek. Significant ($p<0.05$) values are in bold and a star (*) indicates that a factor became significant after other factors were removed following a backwards stepwise selection process.

a) Abundance

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.002791	0.002791	0.00	0.9795
Season	2	681.491968	340.745984	84.45	**<0.0001**
Site	3	1639.958084	546.652695	135.49	**<0.0001**
Site*Season	6	530.025473	88.337579	21.89	**<0.0001**

b) Biomass

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.001999	0.001999	0.00	0.9906
Season	2	108.572306	54.286153	3.96	0.0508
Site	3	1211.586881	403.862294	29.43	**<0.0001**
Site*Season	6	213.582802	35.597134	2.59	0.0812

c) Length

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.00002078	0.00002078	0.00	0.9982
Season	2	5.98785796	2.99392898	0.75	0.4971
Site	3	68.52763274	22.84254425	5.69	**0.0134***
Site*Season	6	8.45413048	1.40902175	0.35	0.8951

Table 3.7. Three-way ANOVA results (Type III Sum of Squares) of emerald shiner abundance per unit effort and biomass per unit effort from 2004 and 2005 sampling in Crane Creek. Significant (p<0.05) values are in bold and a star (*) indicates that a factor became significant after other factors were removed following a backwards stepwise selection process.

a) Abundance

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	4.3896241	4.3896241	0.59	0.4569
Season	2	49.2283482	24.6141741	3.33	**0.0738***
Site	3	473.0425133	157.6808378	21.35	**<0.0001**
Site*Season	6	75.0059529	12.5009921	1.69	0.2126

b) Biomass

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	10.3141910	10.3141910	1.92	0.1936
Season	2	116.3108159	58.1554080	10.81	**0.0025**
Site	3	822.0158398	274.0052799	50.93	**<0.0001**
Site*Season	6	142.2533248	23.7088875	4.41	**0.0164**

c) Length

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.04307336	0.04307336	0.18	0.6830
Season	2	1.44170469	0.72085234	2.94	**0.0946***
Site	3	52.40855154	17.46951718	71.34	**<0.0001**
Site*Season	6	1.54914802	0.25819134	1.05	0.4431

Table 3.8. Three-way ANOVA results (Type III Sum of Squares) of carp abundance per unit effort and biomass per unit effort from 2004 and 2005 sampling in Crane Creek. Significant (p<0.05) values are in bold and a star (*) indicates that a factor became significant after other factors were removed following a backwards stepwise selection process.

a) Abundance

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	48.4404111	48.4404111	2.90	0.1164
Season	2	33.0435218	16.5217609	0.99	0.4021
Site	3	230.7944401	76.9314800	4.61	**0.0253**
Site*Season	6	50.7537920	8.4589653	0.51	0.7911

b) Biomass

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	8.5556312	8.5556312	0.26	0.6233
Season	2	687.3566817	343.6783409	10.26	**0.0031**
Site	3	647.0695146	215.6898382	6.44	**0.0089**
Site*Season	6	180.5366871	30.0894478	0.90	0.5290

c) Length

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.9497527	0.9497527	0.26	0.6200
Season	2	135.7757433	67.8878717	18.60	**0.0003**
Site	3	79.7511749	26.5837250	7.28	**0.0058**
Site*Season	6	35.1281000	5.8546833	1.60	0.2350

Table 3.9. Three-way ANOVA results (Type III Sum of Squares) of longnose gar abundance per unit effort and biomass per unit effort from 2004 and 2005 sampling in Crane Creek. Significant (p<0.05) values are in bold and a star (*) indicates that a factor became significant after other factors were removed following a backwards stepwise selection process.

a) Abundance

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	10.38565472	10.38565472	1.95	0.1898
Season	2	30.79427207	15.39713604	2.90	**0.0977***
Site	3	65.12177728	21.70725909	4.08	**0.0356**
Site*Season	6	34.42674302	5.73779050	1.08	0.4308

b) Biomass

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.09014006	0.09014006	0.59	0.4584
Season	2	1.25071879	0.62535940	4.10	**0.0468**
Site	3	2.78986129	0.92995376	6.09	**0.0107**
Site*Season	6	1.73731501	0.28955250	1.90	0.1693

c) Length

Source	Degrees of Freedom	Sum of Squares	Mean Square	F value	P
Year	1	0.89240490	0.89240490	1.78	0.2088
Season	2	8.54635371	4.27317685	8.54	**0.0058**
Site	3	17.02596571	5.67532190	11.34	**0.0011**
Site*Season	6	9.67291543	1.61215257	3.22	0.0445

Table 3.10. Calculated importance values of the plant species sampled in Crane Creek Upper (CCUp), Crane Creek Lower (CCLow), Pool 2A, and Pool 2B wetland sites. Species richness at each site is noted at the bottom of the columns.

Type	Species	Site			
		CCUp	CCLow	Pool 2A	Pool 2B
Forb	*Abutilon theophrasti* Medikus	0.33		0.42	0.50
	Alisma triviale Pursh.	0.34		2.68	1.02
	Ammannia robusta Heer & Regel	6.61			
	Asclepias incarnata L.			0.42	1.52
	Azolla caroliniana Willd.				6.09
	Bidens cernua L.	1.83		1.40	4.40
	Bidens connata Muhl. ex Willd.	0.34			0.47
	Bidens frondosa L.				0.47
	Bidens sp.	1.11		4.77	
	Boehmeria cylindrica (L.) Sw.				0.92
	Cicuta bulbifera L.				2.33
	Cirsium arvense (L.) Scop.			0.42	
	Decodon verticillatus (L.) Ell.				1.68
	Echinocystis lobata (Michx.) T. & G.	0.33			
	Euthamia graminifolia (L.) Nutt.			1.68	
	Galium trifidum L.	1.01			0.50
	Hibiscus moscheutos L.	0.36			0.90
	Impatiens capensis Meerb.	0.98			
	Lindernia dubia (L.) Pennell	2.18			
	Ludwigia palustris (L.) Elliott	3.08		0.57	
	Lycopus uniflorus Michaux	0.67			3.20
	Lythrum salicaria L.		1.38		
	Malva moschata L.			0.42	
	Melilotus alba Medikus			0.42	
	Mimulus ringens L.	1.40		3.01	
	Mosses (general, non-Sphagnum)				1.04
	Nelumbo lutea Willdenow	18.25	41.54	19.61	
	Nymphaea odorata Aiton			0.45	2.40
	Penthorum sedoides L.			1.55	0.95
	Polygonum amphibium L.			22.25	39.58
	Polygonum hydropiperoides Michx.	0.40			
	Polygonum persicaria L.	0.64		0.87	
	Polygonum punctatum Elliott	0.69			
	Ranunculus flabellaris Raf.			1.31	0.45
	Rorippa islandica (Oed.) Borb.	0.39			
	Rotala ramosior (L.) Koehne	0.79		2.32	
	Sagittaria latifolia Willd.	31.79	10.41	0.42	1.97
	Sagittaria sp.			0.42	

	Saururus cernuus L.	1.59			
	Scirpus cyperinus (L.) Kunth				0.45
	Scutellaria galericulata L.			0.45	
	Scutellaria lateriflora L.			2.90	1.47
	Sparganium eurycarpum Engelm.				5.31
	Urtica dioica L.	2.69			1.40
SAV	*Ceratophyllum demersum* L.	1.51		26.20	9.10
	Chara vulgaris L.			0.45	
	Elodea canadensis Michx.			23.85	2.83
	Heteranthera dubia (Jacq.) MacM.			0.98	3.01
	Lemna minor L.	6.36	16.25	0.42	2.75
	Myriophyllum sibiricum Komarov				3.73
	Myriophyllum spicatum L.			8.97	22.51
	Najas flexilis Willd.	0.37		0.85	
	Najas minor Allioni.		4.47	2.44	0.45
	Nitella flexilis L.		1.38		0.97
	Potamogeton crispus L.	0.70	2.77	0.43	
	Potamogeton foliosus Raf.		5.84		
	Potamogeton nodosus Poiret.	17.85	17.36	24.08	13.41
	Potamogeton pectinatus L.	3.08	7.05	6.32	1.44
	Riccia fluitans L.				2.26
	Ricciocarpus natans (Linn.) Corda				2.78
	Spirodela polyrhiza (L.) Schleiden	0.68	4.06	1.27	3.64
	Vallisneria americana L.		3.00		
	Zosterella dubia (Jacquin) Small				0.47
Tree/ Shrub/ Vine	*Populus deltoides* Marshall			1.29	
	Salix cordata Michx.	0.34			14.41
	Salix eriocephala Michx.			0.45	
	Salix exigua Nuttall	1.36			
	Vitis sp.	0.33			
Invasive	*Butomus umbellatus* L.	4.25		0.42	2.33
	Phalaris arundinacea L.	1.77			
	Phragmites australis (Cav.) Steudel	31.23	41.53		
	Typha angustifolia L.	22.90	29.67	0.42	1.42
Grass/ Sedge/ Rush	*Carex comosa* F. Boott.				0.47
	Cyperus erythrorhizos Muhl.	0.34		3.23	
	Cyperus sp.	1.35			
	Cyperus strigosus L.				0.45
	Echinochloa crusgalli (L.) Beauv.			5.19	
	Eleocharis acicularis (L.) R. & S.	14.83	5.96	15.72	1.92
	Eleocharis erythropoda Steud.	2.95			
	Eleocharis obtusa (Willd.) Schultes			0.45	
	Eleocharis ovata (Roth) Roemer & J.A.	2.38		2.36	4.80
	Eleocharis palustris (L.) Roemer & J.A.	1.02		0.87	5.48

	Eleocharis sp.			0.43	
	Eragrostis hypnoides (Lam.) BSP			0.51	
	Eragrostis pectinacea (Michx.) Nees ex Steud.			0.51	
	Juncus nodosus L.	0.33			
	Leersia oryzoides (L.) Swartz	3.55	1.42	1.82	17.47
	Panicum capillare L.			0.42	
	Schoenoplectus tabernaemontani (C.C. Gmelin) Palla		1.42	0.87	0.61
	Scirpus fluviatilis (Torr.) A. Gray	1.39			2.31
Species Richness		**46**	**17**	**50**	**48**

Table 3.11. Summary statistics of water quality data collected in Crane Creek and diked Pool 2B. The 2009 data are courtesy of Josh Eash, U.S. Fish and Wildlife Service.

a) Data collected from the CCLow site from May 5, 2005 – October 24, 2005

Parameter	Min Daily Mean	Max Daily Mean	Daily Mean	SD	Max Daily Range	Date of Max Range	% days < 3 mg/l	% days < 4 mg/l
Temperature (°C)	9.9	29.8	22.1	4.9	11.7	10/1/05	-	-
DO (mg/l)	5.6	15.2	9.3	1.8	16.6	10/04/05	9.0	21.2
Stage (m)	174.1	174.4	174.2	0.1	0.6	08/31/05	-	-
pH	7.9	9.2	8.6	0.2	2.4	10/18/05	-	-
Turbidity (NTU)	22.3	127.5	59.4	21.7	221.3	05/11/05	-	-

b) Data collected from the diked wetland site (Pool 2B) from June 23, 2009 – September 15, 2009

Parameter	Min Daily Mean	Max Daily Mean	Daily Mean	SD	Max Daily Range	Date of Max Range	% days < 3 mg/l	% days < 4 mg/l
Temperature (°C)	18.6	33.2	23.9	2.0	6.7	09/13/09	-	-
DO (mg/l)	0.5	12.5	5.8	1.3	10.4	09/08/09	48.2	74.1
Stage (m)	174.0	174.6	174.4	0.1	0.4	08/29/09	-	-
pH	7.6	9.5	8.5	0.4	1.6	08/08/09	-	-
Turbidity (NTU)	0.1	71.6	19.4	8.6	65.9	09/04/09	-	-

Figure 3.1. Location of the Crane Creek wetland complex in western Lake Erie. 2005 emergent and submersed vegetation noted for CCUp, CCLow, Pool 2A , and Pool 2B wetland sites.

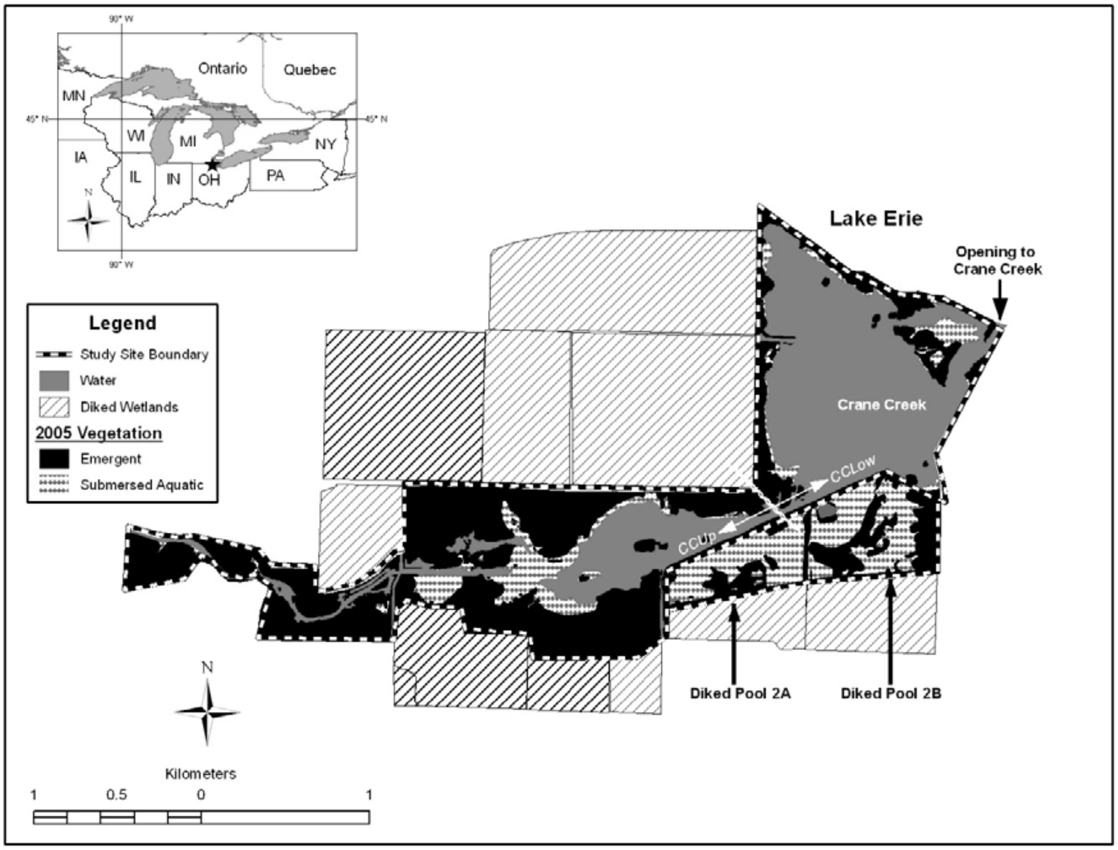


Figure 3.2. Non-metric multidimensional scaling of 2004 and 2005 Crane Creek fish abundance data by species. Fish species only present in the hydrologically-connected Crane Creek sites are underlined. Final 3-dimensional solution stress was 3.57048 after 75 iterations.

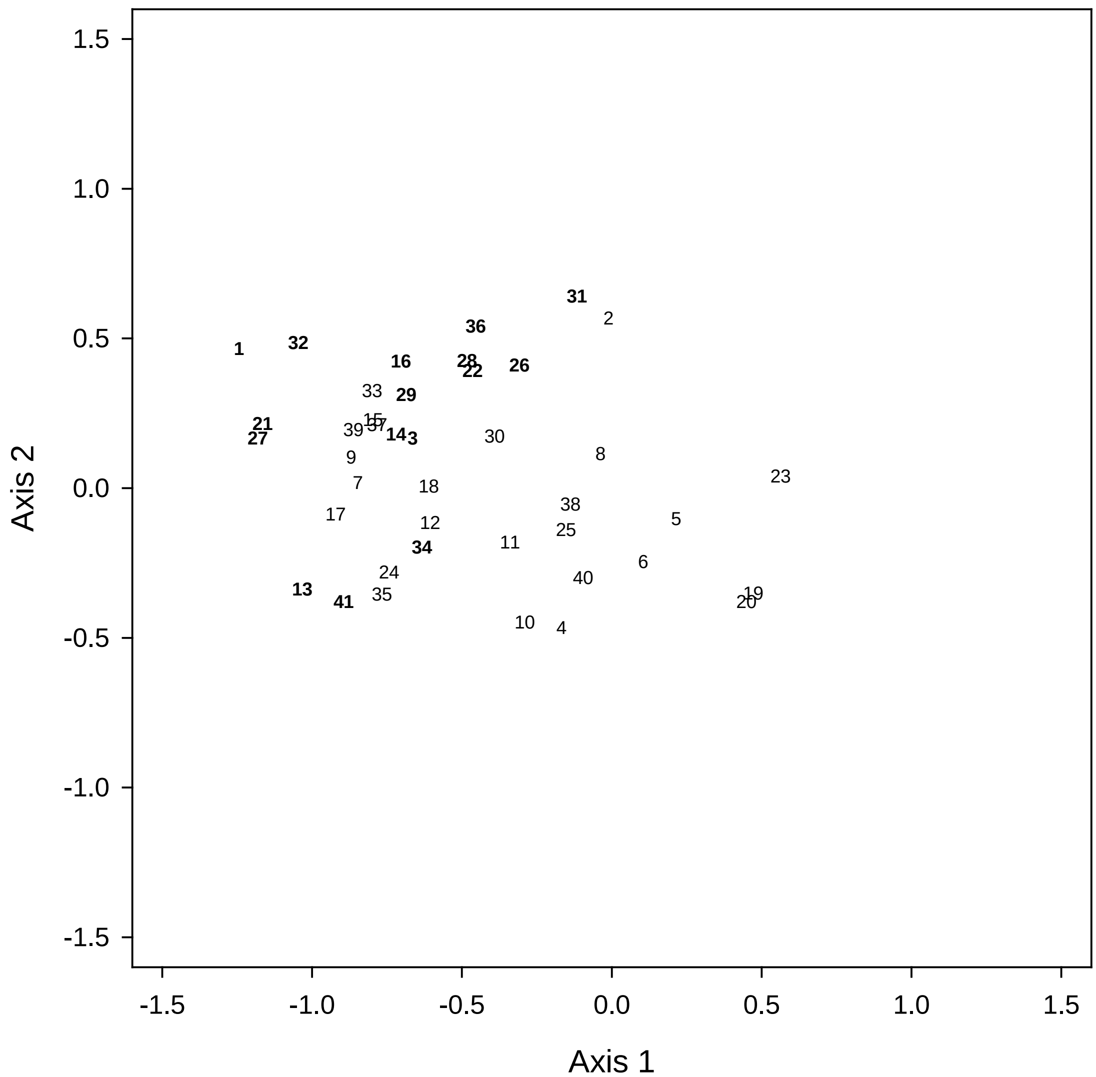


Figure 3.3. Non-metric multidimensional scaling of 2004 and 2005 Crane Creek fish abundance data by site. Circles (●) represent Pool 2A sites, stars (★) represent Pool 2B sites, up-triangles (▲) represent upper Crane Creek sites, and down-triangles (▼) represent lower Crane Creek sites. The data points contain attributes for the sample year (2004 (4) or 2005 (5)), season (spring (Sp), summer (Su), fall (F)), and site (Crane Creek upper (CCUp), Crane Creek lower (CCLow), Pool 2A, Pool 2B). Final 3-dimensional solution stress was 3.57048 after 75 iterations.

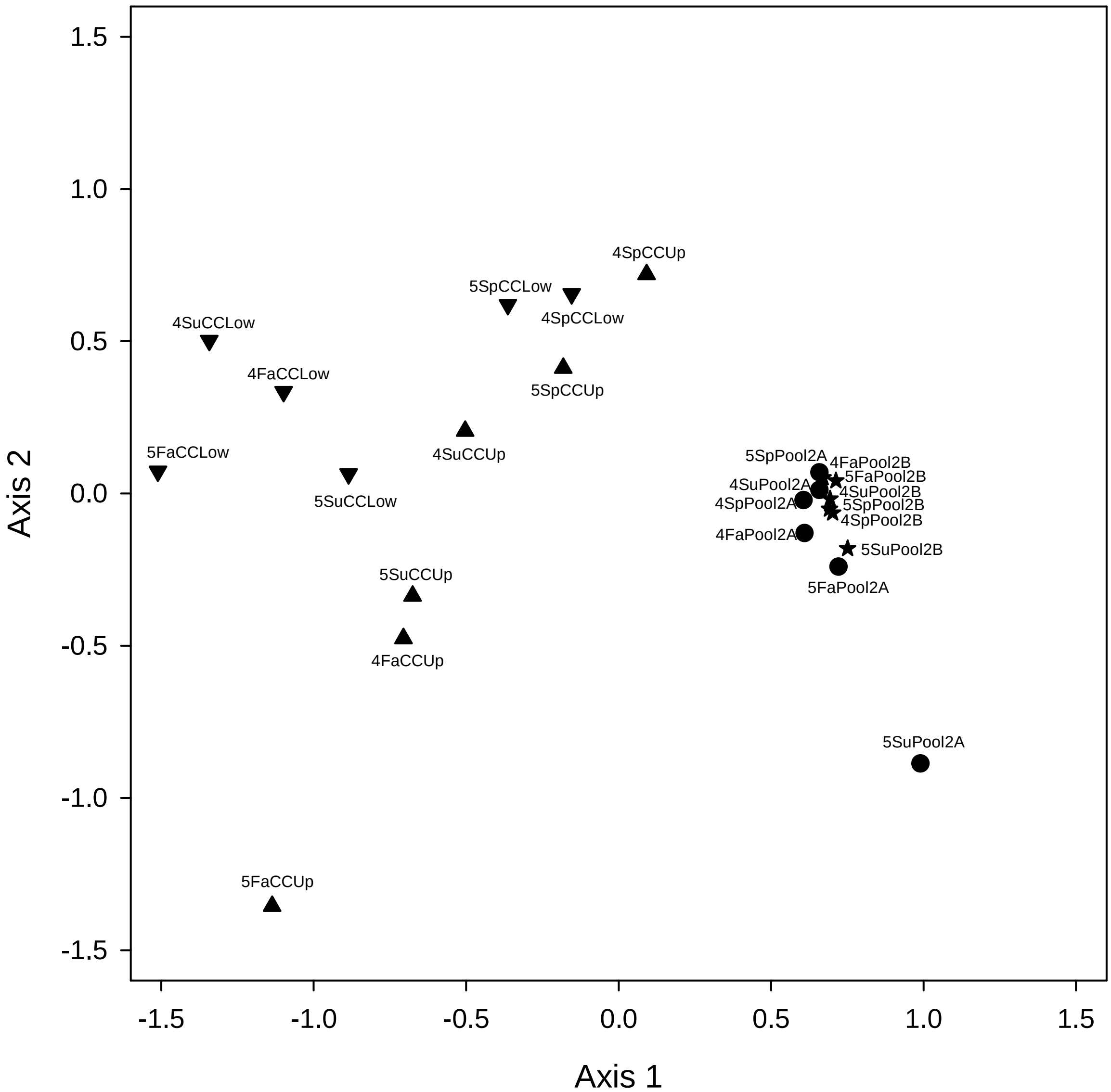
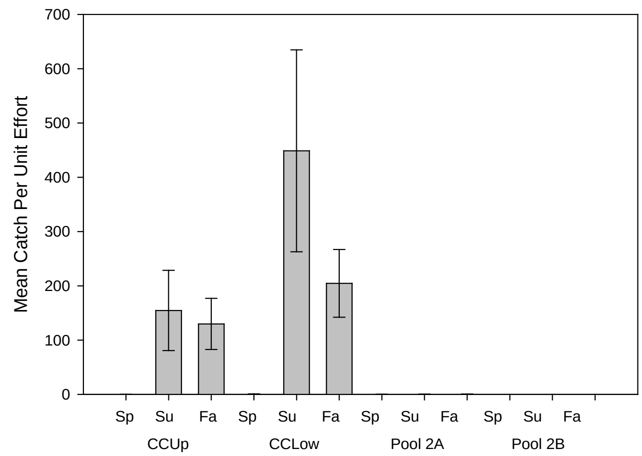
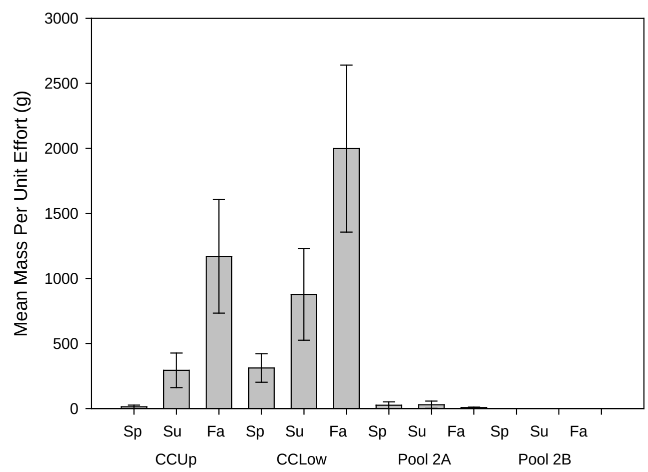


Figure 3.4. Abundance, biomass, and length of gizzard shad separated by site and season. Error bars represent the standard error. 2004 and 2005 data are included. Percent of captured fish longer than 30.5 cm, adult length according to Trautman (1981), noted above bar graphs.

a) Abundance



b) Biomass



c) Percent Adult and Mean Length (cm)

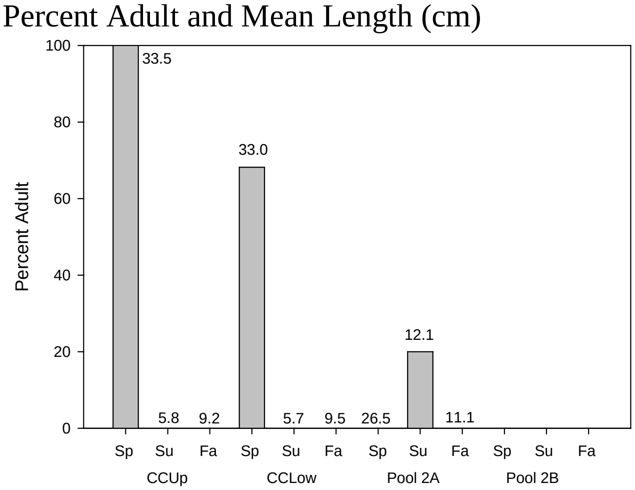
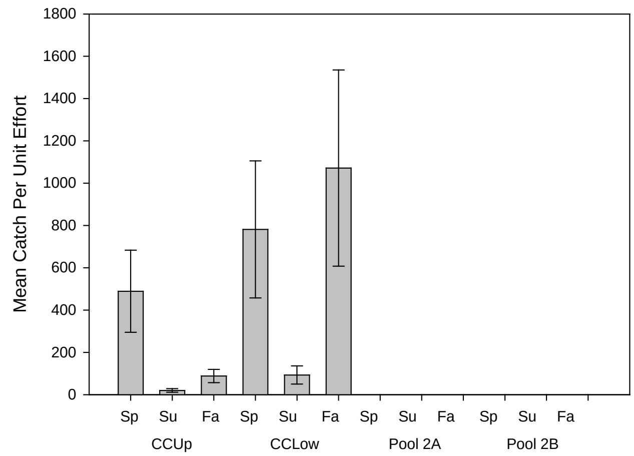
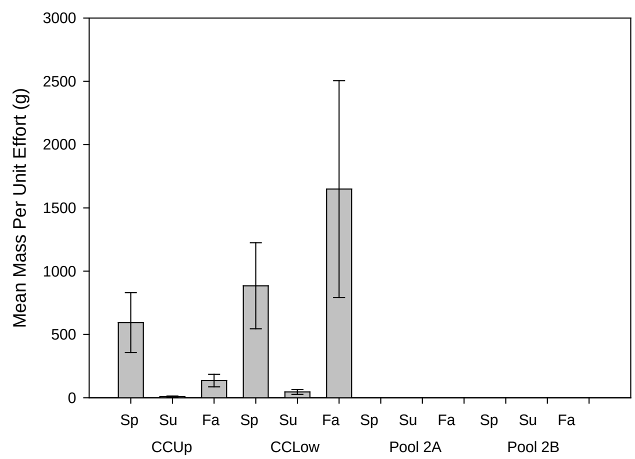


Figure 3.5. Abundance, biomass, and length of emerald shiner separated by site and season. Error bars represent the standard error. 2004 and 2005 data are included. Percent of captured fish longer than 6.4 cm, adult length according to Trautman (1981), noted above bar graphs.

a) Abundance



b) Biomass



c) Percent Adult and Mean Length (cm)

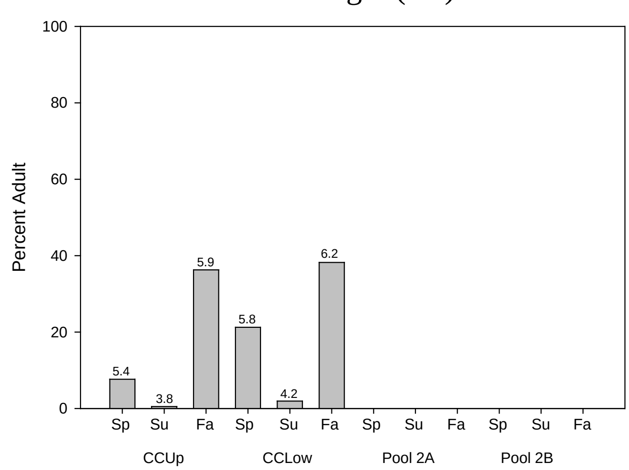
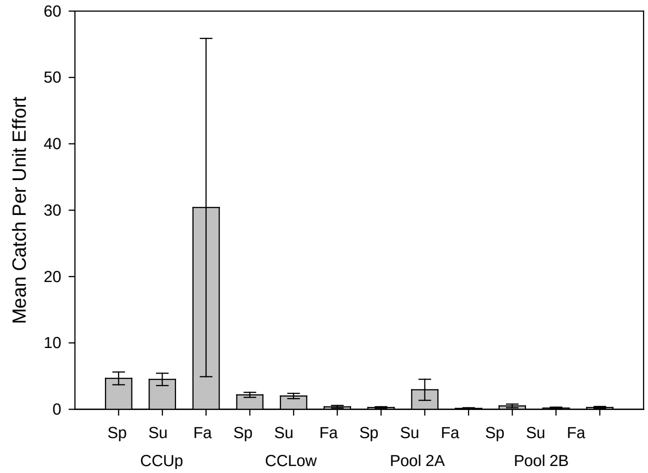
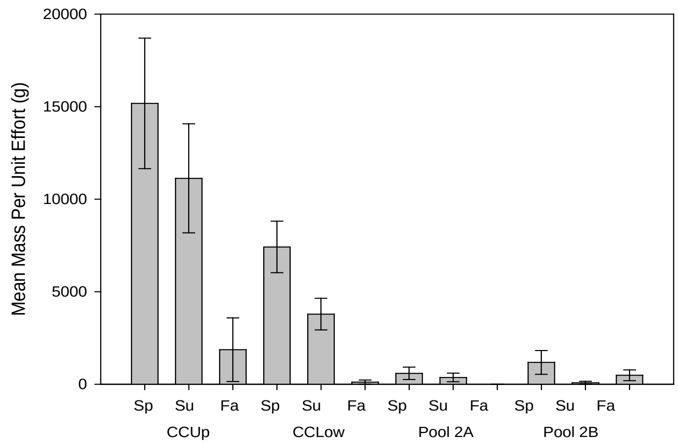


Figure 3.6. Abundance, biomass, and length of carp separated by site and season. Error bars represent the standard error. 2004 and 2005 data are included. Percent of captured fish longer than 30.5 cm, adult length according to Trautman (1981), noted above bar graphs.

a) Abundance



b) Biomass



c) Percent Adult and Mean Length (cm)

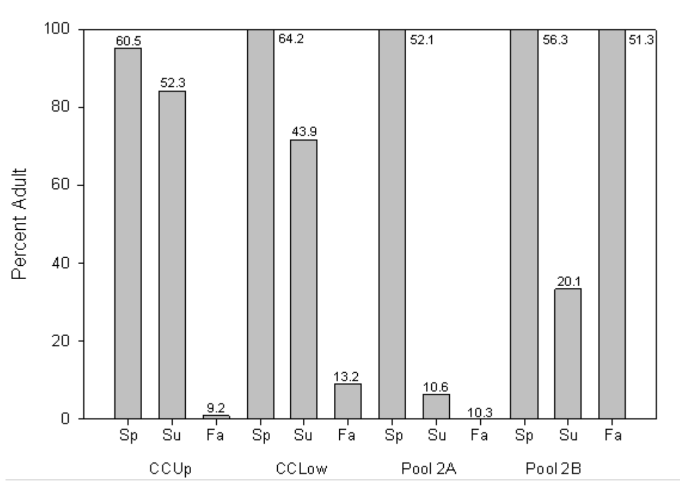
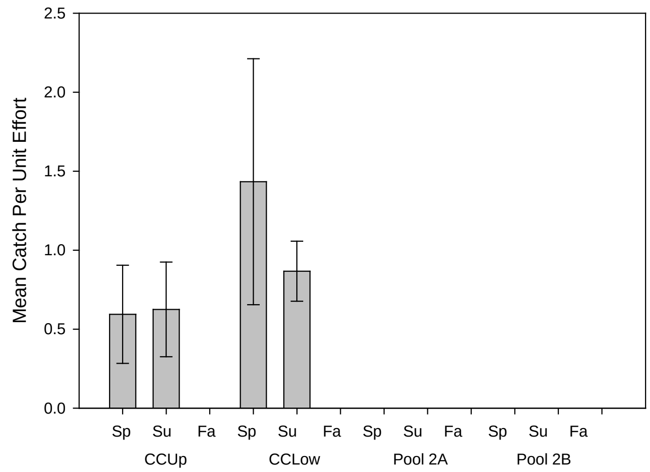
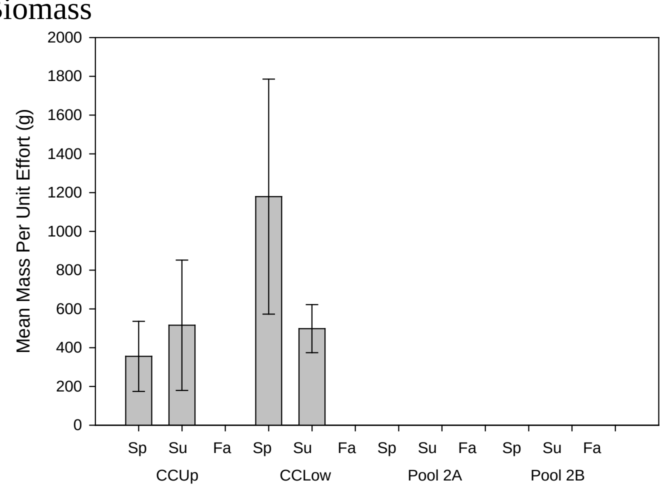


Figure 3.7. Abundance, biomass, and length of longnose gar separated by site and season. Error bars represent the standard error. 2004 and 2005 data are included. Percent of captured fish longer than 60.9 cm, adult length according to Trautman (1981), noted above bar graphs.

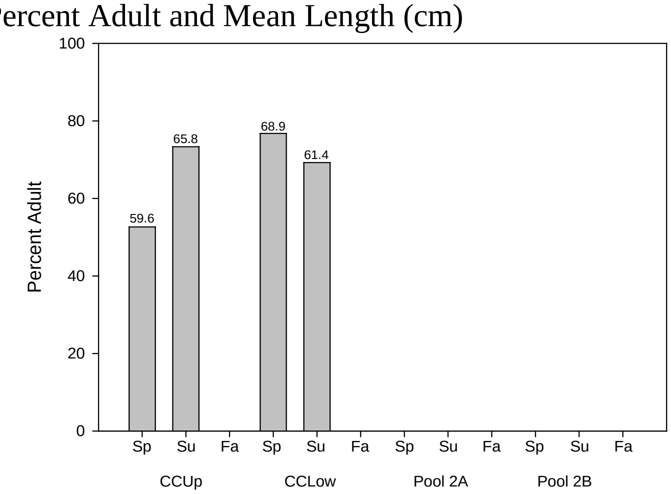
a) Abundance



b) Biomass



c) Percent Adult and Mean Length (cm)



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CHAPTER 4
Fish movement between Lake Erie and a degraded coastal wetland: a first look using an imaging sonar (DIDSON)

4.1 Abstract

Despite poor water quality (e.g. frequent hypoxia) and extensive channel and shoreline modifications, the Crane Creek coastal wetland system in the Ottawa National Refuge supports an abundance of fishes from coastal Lake Erie. Understanding the magnitude and timing of their movements into and out of the refuge wetlands is important from both a fish biology perspective (e.g., understanding the physical cues for fish movement between habitats) and a habitat rehabilitation perspective (e.g., determining when access to wetland habitat is most valuable). A high-resolution sonar (DIDSON) was used to examine the abundance and movement of fishes in a connecting channel between wetland habitats and Lake Erie and characterize relationships between movement rate and abundance with water quality conditions, including water level, flow rate, and water chemistry. In the midst of very dynamic environmental fluctuations, we estimated 5.2 million fish passed through the mouth of Crane creek during our two week study. Approximately 92% were small shoaling fish typically considered forage species for larger predatory fishes. A large number of longnose gar, shoals of small fish, and other unidentifiable large fish appeared to enter and leave the wetlands on a daily basis, with many returning to nearshore Lake Erie during the evening hours to escape diurnally poor water quality. These results support our understanding that coastal wetlands and their connections to Lake Erie serve as important habitats for many species of fishes.

4.2 Introduction

Great Lakes coastal wetlands provide important spawning, nursery, and feeding habitat for a variety of fish species (Cooper 1987, Johnson 1989, Jude and Pappas 1992, Wei et al. 2004, Bouvier et al. 2009; Table 3.1). Some species only use wetland habitats during spawning and for early life stages while others move routinely among wetland habitats throughout their lifetimes and at a range of temporal scales (e.g., seasonally, daily). For example, northern pike, carp, and many other spring spawning species in Lake Erie seek coastal wetland areas early each spring (Becker 1983). Northern pike normally return to Lake Erie immediately after spawning, but most mature carp feed extensively in wetland habitats after spawning and don't leave until water temperatures warm in the summer (Cooper 1987). Diurnal activity patterns of fishes often revolve around food availability or preferred feeding strategies (Helfman 1993, Rypel and Mitchell 2007). For example, longnose gar and black bullhead are reported to feed most heavily at night, often in shallow wetland habitats (Darnell and Meirotto 1965, Becker 1983). Conversely, northern pike are active almost exclusively during daylight hours when they are able to see their prey (Diana 1980). While fishes often move into wetland habitats to reproduce or feed, they must also move between and out of the wetlands when water quality (e.g., temperature, dissolved oxygen) and water levels become problematically low.

Changes in water quality, particularly rapid changes in temperature, pH, or dissolved oxygen can cause fishes to seek new habitats, at least temporarily. Many species of fish have relatively narrow preferred temperature ranges (i.e., thermal niches) and readily move to other areas temperatures are energetically unfavorable (Coutant 1977, 1987). Similarly, diurnal fluctuations in pH and oxygen driven by ecosystem photosynthesis and respiration can often trigger fish movement to sites of higher oxygen availability or less extreme pH (Kramer 1987, Burleson et al. 2001). Fishes also can seek more oxygenated environments to maximize food consumption and growth or gain energetic advantages over prey (Kramer 1987, Jobling 1994). Since water quality conditions can change very quickly in shallow, highly productive systems, the connectivity and access to channeled passage between habitats is likely important for optimal habitat use. Those connections can also offer important opportunities to study fish movement by concentrating fishes in relatively small channels where they can be sampled. However, sampling fishes in these dynamic environments can be difficult (Rozas and Minello 1997).

Traditional fish sampling techniques (e.g., trap nets) can be ineffective if water levels in the wetland fluctuate beyond the gears' operational range. For example, rapidly falling water levels may strand a trap net on a mud flat or rising water levels may overtop a net designed for shallow water, thereby reducing its catch efficiency. Similarly, water levels that create depths appropriate for electrofishing can become unsuitable quickly as water drains from the marsh during seiche events (i.e., wind-driven water level changes in Lake Erie). In addition to high current velocities, debris flowing through the channel during large seiche events can dislodge trap or gill nets set in the channel. Even if traditional techniques are implemented properly, they may not allow analysis of the density or flux of fishes moving into and out of the coastal wetland over a short time frame as water quality and physical conditions change.

Since wetland fishes can move frequently in response to dynamic wetland hydrology and chemistry, high spatial- and temporal-resolution data are needed to characterize the fish assemblages. Dual-frequency identification sonar (DIDSON; Sound Metrics, Inc., Lake Forest Park, WA) is an acoustic camera that uses high-frequency (1.8 MHz) sound waves to collect image data without light and can be used in turbid or dark water where other techniques (e.g., underwater camera, visual observation) often fail. The technology has been used to detect large fish in mangrove habitat (Frias-Torres and Luo 2009), but most fisheries-related applications have focused on quantifying the number of salmonids moving upstream in large rivers (Tiffan and Rondorf 2004, Burwen et al. 2006, Maxwell and Gove 2007), fish passage through turbines (Weiland and Carlson 2003), fish response to trawls (Graham et al. 2004, Handegard and Williams 2008), or fish under ice (Mueller et al. 2006). However, the DIDSON technology had not been used to study fish assemblages in the Great Lakes. We used a DIDSON camera to look at fish movement between Lake Erie and a degraded coastal wetland complex.

The objective of this study was to examine short-term patterns of fish abundance and movement in a channel connecting the Crane Creek coastal wetland complex and Lake Erie. Specifically, we sought to characterize the patterns (e.g., timing, direction) of fish movement between the wetland and Lake Erie, identify correlations between fish movement and changes in water quality and channel hydraulics, and estimate the abundance of fishes using the wetland habitats and moving through the channel on a daily basis.

4.3 Materials and Methods

4.3.1 Study Area

Crane Creek is a small tributary to Lake Erie that flows eastward through a 146 km^2 agricultural watershed and terminating in a partially diked coastal wetland complex within the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge (ONWR; 41.628611° Latitude, -83.207778° Longitude). The refuge wetlands are located along the southern shore of western Lake Erie approximately 48 kilometers southeast of Toledo, Ohio, USA (Figure 4.1). Agricultural field tiling and ditching are common in the low gradient watershed (0.359 m/km; Ohio Department of Transportation 1987), but fringe emergent wetlands dominate the lowest reaches where Crane Creek enters the refuge approximately 6 km upstream from Lake Erie (Kasat 2006). Connection with Lake Erie is made through a short permanently open revetment-lined channel approximately 100-m long and 50-m wide, with a variable depth that can exceed 4 m in places. Most of the wetland area above the connecting channel is dominated by deep deposits of silty sediments, but areas supporting higher water velocity (e.g., channel thalweg) have sand or hard-pan bottom (Bowers 2003).

Water-surface elevation, direction, and velocity of flow in the channel and connected wetlands vary with the dynamic interaction between Lake Erie water levels and Crane Creek watershed discharge. Short-term, wind-driven water-level oscillations (i.e., seiches) in Lake Erie often have an amplitude between 0.7 m and 2 m and can exceed 3 m during extreme storm events (Herdendorf 1987). Depending on the water-surface elevation (WSE) of seiche (e.g. rising water levels in the western basing of Lake Erie), the hydraulic energy slope of the water between Lake Erie and Crane Creek can push water rapidly into or out of the lower Crane Creek wetland complex.

Agricultural practices in the watershed and point-source discharges into the creek contribute to large nutrient loads and concentrations of nitrate, ammonia, and soluble reactive phosphorus, high turbidity, and poor overall water quality in Crane Creek (Kasat 2006; Table 3.11). Extensive mixing of river and lake water occurred in the wetland when clearer Lake Erie water was driven into the wetland, even though water depth in most of the wetland was less than 1 m deep during the study. Although very sparse, submersed vegetation in Crane Creek was dominated by pond weeds (e.g., *Potamogeton nodosus*, *Potamogeton pectinatus*, *Potamogeton foliosus*; Table 3.11). Patches of floating-leaf assemblages (e.g., *Nelumbo lutea, Lemna minor*) were present throughout the wetlands, and emergent vegetation was dominated by *Typha angustifolia* and *Phragmites australis* located along the perimeter of the marsh.

4.3.2 Sampling Sites

For this study, a DIDSON acoustic camera was placed in the connecting channel near the mouth of Crane Creek (i.e., the interface between Crane Creek and Lake Erie: see CCCH, Figure 4.1) along with water quality monitors and a pulse coherent acoustic Doppler profiler (PC-ADP; Sontek Inc., San Diego, CA). Water-quality data and stage also were collected in coastal Lake Erie just up lake current from Crane Creek (called the Erie Nearshore station: ENS), in the submersed wetland zone approximately 770 m upstream from the connecting channel (called Crane Creek lower wetland: CCLW), and approximately 6 km upstream from the connecting channel where Crane Creek enters the refuge (called Crane Creek upper wetland: CCUW). Data for the open waters of Lake Erie were obtained from a National Oceanic and Atmospheric Administration gaging station #9063085 located in the Toledo, Ohio harbor (41.683333° Latitude, 83.466667° Longitude).

4.3.3 DIDSON Sampling

A DIDSON acoustic camera was used to estimate fish density and passage in the connecting channel between lower Crane Creek and coastal waters of Lake Erie (i.e., site CCCH). The unit produces video-like acoustic data regardless of light intensity and turbidity, providing detailed records of fish behaviour and density changes over time. The data in this study were collected using a unibody DIDSON attached to a pan/tilt module manufactured by Remote Ocean Systems (San Diego, CA). The DIDSON supported 2 main modes: detection mode (low frequency; 1.1 MHz) and identification mode (high frequency; 1.8 MHz). The identification mode was used during this study because the highest resolution images were desired and the sonar was stationary during data collection. The unit's 96 beams are oriented 0.3° apart horizontally and 12.0° vertically and are able to generate frames (i.e., images) from a distance of 4.0 m to 14.0

m from the sensor. The down-range pixel size is a function of the distance from the sensor divided by 512, and the cross-range pixel size also varies with distance from the sensor (0.5R/100 where R is the range from the sonar; Maxwell and Gove 2007). Our study collected data from 4.0 m – 14.0 m downrange from the sensor, so the pixel sizes ranged from 0.8 cm – 2.5 cm down-range and 2.0 cm – 7.0 cm cross-range. Depending on window length, cable length, and other factors, the DIDSON can generate up to 20 frames/second in its 29° horizontal field-of-view. A cable from a topside computer to a deployed DIDSON supplied the 30 w of power required to operate the unit and transfer data collected by the sonar back to the computer.

The DIDSON unit was mounted on a 7.6 cm diameter aluminum pipe cross bar attached between two equivalent size pipes anchored into the sediment. The DIDSON was positioned in the middle of the water column of Crane Creek approximately 0.6 m below the water surface 15 m from the south shore of the narrowest part of the creek's connection with Lake Erie (see Figure 4.1). The DIDSON was aimed on an angle toward the bottom and perpendicular to water flow to image the side of fishes moving with or against the current in the channel yet collect data from the sediment surface to the water surface. A small section of the thalweg was not able to be imaged by the sonar because it was approximately 10 cm deeper than adjacent areas. Since fishes often move through the deepest sections of a channel, a one meter wide fyke net lead with 1.27-cm knotted mesh was installed in the thalweg perpendicular to channel flow. Fishes swimming through the thalweg were imaged as they swam over the net. Two USGS divers checked the deployment of the DIDSON to ensure it was angled properly and to ensure that the fyke net in the thalweg was operational. The water volume sampled was a cone approximately 62 m^3; equivalent to about 44 areal square meters (see Figure 4.1 for graphical representation).

A topside laptop computer was set up in a 3-m long box trailer parked on the earthen dike immediately south of the DIDSON installation and connected to the DIDSON through a 30-m cable. Power to run the DIDSON and laptop computer was supplied by two 12 V DC batteries and a sine-wave inverter used convert the current to AC. Sound Metrics' software was used to acquire and record data to the computer's hard drive. The DIDSON software was set to determine the optimal sampling rate automatically (e.g., 6 frames per minute) during high frequency data collection. Intensity and threshold settings were left in their default position.

The acoustic imaging study began at 11:00 am on July 26, 2007 and was terminated 13 days later at 11:00 am on August 8, 2007. Data recording was continuous except during three periods when the DIDSON was shut down because of severe weather or maintenance issues (10:30 pm – 11:45 pm July 26, 2:45 pm July 27 – 12:30 pm July 30, 10:30 am August 4 – 1:00 pm August 6). Data were written into hourly files beginning and ending on the top of the hour, recorded on the hard drive, and backed up to a remote location regularly.

4.3.4 DIDSON Data Analyses

Most hourly files recorded during this study were over 1 GB in size and contained approximately 22,000 data frames that, when viewed in quick succession, appeared similar to data collected with an optical video camera. Quantification and interpretation of the data required extensive manual interpretation (i.e., a person viewing the data frames and recording events) since hydroacoustic software capable of processing DIDSON data was not available at the time of this study. We visually examined and manually coded the entire data record using a stratified random sampling approach. Individual fish greater than approximately 10 cm long were visible in the data set, but only longnose gar were consistently identified to species. Therefore, density and movement data for large individually observed fishes were reported either as gar or "other fish". Shoals of small prey fish (e.g., emerald shiner, gizzard shad) also were clearly identifiable in the DIDSON data and recorded separately. Adjacent fyke and gill net sets were used to identify further the species being recorded by the acoustic camera (see below). Characteristics used to distinguish gar from other taxa included body shape (gar are long skinny fish), tail beat rate (gar have a very rapid tail beat rate), and behavior (gar generally did not linger in the frame). Shoals of fish were identifiable because of their shape (large masses of small fish) and behavior (predator avoidance). All other large fishes were grouped into the Other Fish category because they were not identifiable to species. The direction of water flow in the channel (i.e., in or out) and any other noteworthy observations also were recorded.

A total of 4,004,094 frames of DIDSON data were collected and analyzed during this study. The analysis focused on the density and rates of movement of fishes in the Crane creek connecting channel. The entire data set was first viewed manually at a rate of approximately 20 frames / second and scored for fish “events” by trained personnel. The analysts recorded for each of the three data classes (Gar, Other Fish, Shoals) observed data “events” as either “swimming in”, “swimming out”, or “bi-directional” events (i.e., no net flux observable) while noting the relevant times and frame numbers within each hourly file. Each data event was also assigned a qualitative density estimate (i.e., low, average, high). Although the length of time that fish were observable in the data varied, all observations were then summarized by quarter hour periods (called time units henceforth) to facilitate analysis.

4.3.5 Stratified Sampling

The categorical density coding for gar was used to stratify the entire data record and allocate random sampling efforts. A sample was defined as a 250-frame clip of data (approximately 40 seconds long), the smallest time period we felt we could interpret as an independent observation (i.e., length of time required for a gar to typically move through the field-of-view). Two hundred two random samples (i.e., 50,500 data frames) were drawn proportionally from the four strata based on the total number of frames in each strata. The number and direction of movement of gar, other large fish, and shoals were recorded for each sample. The mean, sum, and standard error of gar, other large fish, and shoals were calculated for the 40-second samples analyzed in each stratum. These data were converted to fish / minute and used to estimate the total number of gar, other large fish, and shoals per strata moving into the wetland, out of the wetland, and net movement in or out (i.e., in – out).

4.3.6 Ancillary Water – Quality and Fyke – Net Sampling

Two fyke nets with 91-cm x 91-cm frames and both 0.48-cm and 1.27-cm standard knotted mesh were used to sample quantitatively (i.e., ground truth) fishes observed with the DIDSON. The two nets were fished for a minimum of one-hour at least once per day and more frequently during times of unusual fish movement. The nets were placed on the marsh side of the DIDSON facing the shore in water approximately 1 m deep with 6- to

15-m leads perpendicular to and reaching shore and 3-m wings extending to each side. An experimental gill net composed of five panels (each approximately 7.2 m long, 1 m tall) with mesh sizes 15.2, 12.7, 10.2, 7.6, and 5.1 cm was set diagonally across the channel and checked for fish within 1 – 4 hours. A 1-m tall gill net composed of two 30-m long sections (one with a 25.4 cm mesh and the other with a 20.3 cm mesh) was set on August 1, 2007 at approximately 10:30 am and checked for fish at 3:30 pm. After all net deployments, fishes were identified, counted, measured for length, and released.

The PC-ADP was mounted on an inflatable pontoon anchored in the Crane Creek channel very near the DIDSON in CCCH. The downward-looking PC-ADP collected and averaged velocity data for the entire water column for approximately 1 minute every 15 minutes. Data were transmitted to the topside laptop via spread spectrum radio. Positive velocity values indicated water was flowing from Crane Creek into Lake Erie and negative values indicated that water was flowing from Lake Erie into Crane Creek.

A YSI model 6920 water quality sonde (YSI Incorporated, Yellow Springs, OH) was stationed in the lower water column of Crane Creek in three locations at the study site: in the marsh approximately 770 m upstream from the channel (CCLW), in the channel adjacent to the DIDSON deployment (CCCH), and approximately 100 m into the nearshore zone of Lake Erie upcurrent from the junction with Crane Creek (ENS). Five parameters (i.e., dissolved oxygen (DO; mg/l), temperature (°C), turbidity (NTU), pH, and water levels (m)) were collected from each sonde in 10-minute intervals from July 26, 2007 through August 8, 2007. A Odyssey water-level recorder (Dataflow Systems PTY Limited, Christchurch, New Zealand) was stationed in Crane Creek where it crosses Route 2 (CCUW) and set to collect data during the study. Data from the sondes were downloaded to a laptop daily, and standard procedures were used to calibrate the sondes before data collection started, once during the study period, and once after the study was completed. Outlier data points in the water quality data set were removed if they were greater than two standard deviations away from the mean or if external information suggested their removal (e.g., data collected when the sonde was out of the water). The resulting data were plotted as time series data to identify patterns among the data and allow comparisons with results of the DIDSON analysis. Land-surveying techniques were used to determine the elevation of the gages and convert water-depth data to elevations reported in International Great Lakes Datum 1985. Finally, hourly weather data, recorded at the Toledo airport, were downloaded from the internet (e.g., http://www.wunderground.com/history/airport/KTDZ/2007/7/24/DailyHistory.html?req_city=NA&req_state=NA&req_statename=NA).

4.4 Results

4.4.1 Hydrology and Hydraulics

Lake Erie water levels exerted a controlling influence on both stream flow and water depth during the study period. When the WSE of nearshore Lake Erie was greater than the WSE in Crane Creek (CCUW) due to seiche events, the normal downstream slope of the water surface was reversed, causing lake water to flow from Lake Erie back into the creek channel and upstream. Lower Crane Creek and hydraulically connected adjacent coastal wetlands then filled with water until the slope of the water surface approached zero. Water levels in the creek, the connecting channel, and the coastal wetland complex were at their highest during these seiche-induced backwater events (e.g., hour 282 in Figure 4.2 and Figure 4.3). As Lake Erie levels fell, water stored in the lower creek system resumed its flow lakeward as the slope between Crane Creek and Lake Erie water levels was again reversed. The velocity of the water flow at any point in time was dependent on the magnitude of the hydraulic slope (e.g., larger slopes drove higher velocities; see Figure 4.2). A cycle of filling and emptying and associated fluctuations in water level was repeated throughout the study period as seiche-driven water levels moved up and down with a 12 – 14 hour periodicity that is typical for Lake Erie (Herdendorf 1987, Trebitz 2006), especially when large pressure changes and storm events are infrequent (Figure 4.4).

The intensity and duration of flows from Lake Erie into the drowned river-mouth system were quite similar to the flows from Crane Creek into Lake Erie, despite several small rain events adding water to the watershed early and late in the study (Figure 4.5). Water levels in Lake Erie pushed water into the Crane Creek system for a total of 82.75 hours during this study. In contrast, water flowed back into Lake Erie for a total of 82.50 hours. At our channel monitoring site (CCCH), the maximum water velocity out to Lake Erie was 39.9 cm/s. Maximum flow velocities into the marsh were slightly larger than those coming out (45.6 cm/s). Mean velocity out was 13.7 cm/s (standard error = 0.5), and mean velocity in was 12.5 cm/s (standard error = 0.4).

4.4.2 Variations in Water Quality

Temperature, DO, pH, specific conductivity, and turbidity at all three monitoring sites (CCLW, CCCH, ENS) varied from hour to hour with changes in time of day and backwater flow dynamics. The differences in day and night radiation levels drive changes in temperature and photosynthesis, which cause changes in many other variables including DO and pH. Water temperatures across the study area varied within the range of 22.38 – 33.27 °C (Table 4.1) with the warmest temperatures occurring near 6:00 pm each day and the coolest temperatures occurring close to 7:00 am each morning (Figure 4.6) following similar fluctuations in air temperature (Figure 4.7). However, changes in the water temperature lagged changes in the air temperature. The temperatures among all sites tended to rise and fall in at similar rates and during similar times of the day, but the CCLW site was consistently warmer than the ENS site (Figure 4.8). The warmest temperatures in the Channel occurred between the late afternoon and early evening and often were greater than the other two sites. All sites had similar temperatures during their early morning lows, except for a short time (hours 156 – 204) when the ENS site remained cooler than the other sites and had the greatest consistent difference with the CCLW site (Figure 4.8). The temperature at CCCH also varied slightly depending on the source of water flowing through it (i.e., Crane Creek or Lake Erie). The temperature typically rose slightly when the warmer creek water was flowing into Lake Erie, regardless of the time of day.

Although the diurnal dissolved oxygen concentrations ranged from hypoxic to supersaturated at all sites, the mean pH did not differ among sites by more than 0.29 pH (Table 4.1). The mean DO concentrations at CCLW (8.44 ± 0.11 mg/l) and ENS (8.81 ± 0.08) were similar, but the higher velocity water at CCCH led to the highest mean DO concentration (9.30 ± 0.07 mg/l). The greatest concentrations of DO and largest pH commonly occurred between 4:00 pm – 6:00 pm, and the lowest concentration generally occurred between 5:00 am – 7:00 am. The CCLW site had the greatest daily range in both DO and pH (an order of magnitude). DO at CCCH was generally between the levels at CCLW and ENS, except during times when the velocity of flow at CCCH was the greatest (e.g., Figure 4.9, hours 276-324). During these times of high flow, DO and pH at CCCH was usually higher than the other sites. At any given time, there usually was a large difference between DO concentrations at CCLW and ENS, but neither site had consistently higher or lower concentrations (Figure 4.10). The pH among sites followed a diurnal pattern and tracked each other well (Figure 4.11). The differences in pH between ENS and CCLW were varied, but the ENS site was nearly always more alkaline (Figure 4.12).

The mean specific conductivity ranged from 0.29 uS to 0.31 uS among the sites (Table 4.1). The specific conductivity at CCLW had the most variation and showed a connection with water flow (Figure 4.13). It was the highest when water was flowing from the CCLW to the ENS site and peaked shortly after the water flow reversed direction. The peak was followed by a decrease in specific conductivity as water from the ENS site mixed with Crane Creek water and pushed upstream.

Turbidity was highly variable within each site and among the sites (Table 4.1, Figure 4.14). The CCLW site had the lowest mean turbidity and the smallest range in values (3.70 – 70.45 NTU). The flowing water in the channel had the largest recorded level (234.40 NTU) and had the greatest mean turbidity level that was nearly 50% more than the lowest mean level. Except for the greatest difference in turbidity between the ENS and CCLW sites (70 NTU) that occurred near hour 295, the two sites normally differed by less than 25 NTU (Figure 4.15).

4.4.3 Fish Density Observed in Connecting Channel

Given the relatively small size and poor water quality of Crane Creek, the abundance of fish at CCCH during the course of this study was surprisingly high (Table 4.2). Based on the random sampling of 40-secong intervals, the number of larger individually detectable fish inside the DIDSON sampling area ranged from 17 to 67 with an average of 36 over the entire the study period. Since the DIDSON beam sampled a cone-shaped area of approximately 62 m^3, with a 2-dimensional projection of approximately 44 m^2, we estimate that the average density of fishes > 10 cm total length was approximately 0.81 fish / m^2 (8,100 / ha) of channel bottom (Table 4.3). Longnose gar, the only species of fish easily identifiable in the data images, had a mean abundance of 0.08 fish / m^2 and ranged from 0 to 0.20 fish / m^2. The many other species of larger fish, taken together, were much more abundant than the gar (Table 4.2). Their density ranged from 0.38 to 1.31 fish / m^2, with an average density of 0.73 fish / m^2. Shoals of small fish were also very abundant. These were treated in the DIDSON analysis as individual entities and were present in 33% of the sample frames with an average density of 0.05 shoals / m^2. Based on a random sampling of 40-second data intervals used to estimate abundance, the number of individuals per shoal varied widely ranging from 38 to 1,157 with a weighted mean group size of 184 fish. This would imply an average density for small shoaling fishes in the connecting channel of about 9.22 fish / m^2 (92,200 / ha; Table 4.3).

Because identification of fish to species level was in most cases impossible with the DIDSON data, fyke and gill net samples near CCCH were taken to help clarify species composition. Combined, these gears captured 20 species of fish while the DIDSON was collecting data (Table 4.4). Fourteen species of fish were generally long enough to be individually visible in the DIDSON data (e.g., 10 cm) but many smaller species (e.g., emerald shiner, largemouth bass, spottail shiner) likely were visible only when travelling as shoals.

4.4.4 Fish Movement Through the Connecting Channel

Most of the fishes observed in the channel were using it as a corridor for movement between Crane Creek and Lake Erie. Based on densities of in and out-going individuals in the sample frames, during this study we observed a total influx of 243,976 individuals into the Crane Creek wetland complex from the Lake Erie nearshore (30,689 / day) and an outflux of 23,927 / day to Lake Erie (Table 4.5). The difference (6,762) represents a net accrual of 53,757 fishes in the Crane Creek wetland complex over the study period. Of these, an estimated 7,426 were longnose gar. Shoals of small fish likewise moved in and out of the Crane Creek wetland, but the number of shoals entering and leaving were similar (not statistically different). The net flux during the study, based on average rates, was slightly negative (i.e., net movement towards Lake Erie).

These transits in and out from Lake Erie varied over time and with respect to fish density. For example, longnose gar and other large fish were observed moving into the marsh (in), out to Lake Erie (out), or in both directions (bidirectional) during at least 84% of the time units analyzed (Table 4.6, Table 4.7, Figure 4.16, Figure 4.17). Most of the Gar observations (293 time periods; 37% of the total) occurred when gar were at a low density, yet gar were observed in high density 20% of the time (Table 4.6). Most of the high density periods (52%) occurred as gar moved toward Lake Erie, often when the water current in the channel was flowing onto Crane Creek (65% of the time gar were moving out as seiche water was entering; Table 4.8). Most high density periods of gar moving out to Lake Erie occurred between 9:00 pm – 11:00 pm each day (Figure 4.18) and often were followed by an average density of gar moving back into the marsh over the next 7 hours (Figure 4.19). The length of time that gar moved in or out before shifting direction (i.e., duration of movement in a direction) or changing density was variable, and gar movement occurred at all times of the day.

The density and movement of other large fishes was much less dynamic than for gar. In fact, these fishes were present at average or low density 82% of the time and did not appear to change movement patterns in response to the direction of water flow in the channel (Table 4.9, Figure 4.17). Both gar and other large fish moved into the marsh at average or low densities more often than they moved in high densities. However, gar movement out of Crane Creek was most often in high densities while other fish moved out in average densities most of the time (Table 4.7). Finally, “other large fishes” most often moved bidirectionally (66% of the time) at all hours of the day, generally entered Crane Creek in the evening and exited during the day, and did not show a preference for periods when water was flowing into or exiting the marsh (Table 4.9).

Individual shoals of smaller fishes were very actively moving through the channel in 33% of the DIDSON samples (46% of the high Gar density samples and 32% of the average Gar density samples), but they had a very different pattern of activity than the Gar (Figure 4.20). Overall, the shoals did not move through in high numbers (i.e., number of shoals) very often (10% of the time; Table 4.10). The shoals generally moved at low to average densities. There was no correlation between activity rates and flow direction (Table 4.11), however most movement did occur during daylight hours.

4.5 Discussion

Despite very dynamic hydrologic, chemical, and physical conditions, the Crane Creek coastal wetland system supported an abundance of fishes from coastal Lake Erie. Understanding the magnitude and timing of their movements was important from both a fish biology perspective (e.g., understanding the cues for fish movement) and a habitat rehabilitation perspective (e.g., determining when access to wetland habitat is most valuable). However, characterizing and quantifying the dynamic fish use of wetland habitats has been a significant challenge given the limitations of common fish sampling techniques (e.g., fyke nets, electrofishing). This study used a new technology (DIDSON acoustic camera) to assess short-term variations in fish abundance and movement directly in the turbid waters of Crane Creek and begin explorations of relationships between fish habitat use and dynamic water quality conditions characteristic of coastal wetlands.

4.5.1 Fish Assemblage Composition and Abundance

Although little submersed aquatic vegetation was present and water quality was poor, the highly productive Crane Creek coastal wetlands provides habitat for over 53 species of Lake Erie fish species (see Table 3.1). Many of those species were observed during this study as they moved through the connecting channel, often in very high densities (Tables 2 and 3). Several species of large fish (e.g., carp, channel catfish, goldfish, longnose gar) were more easily identified in the DIDSON data because of their size, body shape, and swimming motion. Some of these individuals were visible for extended periods of time, suggesting that they were using the channel itself as habitat rather than just as a passageway between Crane Creek wetlands and Lake Erie. In contrast, most large fish were only visible for short periods of time as they moved through the field-of-view (e.g., longnose gar). These fish were visible frequently during the study, reflecting their large overall abundance.

Shoals of small prey fish, likely emerald shiner or gizzard shad, showed a pattern similar to the individually identifiable large fish. They were observed in moderate to low densities for short periods of time as they moved in between Crane Creek and Lake Erie (Figure 20). Because of their small body size was less than the resolution of the acoustic camera, individual fish within the shoals were very difficult to quantify. However, we were able to create a rough estimate of the number of fish within individual shoals by examining the relationship between the surface area of a representative fish within a shoal and the total area the shoal covered within the DIDSON field-of-view. The range in number of fish composing the shoals was great, but the average number of fish in a shoal observed in the random 40-second samples analyzed during this study ranged from 38 to 1,157 (mean = 184). Assuming the 184 fish / shoal is a reasonable estimate, then the mean density of prey fish was approximately 9.22 fish / m^2 over the course of this study, an order of magnitude greater than all of the large-bodied fish combined. Based on this density estimate and the flux estimate in Table 4.5, we hypothesize that over 4.8 million shoaling prey fish went in and out of Crane Creek during this study yielding a net flux of approximately 1 million small fish moving from Crane Creek into Lake Erie. The connecting channel was the critical link between the degraded, yet highly productive, Crane Creek wetlands and Lake Erie that allowed large assemblages of both small prey fish and large-bodied fish to access wetland habitats throughout each day of this study (Jude and Pappas 1992; Figure 4.16, Figure 4.17, Figure 4.20).

4.5.2 Dynamic Fish Movements and Habitat Conditions

The connecting channel was well used as a movement corridor by both large-bodied fish and shoals of small fish. By estimating the flux of fishes (i.e., number per time) in the DIDSON field-of-view, we were able to characterize both the rate and net movement of fishes moving between the Crane Creek wetlands and Lake Erie on a daily basis (Table 4.5). It is not clear whether they were causally related, but the mean flux of gar both in and out of Crane Creek was similar to the mean flux of shoals moving in and out of the creek. However, the net movement of shoals was out (0.48 ± 0.02 shoals / minute were moving toward Lake Erie) while the net movement of gar was in (0.65 ± 0.05 fish / minute were moving into Crane Creek). These results suggest that an estimated 10,000 shoals (as many as 1,844,000 individual small fish) and 25,000 gar moved into the Crane Creek habitat during the study while approximately 15,000 shoals and 17,000 gar moved out to Lake Erie (Table 4.5). The net movement of shoals out of Crane Creek may have been in response to increasingly harsh water quality conditions in the creek during mid-summer (Table 4.1), a transition to a new life stage associated in Lake Erie, or possibly increased density of predators including the gar observed entering Crane Creek. Similarly, net increases in gar in the wetland may have been a response to prey fish density or predatory advantages associated with their ability to tolerate low oxygen conditions (Scott and Crossman 1998). Fishes in the Other Fish category were operating at an order of magnitude greater rate, which translated into an estimated 51,000 more fishes (i.e., net movement) entering Crane Creek or using the channel as a habitat. The channel allowed all of these fishes to move freely when seeking desirable habitat or fleeing undesirable habitat (Jude and Pappas 1992, Bouvier 2006). The ability to avoid undesirable or harsh habitat in Crane Creek by using the connection to Lake Erie may be especially important during the summer months as temperatures are at their highest and dissolved oxygen levels in the wetland routinely drop to hypoxic levels at night (Table 3.11).

The daily variation in and gradients between the physical and chemical conditions associated with the natural rhythms of productivity and hydrology near the channel appeared to drive the large daily movements of fishes in the channel (Figure 4.16, Figure 4.17, Figure 4.20). The longnose gar, for example, exited the marsh in great abundance between 9:00 pm and 11:00 pm each day (Figure 4.18). At first, it appeared that this movement was related to increased flow in the channel from the marsh to the lake as part of the seiche, but the timing of the seiche varies each day depending on wind speed, wind direction, and atmospheric pressure. The timing of the gar movement was much more consistent. Further analysis revealed a stronger correlation with the peak dissolved oxygen and water temperature in the marsh. During the daylight hours, water temperatures rose and photosynthetic processes in the marsh raised the dissolved oxygen concentrations to peak levels between approximately 4:00 pm and 8:00 pm. The temperature began to drop rapidly after this peak. Dissolved oxygen also began to drop, but its rate of decrease was not as great (Figure 4.6, Figure 4.9). During this study, the seiche out to Lake Erie roughly coincided with this peak causing the oxygen rich water to flow through the channel on its way to Lake Erie. We hypothesize that gar using the marsh habitats during the day sensed the abrupt change in temperature and dissolved oxygen levels and began their escape to Lake Erie, a behavior described by Jude and Pappas (1992). It appears that this response was common throughout the population since large numbers of gar evacuated the marsh shortly after conditions changed. Gar began to return to the marsh, often in small numbers, as soon as 2 hours after and in moderate density until dissolved oxygen levels peaked the following day (Figure 4.19). This diurnal pattern was repeated throughout the study and accounted for the asymmetry in the amount of time gar enter and exit the marsh.

The shoals of fish appeared to be more tolerant of variations in temperature and dissolved oxygen concentrations in Crane Creek, because they did not show a strong diurnal pattern of movement out of the creek when conditions became harsh (Figure 4.20). Their passage through the channel was spread out over time more than the gar, likely a function of their swimming speed (they swim slower than gar), location in the marsh (some stay close to the channel and others travel farther upstream), or inherent response to dropping dissolved oxygen levels. These shoals of emerald shiner, gizzard shad, spottail shiners, or other prey fish may be staying in the marsh for longer periods as a strategy to avoid predation as described by Coutant (1987). Overall, it appears that the shoals of fish are spending more time going both in and out in moderate density, possibly a strategy to gain benefit from alternating use of the marsh habitats and lake habitats on a shorter time scale and maintain the ability to respond quickly to changes in water chemistry, predator pressure, or other factors.

The carp, goldfish, channel catfish, and other species that likely comprised the bulk of the Other Fish category showed the highest tolerance for and least response to variations in dissolved oxygen levels, water flow velocity or direction, temperature, or other parameters. They appeared to be using the channel as habitat quite often, rather than as a corridor for movement into or out of the marsh. This created a complex situation where some fishes were milling about the channel throughout the day while longnose gar and shoals of fish move through in varying densities. In addition, fishes with smaller body size, often with different swimming styles, were present but unquantifiable in the DIDSON data. It is possible that some of these fish were following movement patterns similar to longnose gar and the shoals of forage fish, but our DIDSON data didn’t support this level of analysis.

4.5.3 Sampling Device and Challenges with Data Analysis

This study represented the first time that the DIDSON technology has been used to examine fish movement and behavior in Great Lakes coastal wetlands. The sonar provided a useful first high-resolution look at the fishes entering and exiting the Crane Creek wetland complex at all times of the day and under all water quality conditions. Optical video cameras have been used in similar ways (Frezza et al. 2003), but this technology is of little use when light levels decline at night or when turbidity is high. The acoustic energy of the DIDSON was able to penetrate the turbid (100+ NTU) water and collect data at all times of the day and at all turbidity levels.

The amount of data generated by the DIDSON sensor presented a challenge because the 6 frames/second collection rate translated into over 1 GB of data per hour. Quantifying and describing the fishes in the over 4 million frames of data collected during this study was difficult, especially since the fishes behavior often was complex (e.g., varying densities of fish of all different sizes were moving in all directions; see Figure 4.16, Figure 4.17, Figure 4.20). This presented a unique set of challenges not encountered with more standard sampling gears. Manual interpretation of the data (i.e., a person watching the data stream and recording observations) was both time intensive and subjective. The interpreter was required to view portions of the data over and over again until s/he was able to identify, quantify, and describe the observable fishes, often over 3,000 fishes per hour. Experience of the interpreter, fatigue, complexity of the scene, and many other factors made uniform interpretation of the data difficult over long periods of time.

The stratified random sampling approach we used sought to provide both a general description of the data through the use of density strata and a targeted intensive analysis through the data frame sampling. It was an effective way to analyze the data in the absence of a more automated process (Boswell et al. 2008). However, continued development of automated analysis methods is essential for DIDSON sampling of fish populations to reach its full potential. During our short study, it was clear that large numbers of fish were accessing the Crane Creek wetland complex, even though it was the warmest time of the summer in northwest Ohio, oxygen levels were quite low in the wetland at night, and a time when storm events were infrequent. We anticipate that this kind of data would be much more complex in the spring when adult fishes enter the marsh to spawn (see pages 46-49, Chapter 3) and in the fall when both shoals of juvenile fish leave the marsh prior to ice formation and large storms simultaneously drive more extreme seiches and watershed flooding. Increased abundance and flux of fishes through the channel during these times would make our method of analysis much more difficult but still useful if adequate time and resources are available.

4.5.4 Implications for Data Collection and Habitat Rehabilitation

As a result of this and previous studies, we now know that many large fish, including longnose gar, and shoals of many small (potentially forage) fish use the channel connecting Crane Creek wetlands with Lake Erie as both a pathway between systems and as temporary habitat on a daily basis. The high abundance of fish we observed supports our understanding that coastal wetland habitats, even in a degraded condition, are important resources for many species of Great Lakes fishes (Jude and Pappas 1992, Brazner and Beals 1997). Those fishes appear to be using the wetland habitats throughout the day and use the pathway to Lake Erie as an escape route when temperature and dissolved oxygen concentrations in the marsh begin their diurnal decline. Since this study just covered a short period in the summer, we can only speculate about whether similar patterns occur with other fish species, at other times of the year, or in similar coastal wetland systems. However, the large abundance of Lake Erie fishes captured with fyke nets in Crane Creek at other times of the year, which were even greater than the abundance of fish captured in the summer, suggest that the observations in this study are likely to be an underestimate of wetland utilization by fishes in the spring and fall (see page 46-49, Chapter 3).

Regular movements to and from the wetland may have been primarily a response to periodically harsh water quality conditions in the shallow marsh water, so improvements in water quality could alter how fishes use the marsh habitats. If the dissolved oxygen levels didn't plummet each evening, it is possible that more species of forage fish, including those with lower tolerance for poor water quality, would be able to occupy marsh habitats throughout the day rather than having to seek better conditions in Lake Erie. Longnose gar and other piscivorous fish likely would respond to the greater abundance of forage fish and improved water quality conditions by remaining in the marsh long after temperature, pH, and dissolved oxygen peak, thus improving the habitats available to them.

4.5.5 Future Work Needed

Although this study provided a first look at fish movement between Lake Erie and a coastal riparian wetland, there are many areas that need further exploration. First and foremost, there is a need for software and methods to identify and quantify fishes and allow massively complex DIDSON data sets to be analyzed in a semi-automated way. Additional methods also could be developed to help identify the species of fish categorized as Other Fish in this study and examine their abundance and movement in response to variable water quality conditions. Continuous DIDSON data need to be collected at Crane Creek in the spring to target spawning adult fish and in the fall to target juvenile fish. These data could be compared to the fish abundance and movement patterns observed in this study need to evaluate seasonal shifts in fish behavior. Similarly, these types of observations could be conducted at other Lake Erie tributaries to explore other spatial and temporal patterns of fish usage, or the DIDSON technology could be used in multiple locations within a wetland to evaluate how far gar and shoals of fish travel into the wetland. Finally, this technology is well suited to examine how fishes respond to restored access to diked wetlands, impoundments, or rehabilitated wetlands. Hydrologic connectivity plays an important role in the composition and structure of fish assemblages of coastal wetlands (Bouvier 2006), but the short- and long-term response (e.g., abundance, flux, species richness, biomass) of Great Lakes fishes to newly rehabilitated habitat remains to be assessed.

4.6 Acknowledgments

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Table 4.1. Descriptive statistics of water quality at three sites: Crane Creek Lower Wetland (CCLW), Crane Creek Channel (CCCH), and Erie Nearshore (ENS). Specific conductivity data for ENS site not available.

Site		**CCLW**	**CCCH**	**ENS**
Temp (°C)	Min	23.00	22.38	22.62
	Max	30.57	33.27	28.93
	Mean ± SE	26.90 ± 0.07	26.19 ± 0.07	25.71 ± 0.05
Dissolved Oxygen (mg/l)	Min	2.25	2.93	6.76
	Max	13.34	13.89	11.50
	Mean ± SE	8.44 ± 0.11	9.31 ± 0.07	8.81 ± 0.08
pH	Min	7.54	7.59	7.71
	Max	9.06	9.01	8.98
	Mean ± SE	8.41 ± 0.02	8.63 ± 0.01	8.70 ± 0.01
Specific Conductivity (uS)	Min	0.26	0.26	-
	Max	0.40	0.36	-
	Mean ± SE	0.30 ± 0.00	0.29 ± 0.00	-
Turbidity (NTU)	Min	3.70	6.25	9.20
	Max	70.45	234.40	91.15
	Mean ± SE	22.49 ± 0.57	32.84 ± 0.97	23.25 ± 0.69

Table 4.2. Mean number of fish and shoals of small fish observed in the DIDSON field (± SE) in 202 random 40-second subsamples by sampling stratum. As detailed in the text, strata were based on relative density of gar in the acoustic image data.

Sampling		Gar			Shoals			Other Fish		
Stratum	**N (%) Samples**	**In**	**Out**	**Total**	**In**	**Out**	**Total**	**In**	**Out**	**Total**
High	106 (52%)	2.46 (0.27)	3.41 (0.42)	5.87 (0.41)	0.22 (0.09)	0.80 (0.19)	1.02 (0.21)	16.25 (1.28)	22.31 (1.72)	38.57 (2.08)
Average	77 (38%)	2.42 (0.29)	0.92 (0.17)	3.40 (0.33)	0.26 (0.09)	0.56 (0.16)	0.82 (0.20)	13.38 (0.92)	6.87 (0.60)	20.25 (1.22)
Low	18 (9%)	0.83 (0.29)	0.22 (0.22)	1.06 (0.36)	0.39 (0.20)	1.00 (0.28)	1.39 (0.27)	10.50 (1.62)	7.06 (1.55)	17.56 (2.44)
Absent	1 (1%)	0.00	0.00	0.00	2.00	2.00	4.00	9.00	2.00	11.00

Table 4.3. Estimated mean density (number fish / m^2), standard deviation, and 90% confidence intervals of fish observed in the connecting channel during the DIDSON sampling period.

Fish sonar signatures	Mean	Standard Deviation	90% Confidence Interval
All large individuals	0.81	0.37	$0.79 < u < 0.83$
Gar	0.08	0.07	$0.08 < u <0.09$
Other	0.73	0.31	$0.71 < u < 0.74$
Discrete shoals	0.05	0.04	$0.05 < u < 0.06$
Est. # shoaling fishes	9.22	-	-
Total	10.01	-	-

Table 4.4. Fish species captured by fyke nets and gill nets set in the Crane Creek channel. Commonly shoaling prey fish species are indicated by "*".

Common Name	Species
Black crappie	*Pomoxis nigromaculatus* (Lesueur)
Bluegill	*Lepomis macrochirus* (Rafinesque)
Brook silverside	*Labidesthes sicculus* (Cope)
Brown bullhead	*Ameiurus nebulosus* (Lesueur)
Carp	*Cyprinus carpio* (Linnaeus)
Channel catfish	*Ictalurus punctatus* (Rafinesque)
Emerald shiner*	*Notropis atherinoides* (Rafinesque)
Freshwater drum	*Aplodinotus grunniens* (Rafinesque)
Gizzard shad*	*Dorosoma cepedianum* (Lesueur)
Goldfish	*Carassius auratus* (Linnaeus)
Largemouth bass	*Micropterus salmoides* (Lacepède)
Logperch darter	*Percina caprodes* (DeKay)
Longnose gar	*Lepisosteus osseus* (Linnaeus)
Rock bass	*Ambloplites rupestris* (Rafinesque)
Spottail shiner*	*Notropis hudsonius* (Clinton)
White bass	*Morone chrysops* (Rafinesque)
White crappie	*Pomoxis annularis* (Rafinesque)
White perch	*Morone americana* (Gmelin)
Yellow bullhead	*Ameiurus natalis* (Lesueur)
Yellow perch	*Perca flavescens* (Mitchill)

Table 4.5. Estimated number of fish moving into Crane Creek or out to Lake Erie during the study period. Positive flux values indicate fish movement from Crane Creek into Lake Erie.

	Mean #/min ± SE	Total During Study Period
Gar – In	2.15 ± 0.05	24,649
Gar – Out	1.50 ± 0.07	17,223
Net Gar Flux	0.65 ± 0.05	7,426
Other Fish – In	18.23 ± 0.13	208,695
Other Fish – Out	13.73 ± 0.36	157,184
Net Other Fish Flux	4.50 ± 0.05	51,510
Shoals – In	0.90 ± 0.03	10,335
Shoals – Out	1.38 ± 0.03	15,812
Net Shoals Flux	-0.48 ± 0.02	-5,477

Table 4.6. Number and percentage of 0.25 hour time units (out of 800 observations) that longnose gar were observed in each direction category (In, Out) and density stratum (High, Average, Low, Absent) as derived from initial analysis of the DIDSON data.

	Total	In	Out	Bidirectional
High	160 (20%)	84 (11%)	72 (9%)	4 (1%)
Average	219 (27%)	154 (19%)	34 (4%)	31 (4%)
Low	293 (37%)	199 (25%)	32 (4%)	62 (8%)
Absent	128 (16%)	-	-	-
Total	-	437 (55%)	138 (17%)	97 (12%)

Table 4.7. Number and percentage of 0.25 hour time units (out of 800 observations) that Other Fish were observed in each direction category (In, Out) and density stratum (High, Average, Low, Absent) as derived from initial analysis of the DIDSON data.

	Total	In	Out	Bidirectional
High	63 (8%)	24 (3%)	12 (2%)	196 (25%)
Average	452 (57%)	112 (14%)	110 (14%)	230 (29%)
Low	200 (25%)	57 (7%)	43 (5%)	100 (13%)
Absent	85 (11%)	-	-	-
Total	-	193 (24%)	165 (21%)	526 (66%)

Table 4.8. Percentage of 0.25 hour time units (out of 800 observations) that longnose gar were observed moving in each direction category (In, Out, Bidirectional), presented by density stratum (High, Average, Low) and direction of water flow in the channel.

Fish Movement	**In**			**Out**			**Bidirectional**		
Water Flow	**In**	**Out**	**Total**	**In**	**Out**	**Total**	**In**	**Out**	**Total**
High	25%	15%	19%	65%	35%	52%	5%	3%	4%
Average	29%	39%	35%	17%	35%	25%	29%	36%	32%
Low	46%	45%	46%	18%	30%	23%	66%	64%	64%

Table 4.9. Percentage of 0.25 hour time units (out of 800 observations) in each direction category (In, Out), presented by density stratum (High, Average, Low), that Other Fish were observed during initial analysis of the DIDSON data.

Fish Movement	**In**			**Out**			**Bidirectional**		
Water Flow	**In**	**Out**	**Total**	**In**	**Out**	**Total**	**In**	**Out**	**Total**
High	8%	16%	12%	8%	6%	7%	32%	42%	37%
Average	65%	52%	58%	71%	61%	67%	43%	44%	44%
Low	27%	32%	30%	21%	33%	26%	24%	14%	19%

Table 4.10. Number and percentage of 0.25 hour time units (out of 800 observations) that shoals of fish were observed in each direction category (In, Out) and density stratum (High, Average, Low, Absent) as derived from initial analysis of the DIDSON data.

	Total	In	Out	Bidirectional
High	76 (10%)	25 (3%)	34 (4%)	17 (2%)
Average	271 (34%)	71 (9%)	91 (11%)	109 (14%)
Low	285 (36%)	79 (10%)	93 (12%)	113 (14%)
Absent	168 (21%)	-	-	-
Total	-	175 (22%)	218 (27%)	239 (30%)

Table 4.11. Percentage of 0.25 hour time periods (out of 800 observations) in each direction category (In, Out), presented by density stratum (High, Average, Low), that shoals of fish were observed during initial analysis of the DIDSON data.

Fish Movement	**In**			**Out**			**Bidirectional**		
Water Flow	**In**	**Out**	**Total**	**In**	**Out**	**Total**	**In**	**Out**	**Total**
High	16%	11%	14%	18%	15%	16%	9%	6%	7%
Average	42%	36%	41%	48%	39%	42%	46%	45%	46%
Low	42%	53%	45%	34%	46%	43%	45%	49%	47%

Figure 4.1. Location of the Crane Creek wetland complex in western Lake Erie. Approximate location of DIDSON sampling is noted. Schematic cross section of DIDSON deployment. Water-quality sampling sites indicated with yellow star.

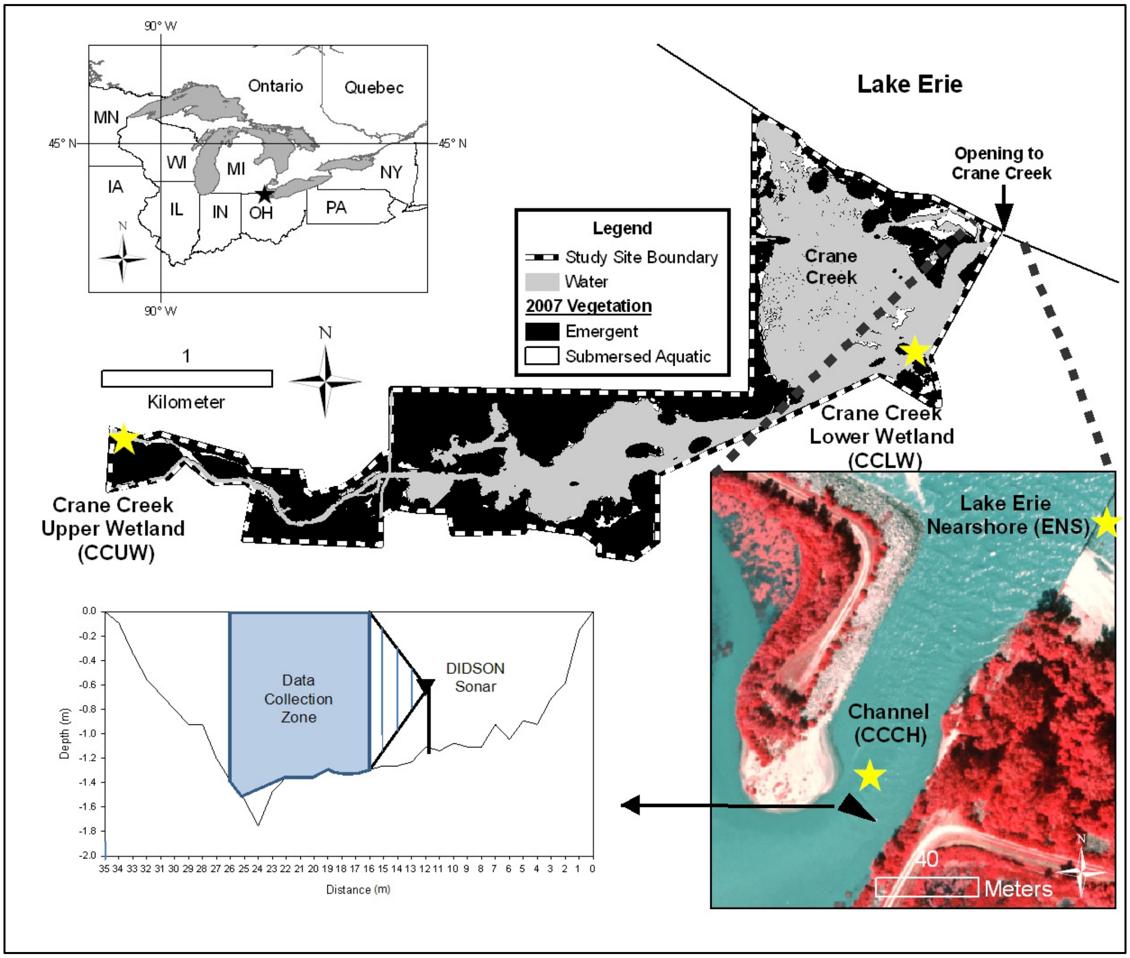


Figure 4.2. Water-surface elevation gradient between the Lake Erie at the Toledo Harbor and site CCUW in Crane Creek at the intersection with Route 2 road (i.e., Lake Erie data – Crane Creek data). Positive values indicate greater Lake Erie water-surface elevation, and negative values indicate greater Crane Creek water-surface elevation. Positive velocity values indicate water flowing from Crane Creek through the channel toward Lake Erie.

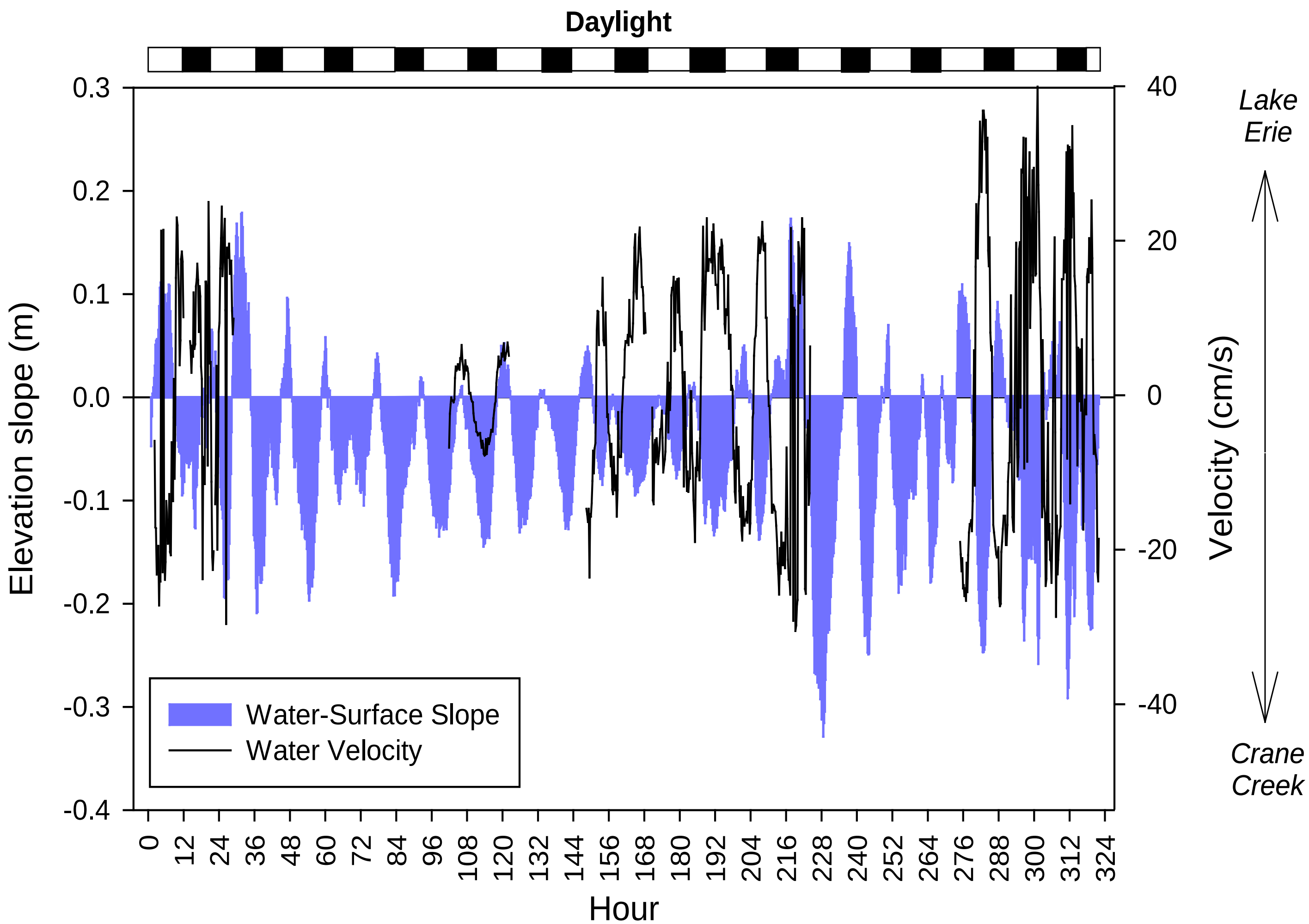


Figure 4.3. Water depth and velocity in Crane Creek connecting channel.

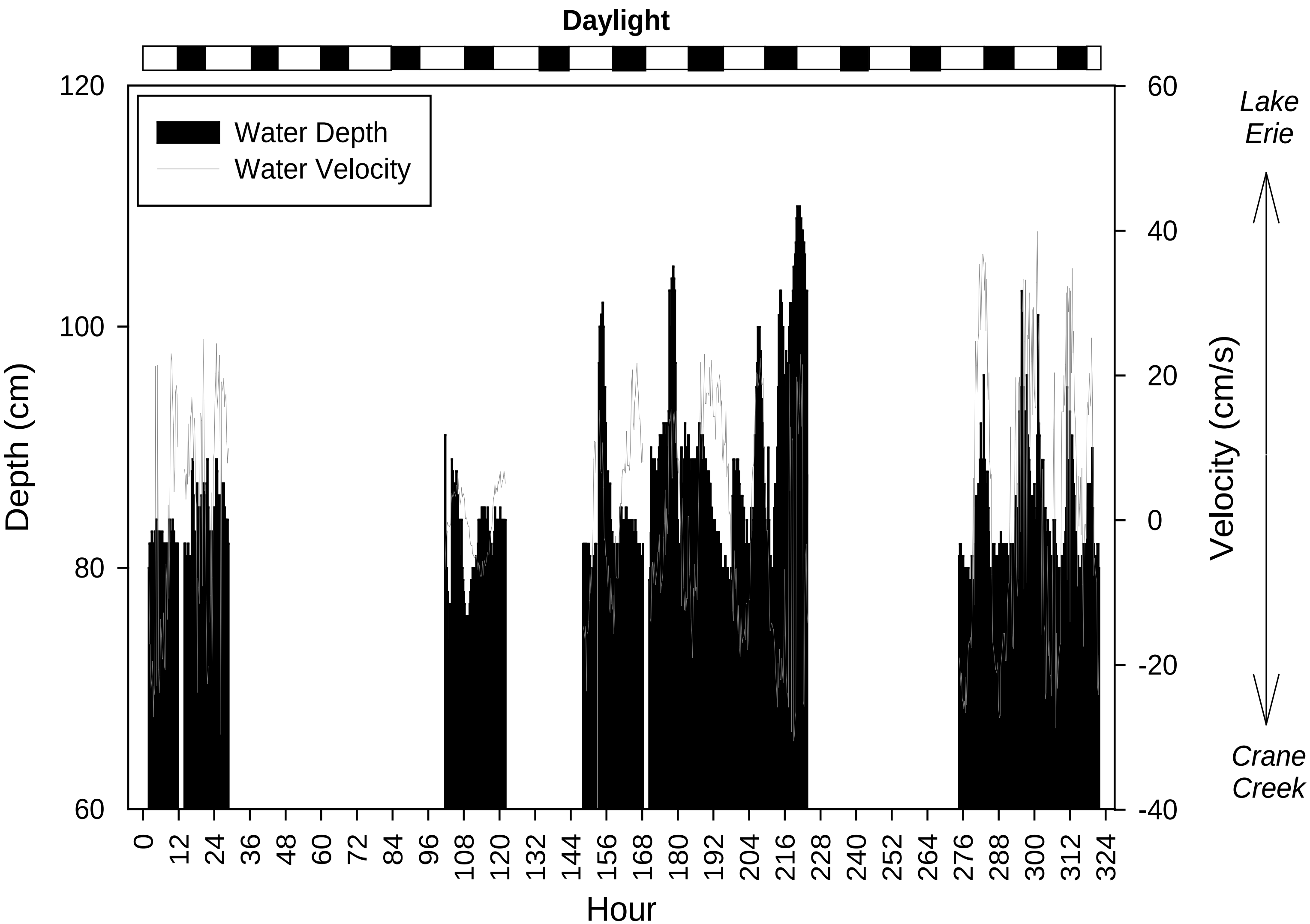


Figure 4.4. Air pressure recorded in hPa and displayed in hPa * 0.01.

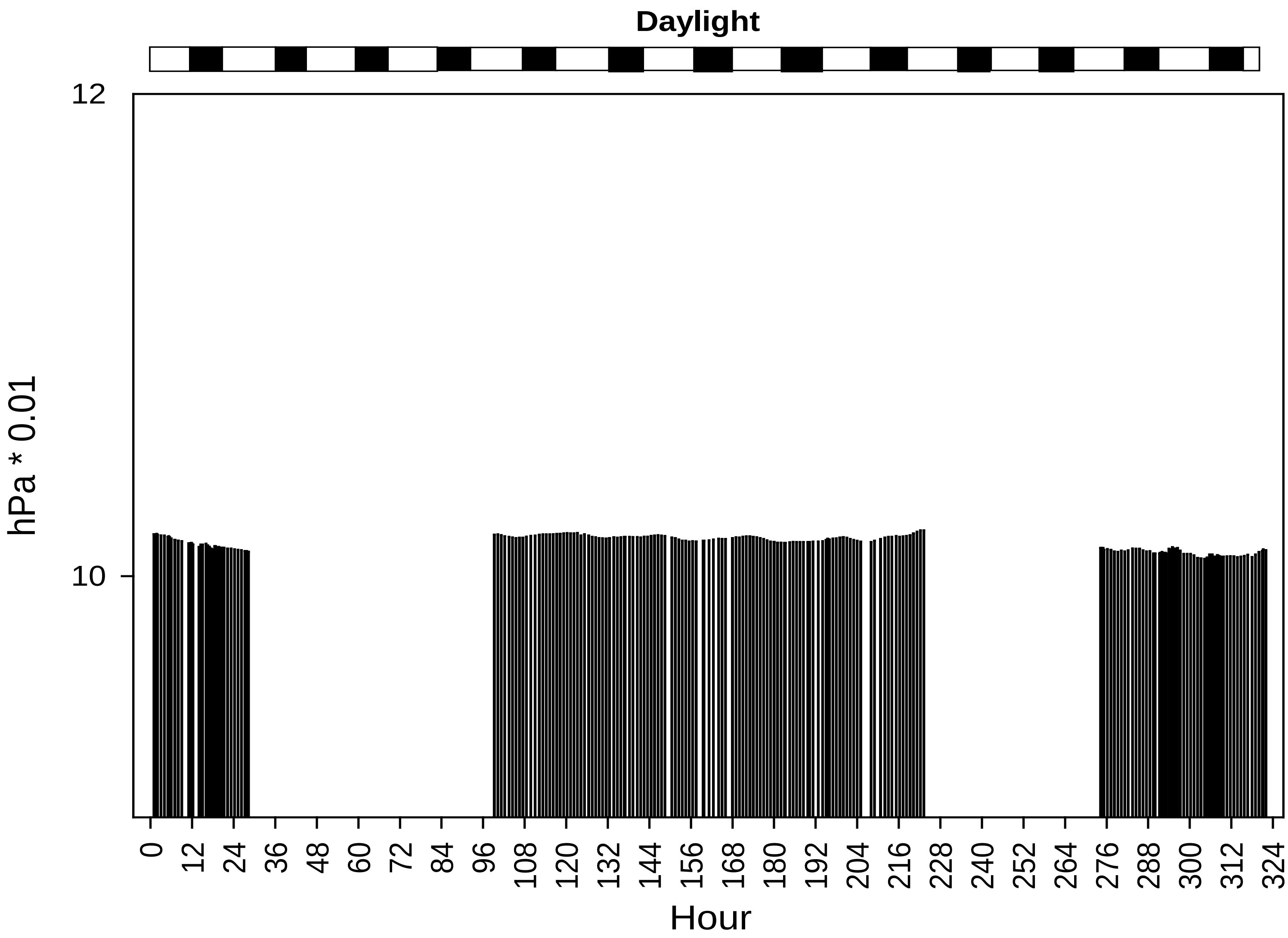


Figure 4.5. Total rainfall at the study site during sampling.

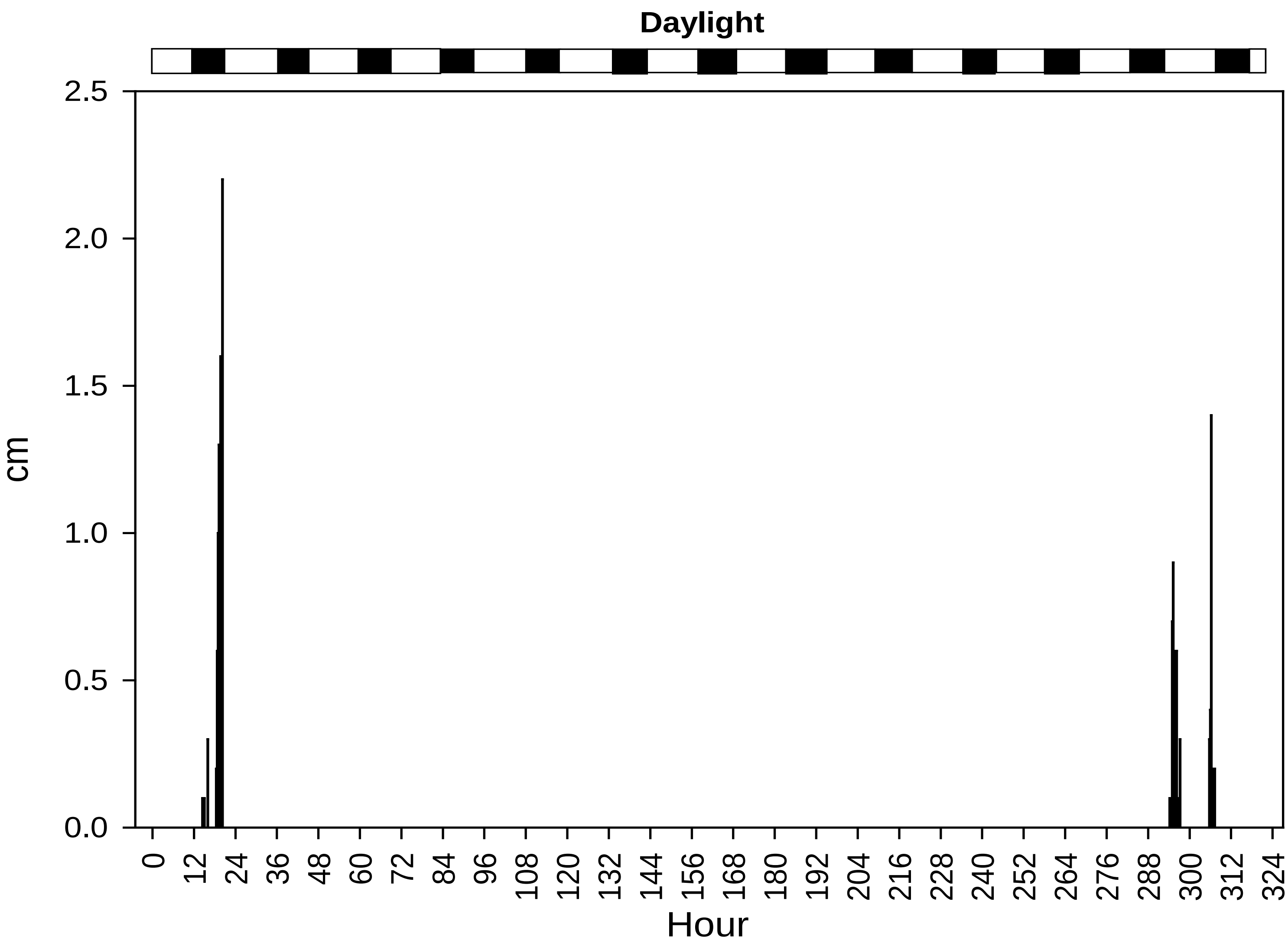


Figure 4.6. Water temperature values from YSI sondes located inside the wetland at site CCLW, at site CCCH adjacent to the DIDSON sensor in the channel between the wetland and Lake Erie, and in Lake Erie near the mouth of Crane Creek (ENS).

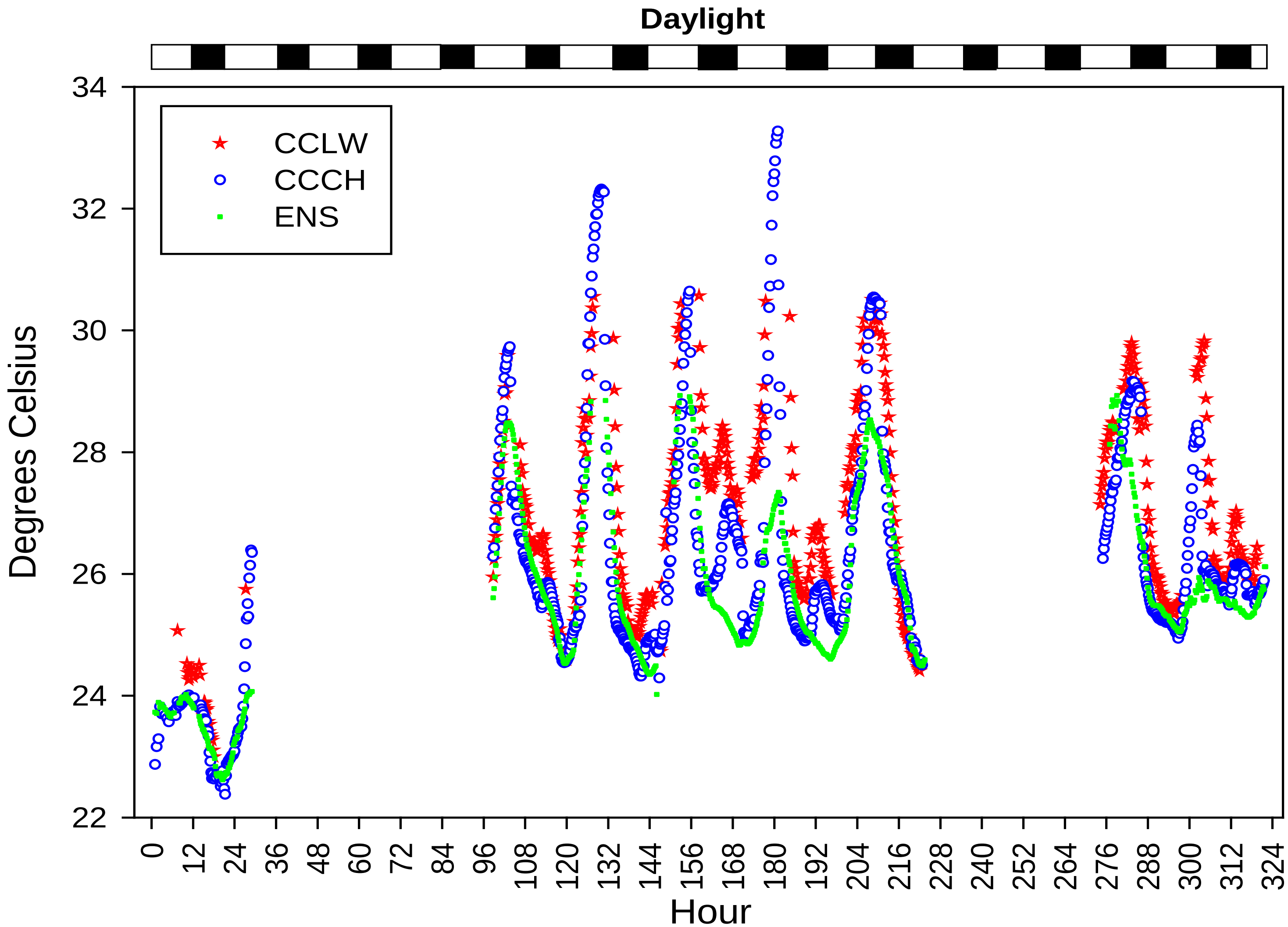


Figure 4.7. Air temperature recorded near the study site.

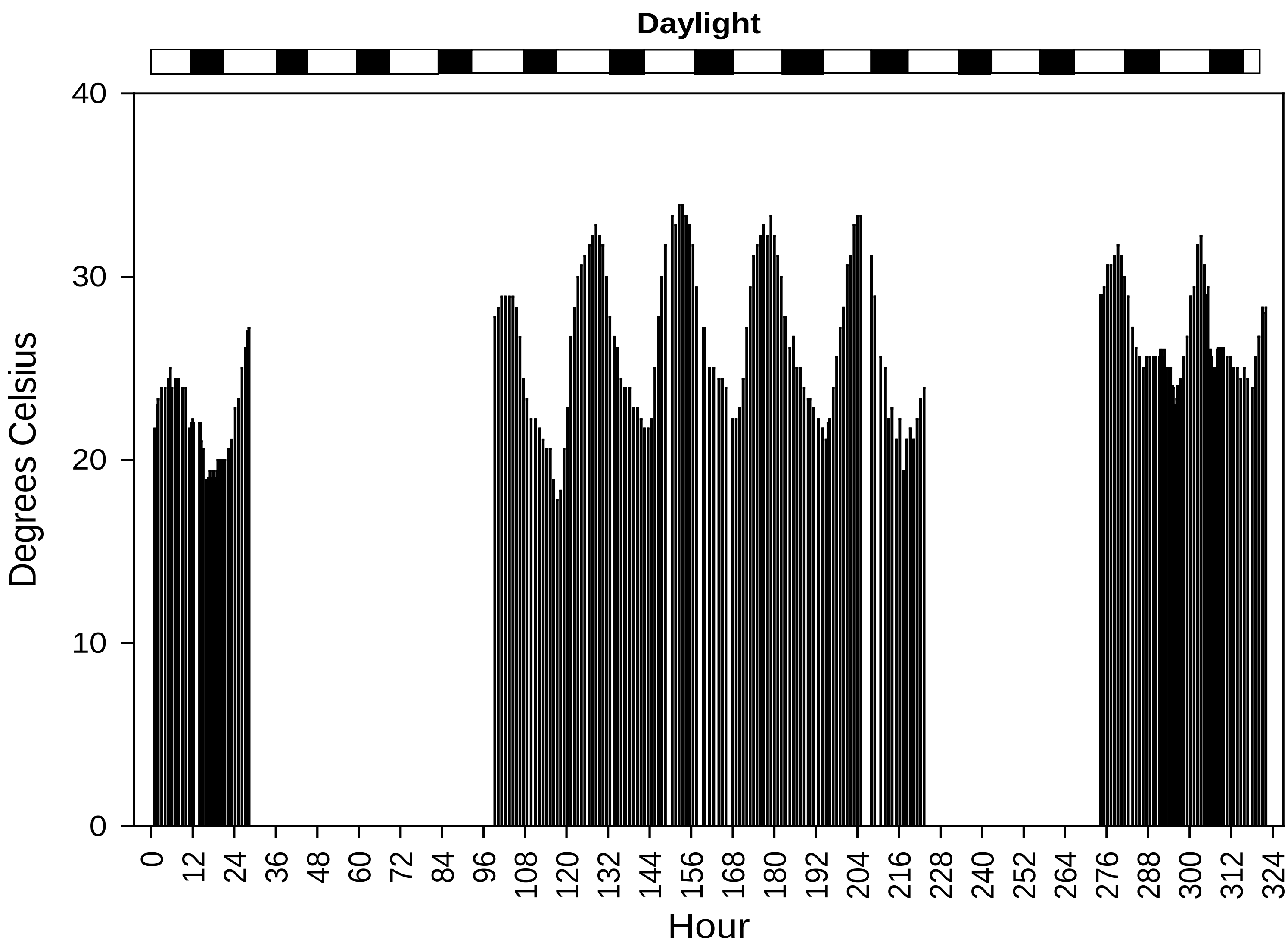


Figure 4.8. Temperature gradient between sites ENS and CCLW (i.e., Lake Erie nearshore data – lower Crane Creek wetland data). Positive values indicate greater Lake Erie temperatures, and negative values indicate greater Crane Creek temperatures.

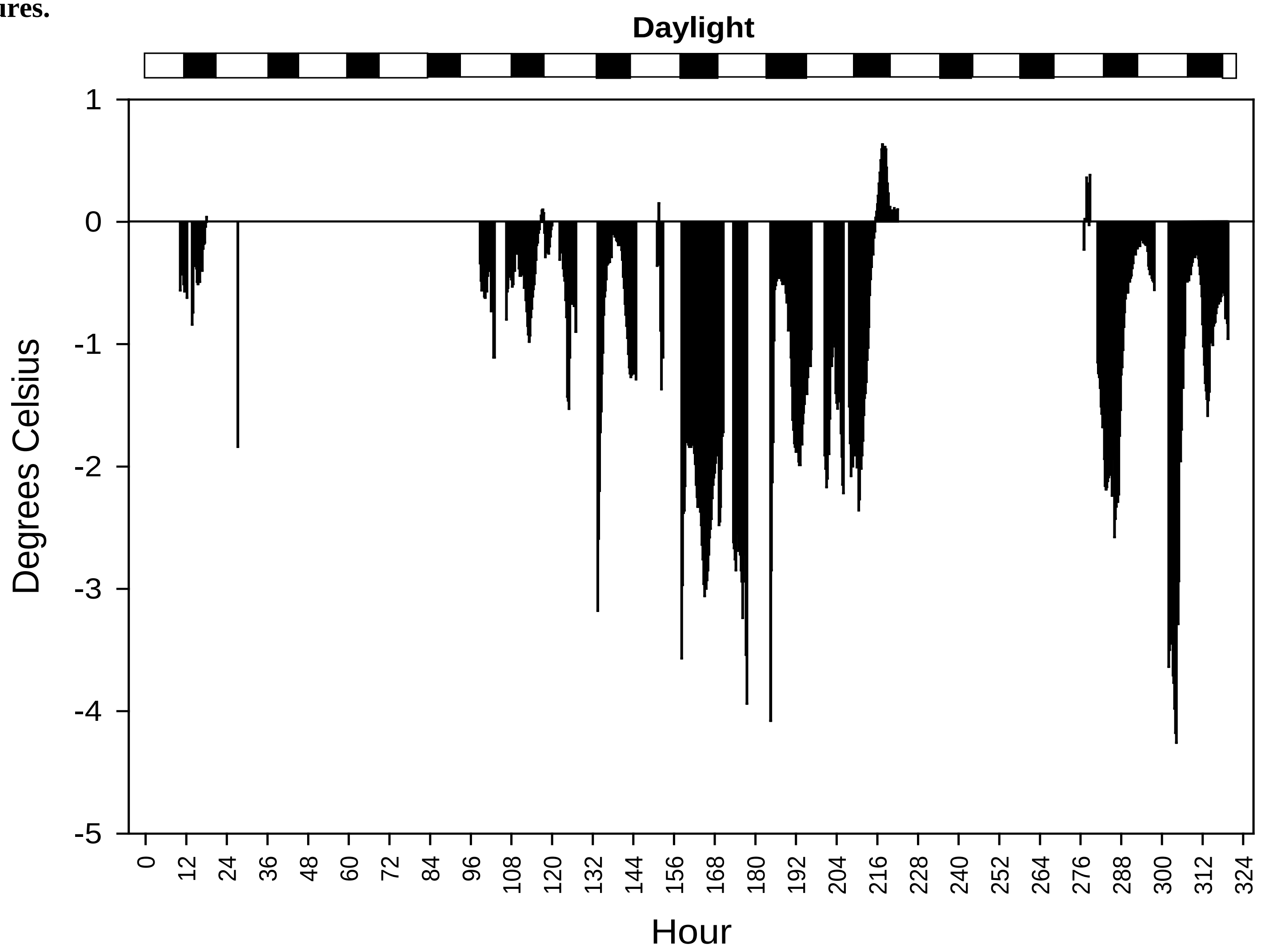


Figure 4.9. Dissolved oxygen values from YSI sondes located inside the wetland at site CCLW, at site CCCH adjacent to the DIDSON sensor in the channel between the wetland and Lake Erie, and in Lake Erie near the mouth of Crane Creek (ENS).

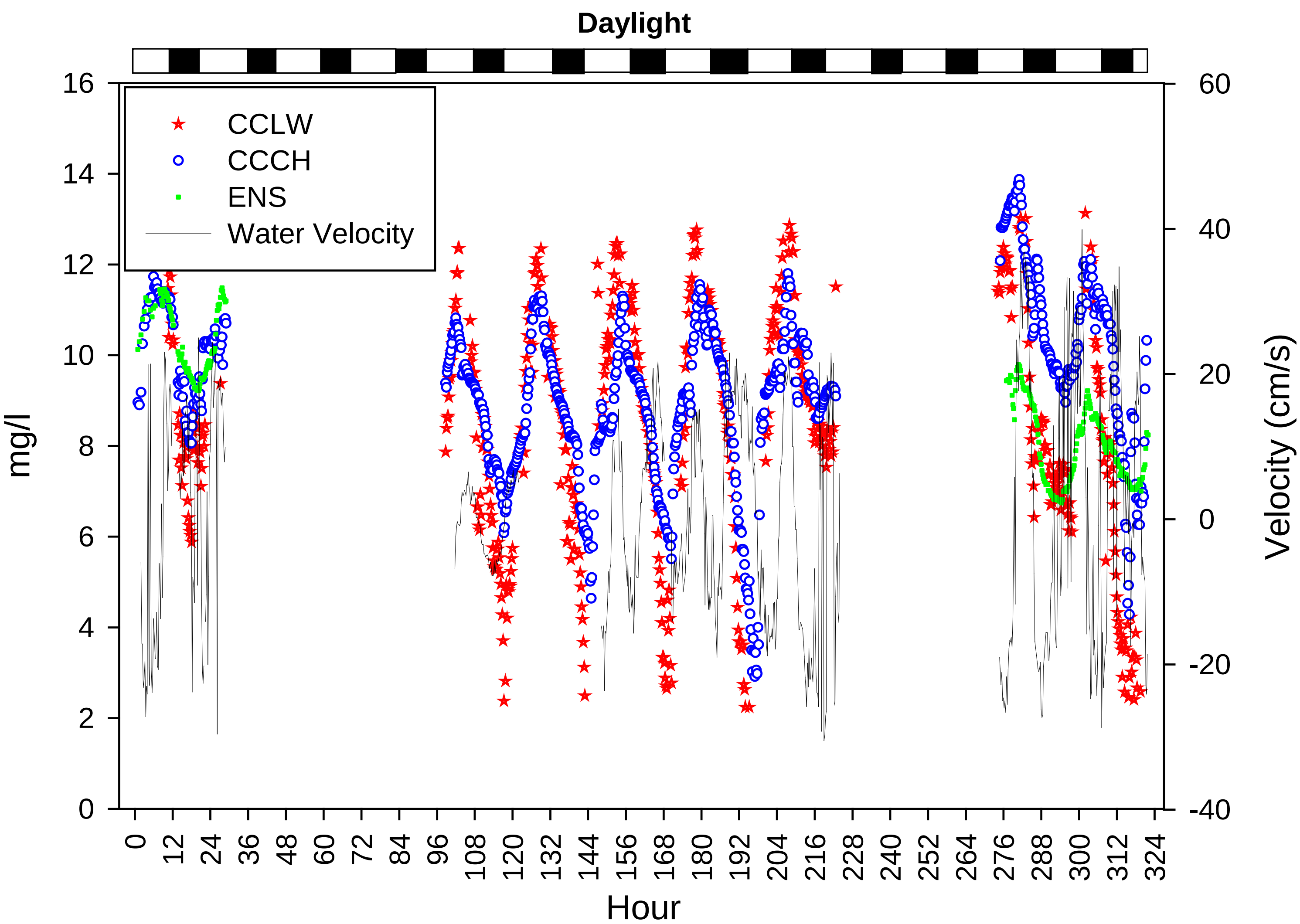


Figure 4.10. Dissolved oxygen gradient between sites ENS and CCLW (i.e., Lake Erie nearshore data – lower Crane Creek wetland data). Positive values indicate greater Lake Erie concentrations, and negative values indicate greater Crane Creek concentrations.

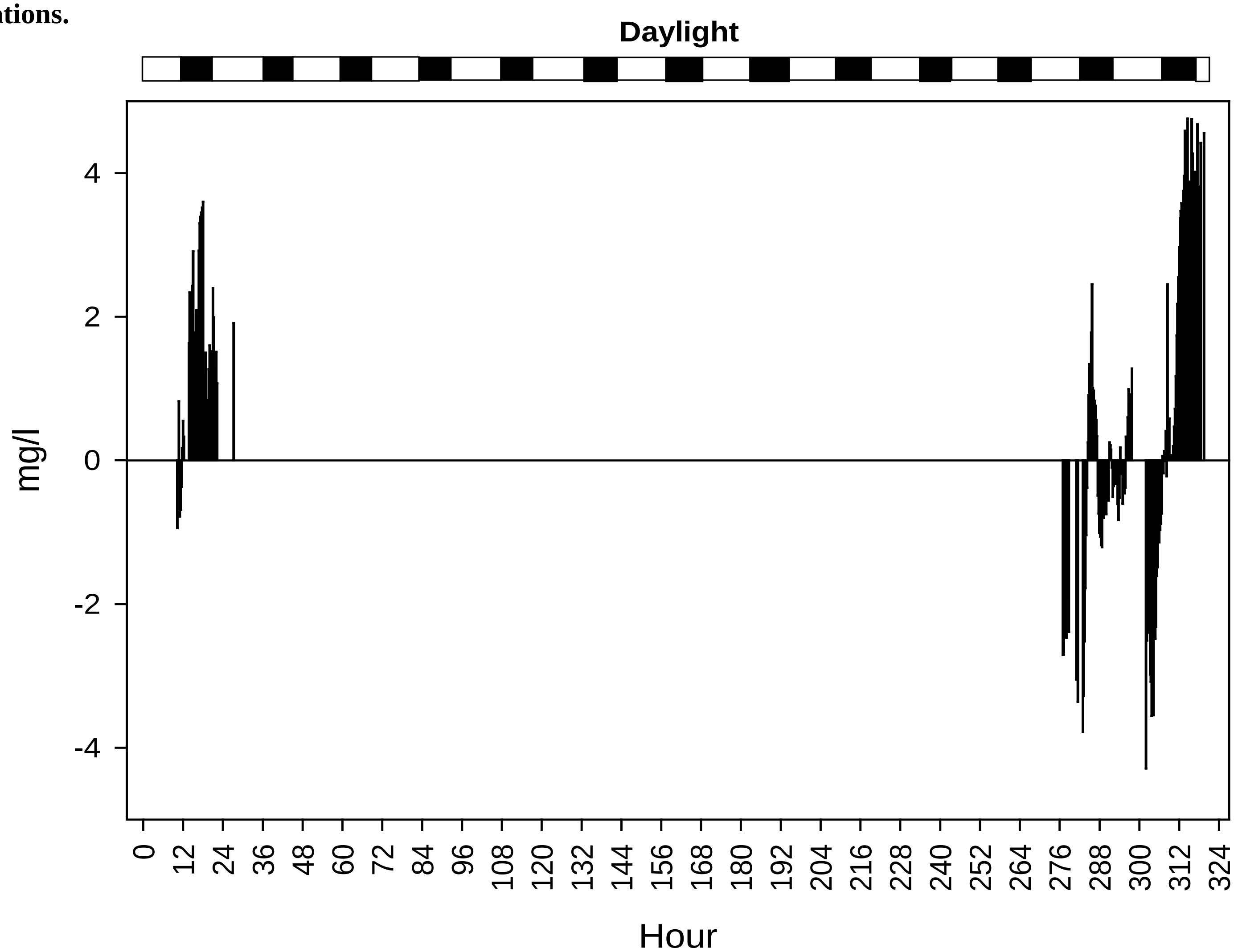


Figure 4.11. pH values from YSI sondes located inside the wetland at site CCLW, at site CCCH adjacent to the DIDSON sensor in the channel between the wetland and Lake Erie, and in Lake Erie near the mouth of Crane Creek (ENS).

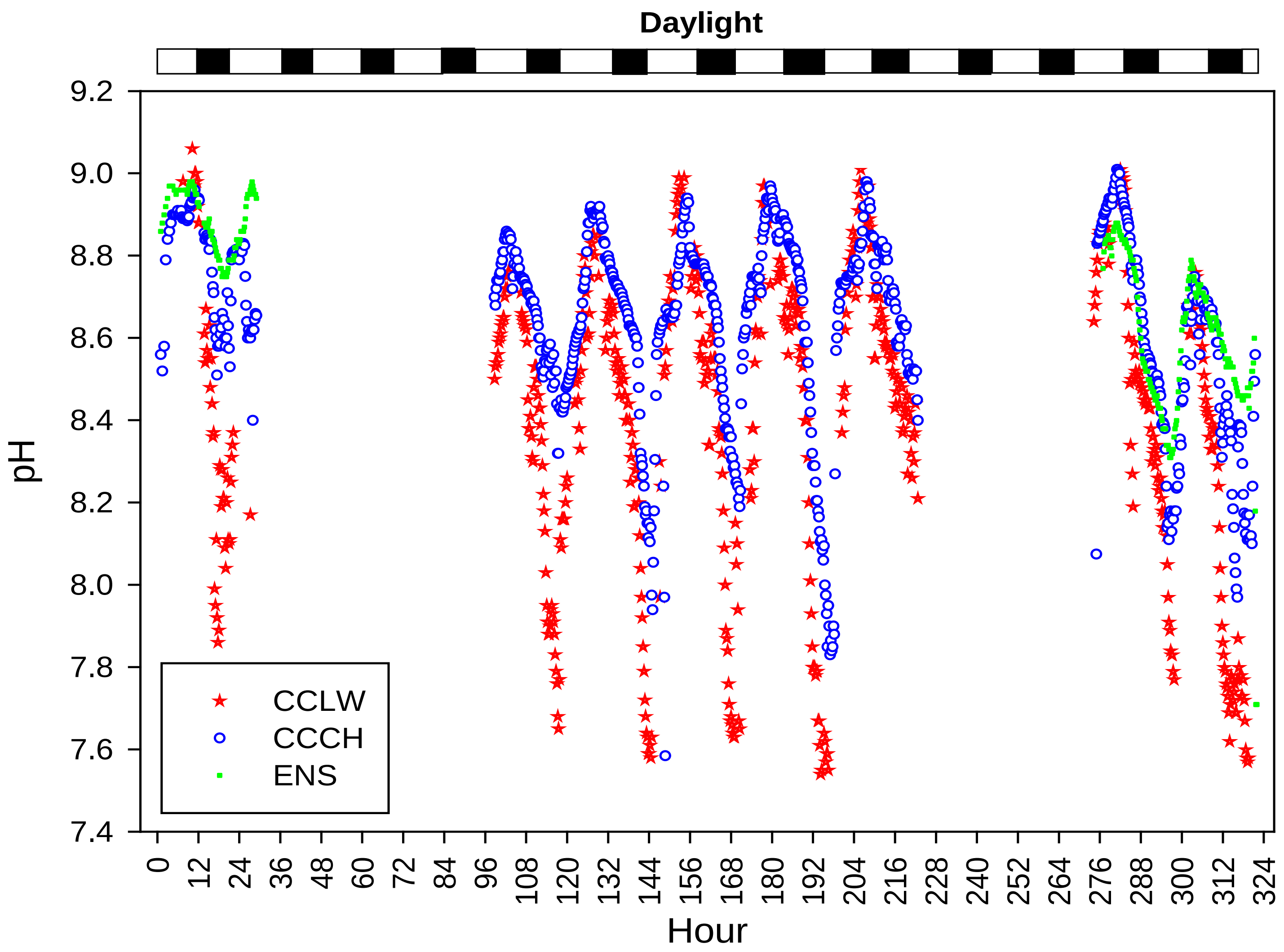


Figure 4.12. pH gradient between sites ENS and CCLW (i.e., Lake Erie nearshore data – lower Crane Creek wetland data). Positive data indicate greater Lake Erie values, and negative data indicate greater Crane Creek values.

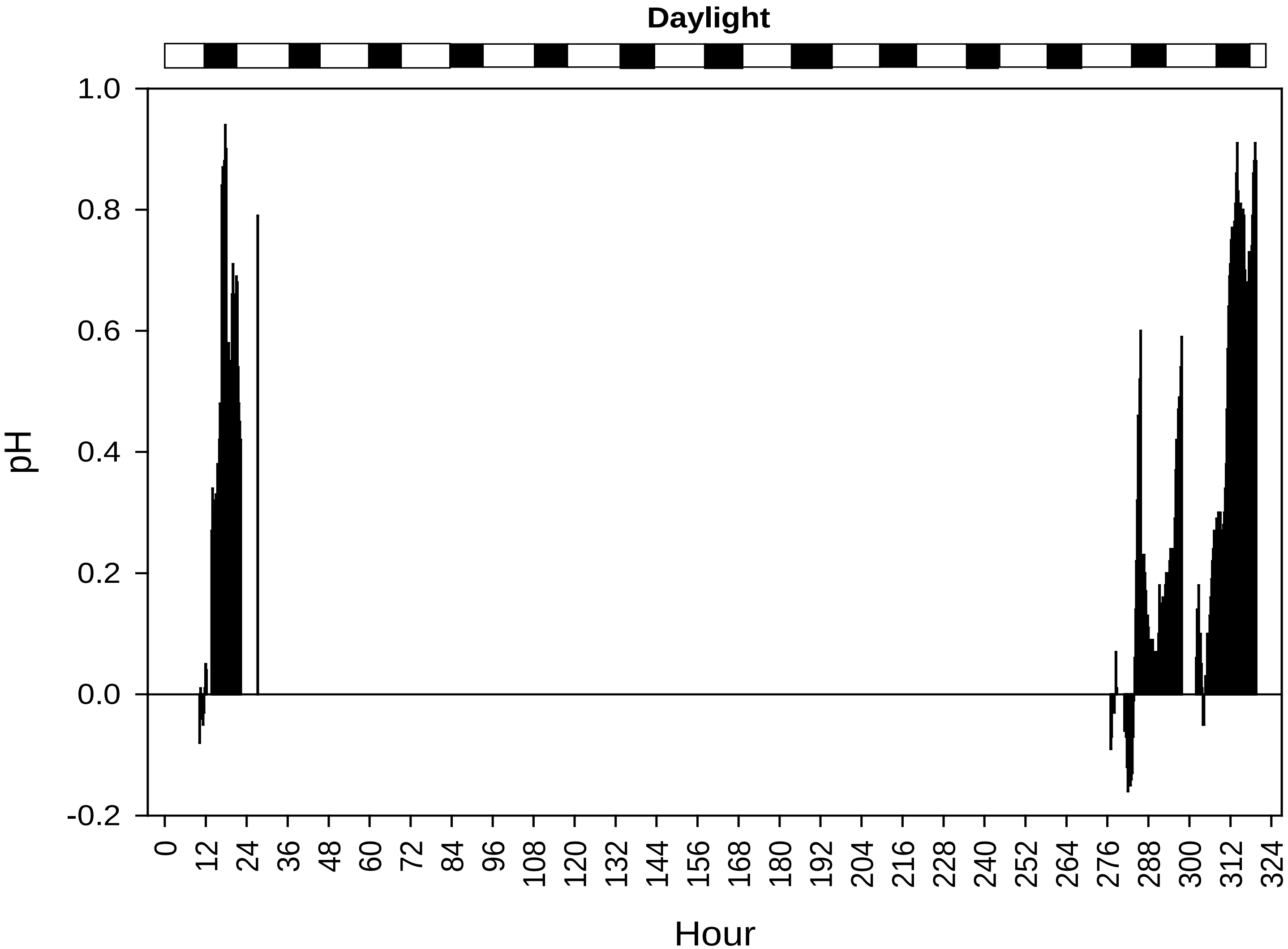


Figure 4.13. Specific conductivity values from YSI sondes located inside the wetland at CCLW and at site CCCH adjacent to the DIDSON sensor in the channel between the wetland and Lake Erie.

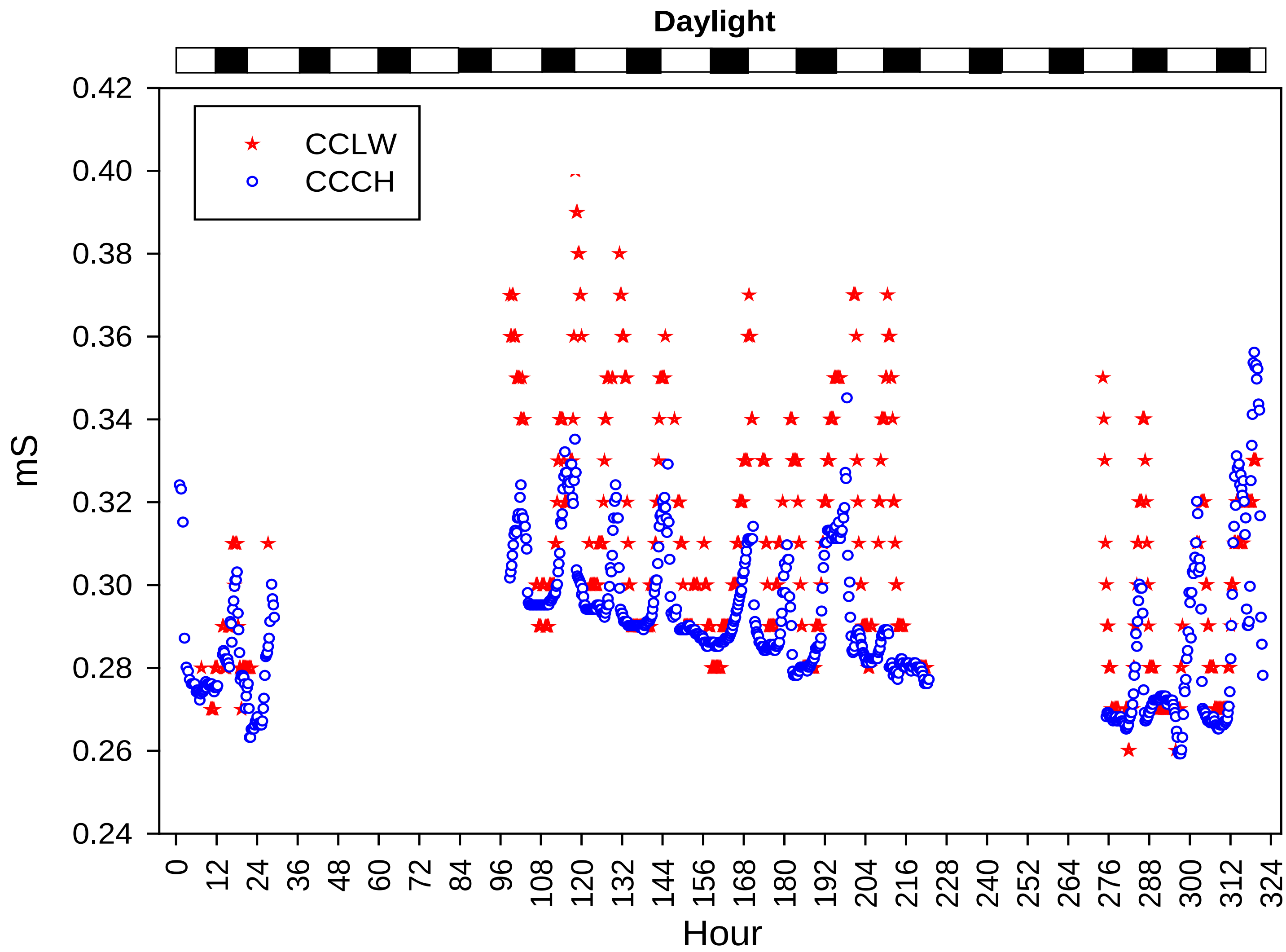


Figure 4.14. Turbidity values from YSI sondes located inside the wetland at CCLW, at site CCCH adjacent to the DIDSON sensor in the channel between the wetland and Lake Erie, and in Lake Erie near the mouth of Crane Creek (ENS).

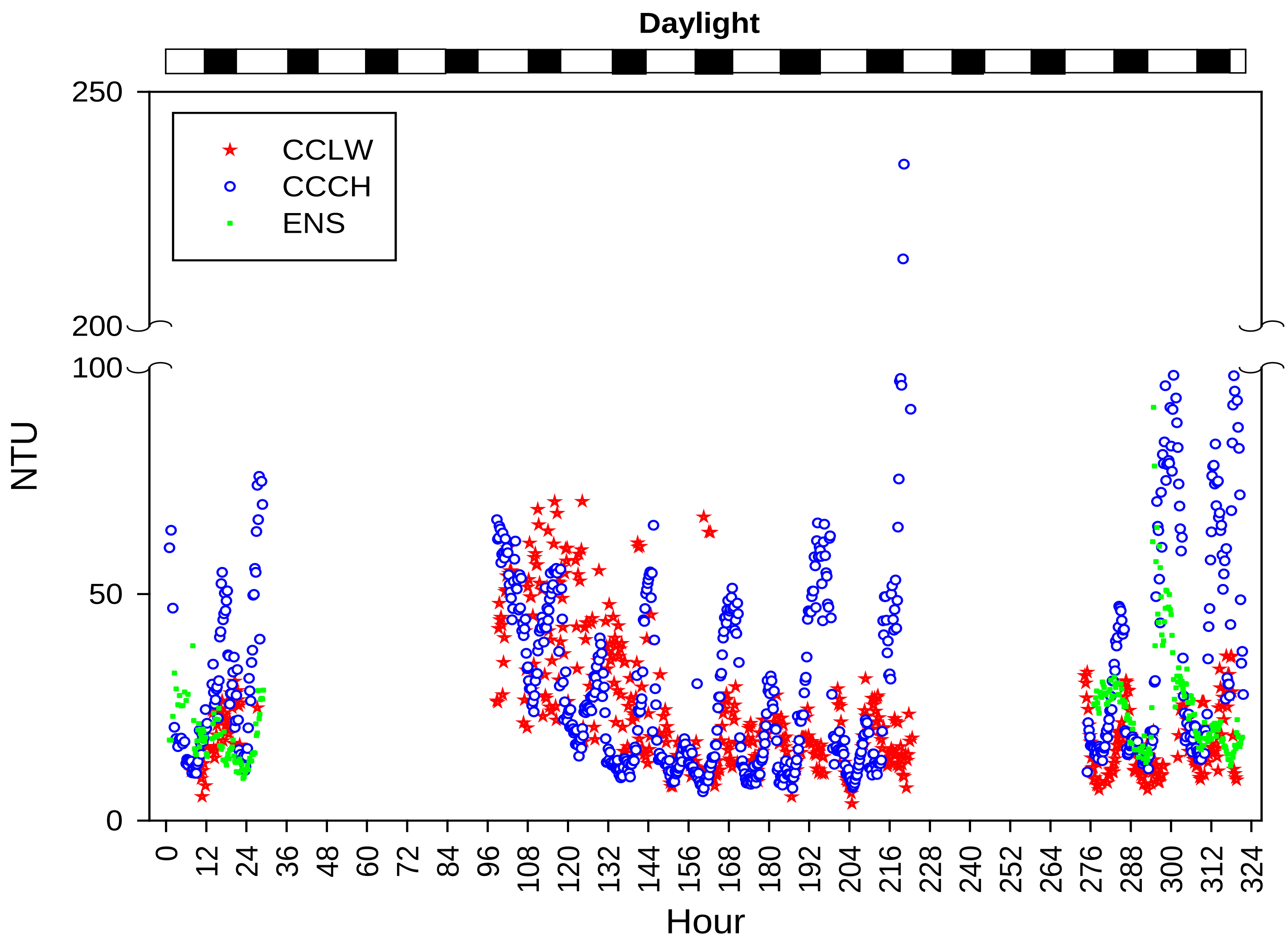


Figure 4.15. Turbidity gradient between sites ENS and CCLW (i.e., Lake Erie nearshore data – lower Crane Creek wetland data). Positive data indicate greater Lake Erie values, and negative data indicate greater Crane Creek values.

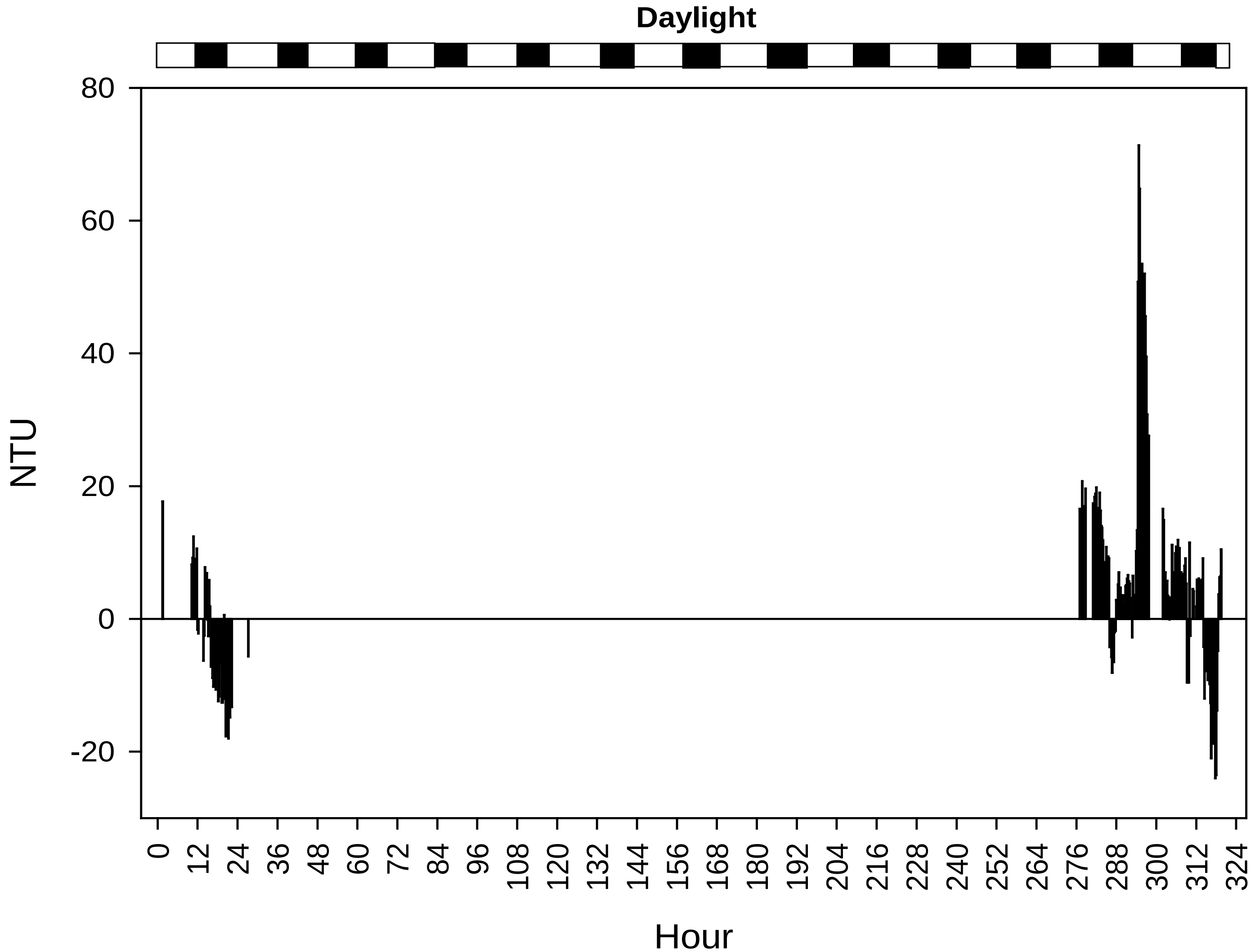


Figure 4.16. Categorical density data for longnose gar imaged with the DIDSON technology. Velocity data for water moving through the channel to Lake Erie. Positive values indicate water flowing from Crane Creek toward Lake Erie.

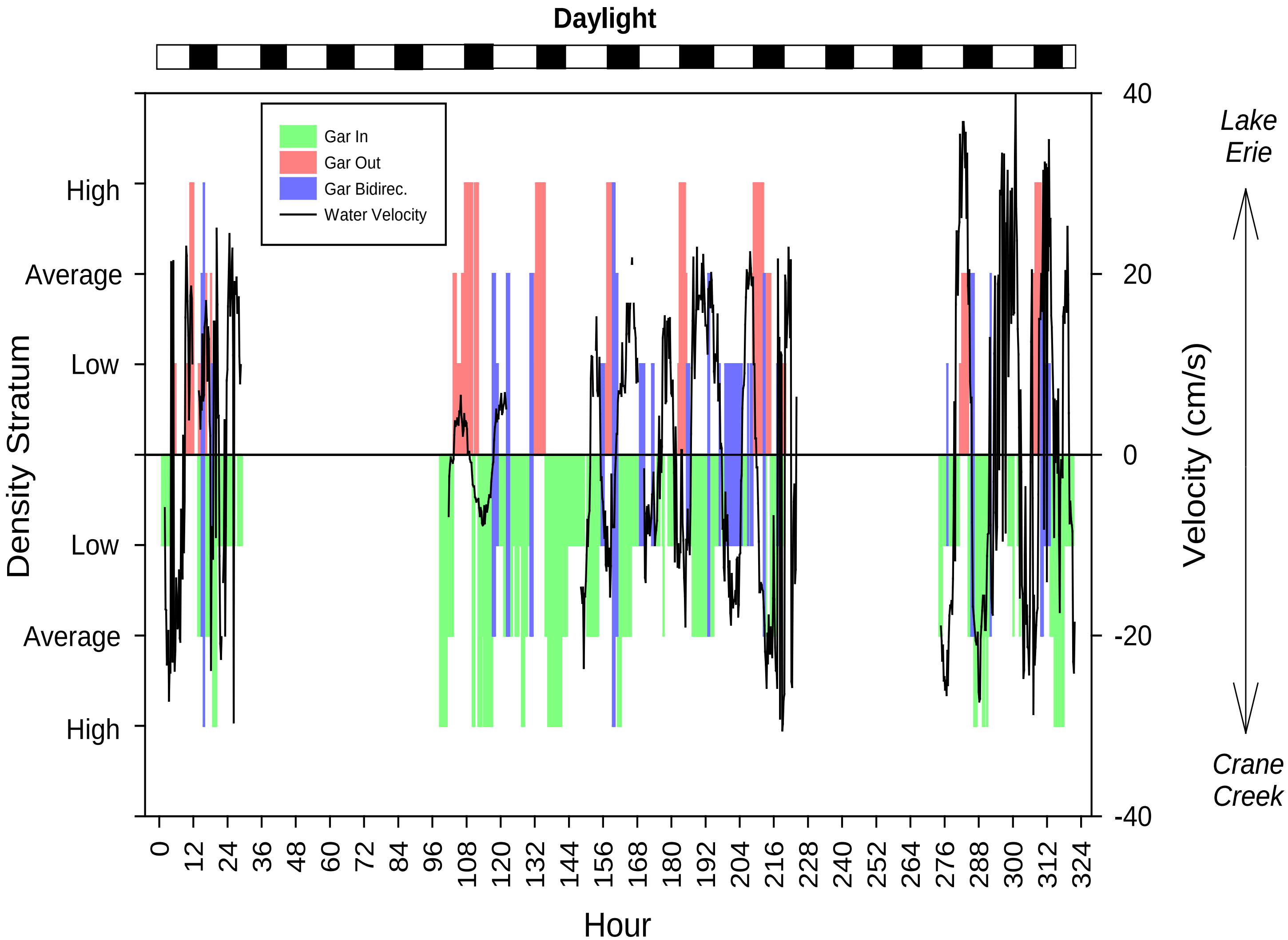


Figure 4.17. Categorical density data for Other Fish observed with the DIDSON technology. Velocity data for water moving through the channel to Lake Erie. Positive values indicate water flowing from Crane Creek toward Lake Erie.

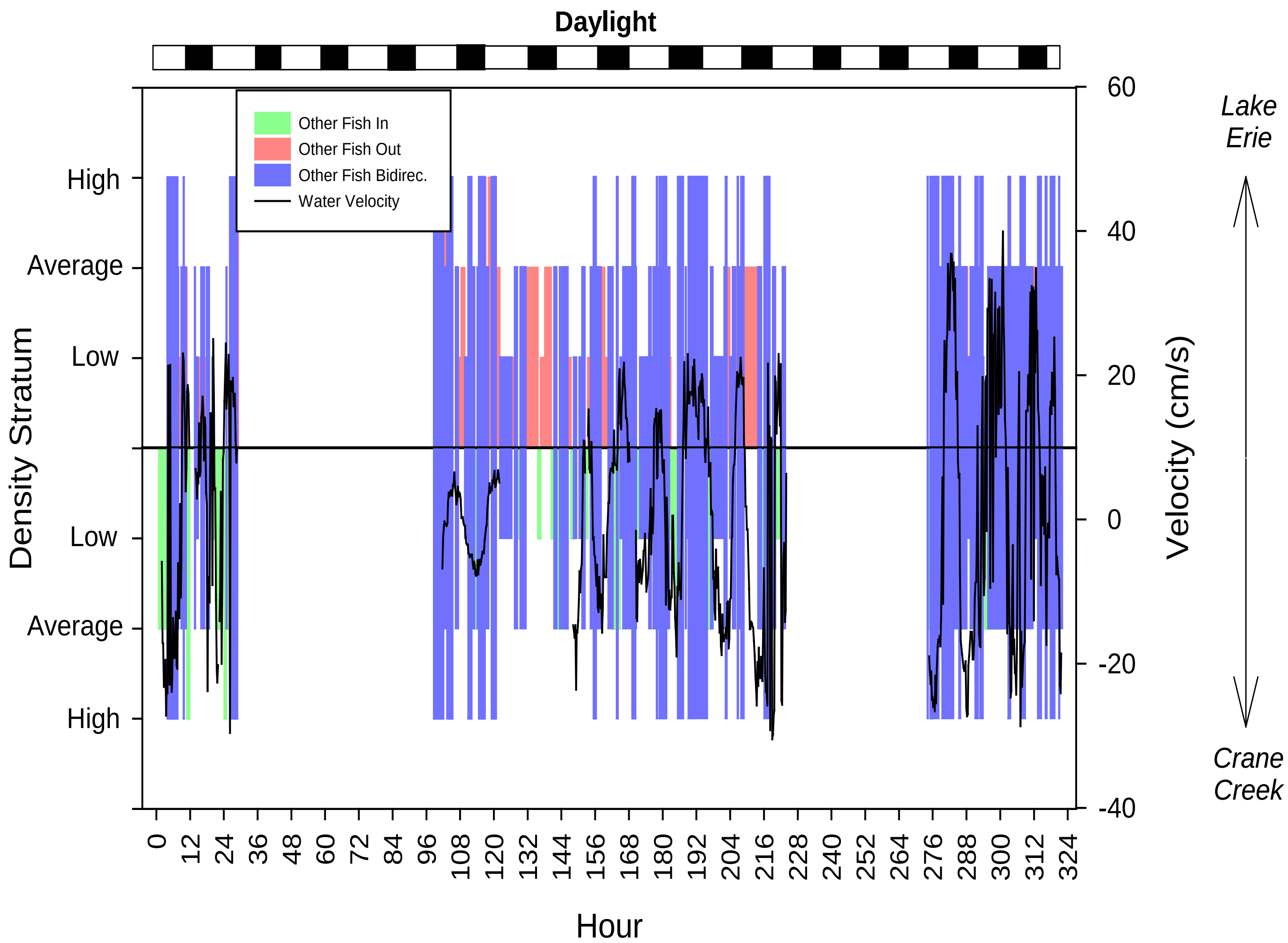


Figure 4.18. Number of 15-minute time periods classified as having high density of longnose gar moving out into Lake Erie from Crane Creek.

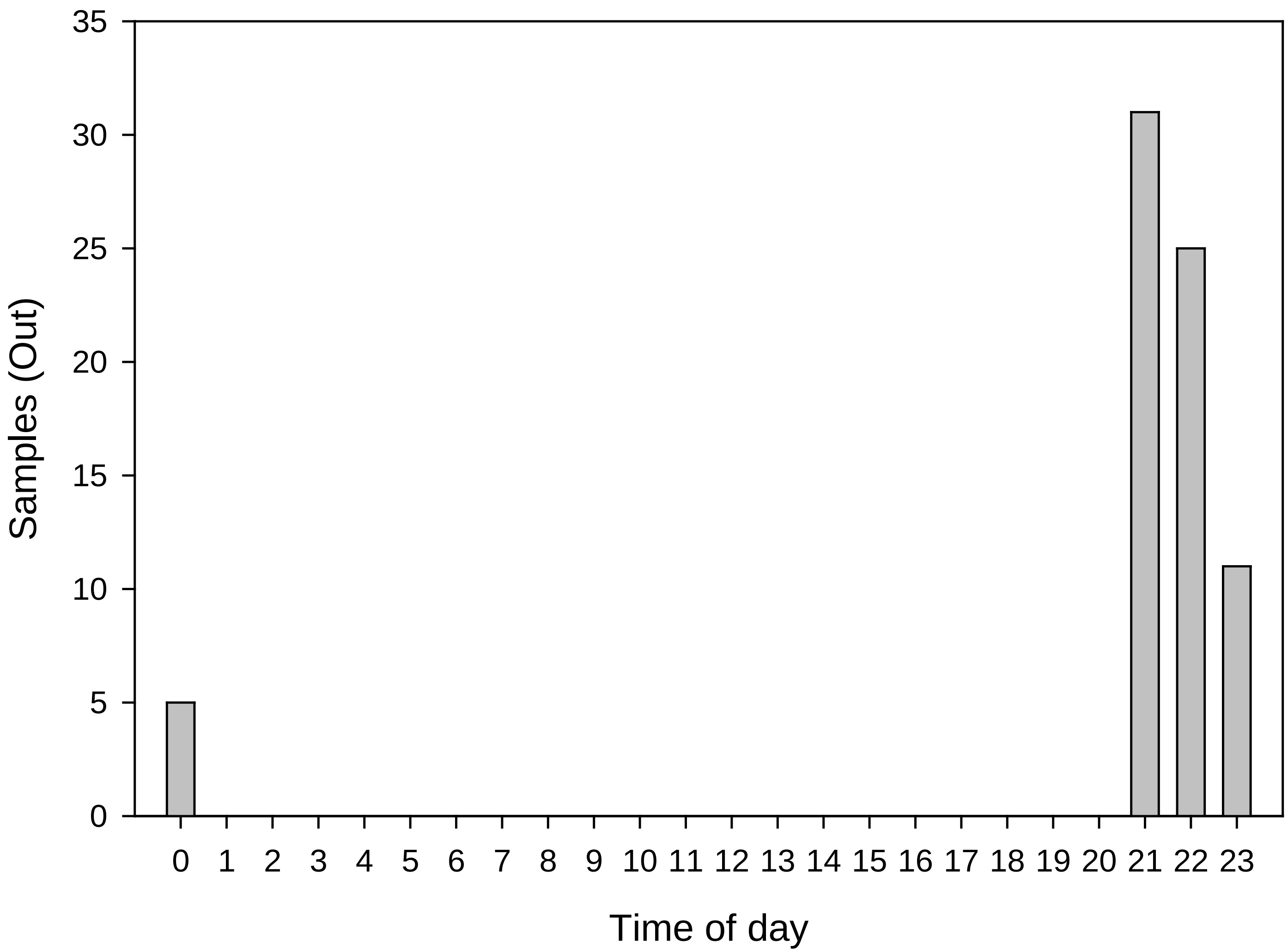


Figure 4.19. Number of 15-minute time periods classified as having average density of longnose gar moving in from Lake Erie to Crane Creek.

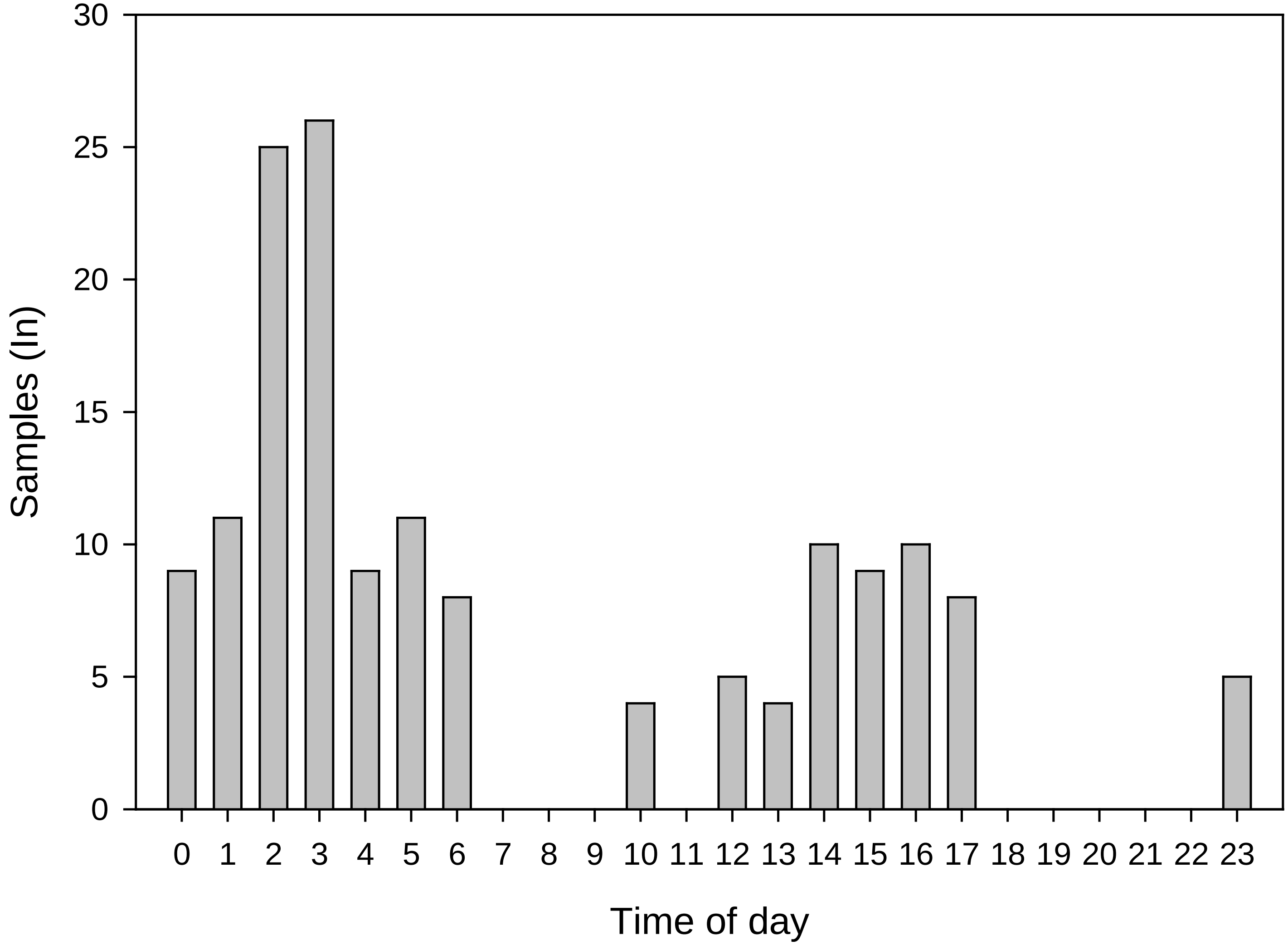
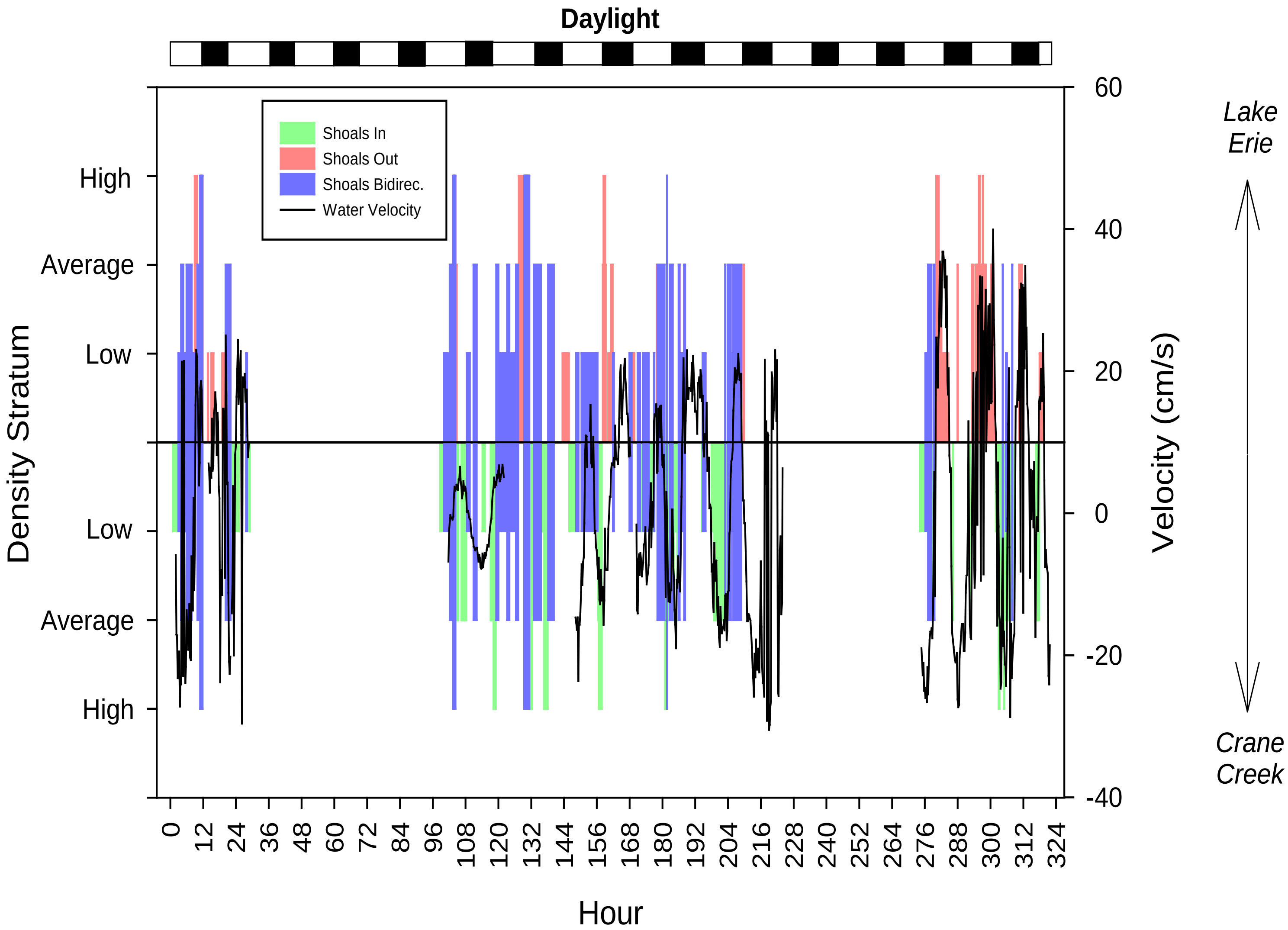


Figure 4.20. Categorical density data for Shoals of prey fish observed with the DIDSON technology. Velocity data for water moving through the channel to Lake Erie. Positive values indicate water flowing from Crane Creek toward Lake Erie.



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Chapter 5
Conclusions

5.1 Purpose

Coastal wetland habitats in western Lake Erie are in bad shape. Historically expansive and diverse systems have been reduced to a limited number of highly degraded coastal wetlands or actively managed diked wetlands, generally associated with tributaries that drain agricultural watersheds. Poor water quality, shoreline armoring, invasive species, and altered hydrology are a few of the key categories of stressors affecting these highly complex and dynamic systems. The multidimensional nature of these problems (i.e., many different factors contribute to degraded conditions) complicates habitat rehabilitation because the benefits realized by individual rehabilitation actions targeting specific problems (e.g., herbicide application to manage invasive plant species) can interact with other dimensions of the problem (e.g., altered hydrology) to produce unanticipated and possibly unwelcome consequences. The optimal combination of rehabilitation actions is not clear, given the complexity of wetland ecosystems and associated stressors, but management actions that mimic and restore natural ecological function could be smart moves in multiple dimensions.

My research explored the potential benefits and drawbacks of two wetland habitat rehabilitation strategies that restore particular aspects of historical ecosystem hydrology and related ecosystem function. It was my goal to develop a comprehensive understanding of the Crane Creek wetland ecosystem's potential response to 1) short-term, management-induced dewatering to mimic cyclic low water levels and 2) hydrologic reconnection of diked wetland units. In short, I tried to evaluate whether these specific rehabilitation strategies, both related to recovering more natural hydrologic variability, might be clearly smart moves in multidimensional ecological design space.

5.2 Summary of Chapter Findings

Chapter 2 explored the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural seed-bank driven growth of emergent vegetation. A nearly two-month drawdown stimulated a rapid seed-bank response by 45 plant taxa, and bird herbivory had little effect on recovering plant species richness, regardless of the location along an elevation gradient. I found that a number of important issues must be considered for effective long-term implementation of portable cofferdam technology to stimulate activation of wetland seed banks. Important considerations include the duration of dewatering, product size, source of clean water, replacement of damaged dams, and regular maintenance. Portable cofferdams could be of particular interest to managers of highly degraded coastal wetland habitats because they have the potential to provide the benefits of short-term hydrologic isolation without causing long-term damage to wetland sediments or permanently altering the basin hydrology. Although it may not be possible to rehabilitate whole wetland complexes at once, relatively small-scale, incremental habitat rehabilitation projects can provide localized benefit to the ecosystem and, in aggregate, improve the habitat available to Great Lakes biota. The temporary and highly customizable (e.g., height, length) design of portable cofferdams also supports their repeated use in one area over time or in multiple areas within a wetland. This technology, therefore, can be a potentially important tool in the arsenal used by Great Lakes resource managers.

Chapter 3 focused on the differences in water quality, composition of plant assemblages, and the utilization by fishes of diked and undiked areas in the lower Crane Creek wetland complex. My purpose was to examine potential benefits of hydrologically reconnecting diked wetlands to Lake Erie. I found that the diked wetlands had over 50% fewer fish species than the connected wetland units and a much lower abundance, despite extensive wetland vegetation in the diked units. The abundance of most of the 52 species of fish found in Crane Creek varied seasonally, a pattern not reflected in the diked wetland fish assemblages. Therefore, I concluded that restoring long-term hydrologic connection between diked and coastal wetlands in Lake Erie could have extensive benefits to Lake Erie fishes and fisheries. However, abiotic and biotic stressors in this ecosystem (e.g., extended high water levels, degraded water quality, sediment resuspension by common carp) likely would begin degrading newly connected wetland habitats. Periodic re-isolation of diked units to allow system reset through dewatering may be necessary to maintain high quality habitat. An alternating scheme of isolation and reconnection may be the simplest way to mimic what were once common cycles of periodic drying in fully connected coastal wetlands. Additional management actions (e.g., maintaining carp-exclusion grates in water-control structures) also may be required to reduce the rate of wetland degradation.

In chapter 4, I took a more detailed look at quantifying wetland use by Lake Erie fishes. As a part of that study, I used a new technology (DIDSON acoustic camera) to track short-term fish abundance and movement in the turbid waters of Crane Creek and directly explore relationships between fish habitat use and dynamic water quality conditions characteristic of coastal wetlands. My analysis revealed that the very dynamic hydrologic, chemical, and physical conditions in the Crane Creek coastal wetland system supported a surprisingly large daily flux of Lake Erie fishes through the channel separating Crane Creek and the open lake. I estimated 4.8 million prey fish went in and out of Crane Creek during this short 2-week study, yielding a net flux of approximately 1 million fish moving from Crane Creek into Lake Erie. This translates to an estimated mean density of small prey fish in the connecting channel of 9.22 fish / m^2 (>10 fish / m^2 when larger fish are included) over the course of this study. Large diel variations in physical and chemical conditions associated with the natural rhythms of productivity and hydrology near the channel appeared to strongly influence the daily movements of fishes in the channel. The Crane Creek connecting channel was the critical link between the degraded, yet highly productive wetlands and Lake Erie. Free passage allowed large numbers of both small prey and other fishes to access wetland habitats when advantageous and escape the especially harsh water quality conditions that occurred each night of this study.

5.3 Synthesis and Strategic Discord

The multidimensional nature of coastal wetland stressors presents both challenges and opportunities for habitat rehabilitation efforts. It can be challenging to implement sustainable rehabilitation that addresses the complex suite of stressors sufficiently, especially since many of them are interrelated (e.g., low dissolved oxygen concentrations, high nutrient concentrations). However, there is great opportunity to tease apart this complexity and identify the ecological processes shaping the system and then use that information to plan smart moves toward a better position in ecosystem design space. Understanding the physical, chemical, and biological processes that shape current ecosystem condition is a necessary step in determining which management actions may be most appropriate to improve ecosystem condition, function, and ecological services (Grenfell et al. 2007).

There are many ways that coastal wetland processes might be manipulated to promote desirable functions or values (e.g., actively manage water levels, remove earthen dikes to increase fish habitat, add earthen dikes to promote waterbird habitat), but the challenge is to look for smart moves (i.e., effective, efficient, sustainable) that will minimize negative impacts on other components of the ecosystem. The results of my research suggest that using management actions to mimic natural processes (per Wilcox and Whillans 1999) could be a smart move across multiple dimensions. Specifically, mimicking low-water conditions during periods of extended high-water levels and increasing hydrologic connectivity in diked wetlands can improve multiple aspects of ecosystem function and help meet important management objectives.

To maximize benefits in ecosystem space, it is important that progress in one dimension not impact another negatively. For example, my research suggests that reconnecting diked wetlands will benefit Lake Erie fish assemblages. A full hydrologic connection throughout the entire year is needed to minimize the negative impacts on coastal fish productivity and diversity, although some benefits can be gained by maintaining a periodic connection (Rogers et al. 1994). Unfortunately, this means that a restored hydrologic connection also could restore the influence of ecosystem stressors plaguing connected coastal wetlands (e.g., extended high water levels, increased turbidity caused by carp) to the now isolated diked units. Given the highly modified shoreline geomorphologies and potential accretion/subsidence of the sediment surface inside the dikes, a restored hydrologic connection may lead to the degradation of existing wetland vegetation. High Lake Erie water levels and the hard boundaries created by earthen dikes could also increase the water depth inside newly connected wetlands and flood emergent vegetation. Deeper water and increased turbidity from input water might limit the amount of light penetrating the water column and lead to reduced submersed aquatic plant production. My work demonstrated that wetland plants can rapidly grow from established seed banks in Crane Creek, but the timing and duration of seed bank exposure will be determined by the annual water levels in Lake Erie unless management actions are taken (e.g., drawdown sustained by a portable cofferdam). Essentially, the wetland vegetation is in a different ecosystem dimension that will be impacted by a move in the hydrologic dimension.

Similarly, moving a degraded wetland ecosystem toward optimal plant or bird habitat by using hydrologic isolation (diking of units), drawdowns, and other resource-intensive management actions has historically damaged their value as fish habitat and increased the frequency of hypoxic conditions. Diking as a habitat enhancement strategy also may increase the abundance of exotic plant species (Herrick and Wolf 2005), a negative consequence that currently consumes significant management energy to control. Therefore, we should be looking to identify smart moves that account for the complexity of these coastal ecosystems and can help us move forward in both the water-level variability and habitat-connectivity dimensions.

5.4 Integrated Rehabilitation Strategies in a Landscape Context

Restoring access to valuable wetland habitat currently isolated by earthen dikes could begin by installing an appropriately-designed fish-passage structure that allows fishes of all shapes and sizes, with the possible exclusion of adult common carp (French et al. 1999), to pass through without harm. This hydrologic connection could be maintained year-round until conditions in the rehabilitated wetland become severely degraded or management objectives can no longer be met. At this point, the fish-passage structure could be closed to allow dewatering and other management actions to "reset" the wetland similar to what might occur naturally during several seasons of low water levels (Keddy and Reznicek 1985).

Using an adaptive management approach (Thom 2000) and keeping in mind Lake Erie water-level patterns, an optimal frequency and duration for temporary isolation of individual wetlands could be determined. It would be critical to isolate the wetland only long enough to reestablish perennial emergent wetland plants (e.g., two years) and address any invasive species problems. Fishes would not be able to access the wetland habitats while these actions were taking place, similar to when low water levels limit access to upslope habitats, but higher quality habitats would be made available each time the diked wetlands are reconnected. Rotation of such dewatering actions among multiple diked wetland sites would reduce loss of access to wetland habitat, as described below. Once the perennial vegetation in the project area has reestablished sufficiently (i.e., grown tall enough) to survive natural water levels, the water-control structure(s) could be opened to start the cycle again (Ball 1985). In essence, this cycle would use periodic management actions to provide intermediate levels of disturbance (i.e., low water levels) that reset the system during times of extended high water levels in Lake Erie, similar to the efforts to isolate coastal marsh temporarily using portable water-filled cofferdams (Kowalski et al. 2009; Chapter 2). Cyclic isolation would maintain coastal wetland habitat for Lake Erie fish assemblages throughout the year, allow resource managers enough control to maintain high quality wetland habitat and sustainably achieve management objectives, and minimize the costs associated with actively managed wetlands (Pankau 2008).

The benefits of such an approach could be enhanced further by taking advantage of the natural variability among coastal wetlands in the region (Landres et al. 1999). There are many coastal wetland complexes in western Lake Erie that currently are managed as independent units on the landscape rather than as components of the Lake Erie coastal wetland zone. If the view of the Operational Landscape Unit (Verhoeven et al. 2008) for managers was broadened from individual wetlands or wetland complexes to a regional wetland resource, then the natural variability and progression of wetland ecosystem conditions in response to fluctuating water levels (e.g., mudflat, shallow water with emergent vegetation, deep water with submersed vegetation) that are critical to fish and wildlife could be maintained even when local actions are implemented. For example, if high water levels in Lake Erie cause wetland conditions to change from dense vegetation to sparse vegetation to a lack of vegetation within five years after being hydrologically reconnected to Lake Erie, then a management action to reset the system every five years may be necessary to meet sustainable management objectives associated with densely vegetated wetlands. Since each wetland in the region has a unique combination of plants, seed bank, topography, soil type, morphometry, water-level history, and other characteristics, each wetland in this example will be at a different stage of the degradation at any point in time. Therefore, when one wetland is isolated for management actions, there are one or more wetlands in the region able to provide the temporarily lost functions (e.g., fish habitat). Alternatively, the isolated unit may support a new and equally important set of functions and values as it is being prepared for reconnection (e.g., mudflat habitat for migrating shorebirds).

This type of regional management strategy would also work even if annual water levels in Lake Erie are lower in the future. For example, if Lake Erie water levels were at a lower elevation than the substrate of most of the diked and coastal wetlands, then the ecosystem reset could occur when wetland units are isolated and water levels are raised through management actions (although access to water at lower elevations may be difficult given the current design of water-delivery systems). The expected succession of plants after fertile wetland sediments are exposed could be disrupted by periodic, management-driven flooding events. After shrubs and trees associated with the later stages of succession are flooded out, the system could be reconnected to Lake Erie and more cost-effective passive management strategies could be employed (Pankau 2008). Because there are many individual components to the Lake Erie coastal wetland resource, each wetland does not need to provide all functions at one time (e.g., fish habitat, bird habitat, flood mitigation). It seems that system-wide benefits could be gained by taking advantage of the natural variability of wetlands in the region.

Adaptive management principles could be applied to determine the best timing for each stage of the functional rotation and ensure that the individual and regional management objectives are being met adequately. This approach would take extensive cooperation and coordination among landowners and management agencies, including the development of geospatial data sets and decision-support systems accessible to all managers (e.g., an online GIS-based decision-support system that could generate reports on the spatial distribution of wetlands, functional condition of individual wetlands, planned management actions). A regional effort also may require a flexible and dynamic management approach that rivals the dynamic nature of the wetlands themselves. The outcome could be the rehabilitation of Lake Erie coastal and diked wetland habitats, including the functions and values that they provide, without hindering progress toward individual management objectives.

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Appendices

Appendix A. List of recommendations for future Great Lakes coastal wetland habitat rehabilitation projects involving water-filled portable cofferdams.

Site Selection:

- Plan where the cofferdams will connect to upland areas and prepare the site by removing vegetation, rocks, and other debris. Easy access to the installation site by boat, truck, and heavy equipment saves time, money, and resources for the duration of the project.
- Ensure that clean water can be transported to each of the cofferdam sections the entire time they are installed.
- Consider the impacts of dam installation and drawdown occurring at different times of the year (e.g., spring fish spawning, summer seed dispersal).
- Prior to dam installation, search and remove debris, native clams, and any other objects from the intended cofferdam location.
- A seed bank study, historical observations, or other data are needed to verify presence of a seed bank prior to implementation of cofferdam technology.
- Ensure that soft sediments in the dam location are not too deep or otherwise unsuitable for the cofferdam to seal to the bottom.
- Install highly visible rope from endpoint to endpoint to serve as a guide during installation.

Dam Characteristics:

- Calculate manufacturer-recommended cofferdam height based on evaluation of historical hourly high water levels recorded at the nearest Great Lakes water-level gage.
- Because of the high variability in water level fluctuations in the Great Lakes, use larger dams than recommended by the manufacturer to promote a tight seal on the bottom, accommodate unexpectedly high water levels, and maintain dewatered conditions long enough to allow plants emerging from the seed bank to reach maturity.
- Minimize the number of dam sections used to isolate the project area because each connection is the most likely place where the integrity of the dam will be compromised.
- Prepare for over four weeks of dam manufacture time for large projects and the time associated with shipping replacement dams from distant locations.
- Request installation of air release valves in the dam bladders. Pumps often fill dams with air that is very difficult to remove without a release valve.
- If dams are purchased with the intent to reuse, plan for a labor-intensive effort to remove sediment and debris inside dams before storage in a dry, pest-free location.
- Consider the effects of water currents and wind on dams being prepared for installation.
- Install sheets of plastic connected to the dams and anchored by chains on the bottom of the uncontrolled side of the dam to minimize leakage under the dams.
- Install an appropriately sized culvert and water-control structure to allow controlled movement of water from one side of the dam to the other.

Operation:

- After the dams are installed, attach a network of hoses to the dams to allow the dams to be filled with minimal movement of the supply pump (see Kowalski et al. 2006).
- Routinely monitor water levels inside the dams. During our project, the dams required maintenance pumping nearly twice per week to maintain their full size and minimize water movement under the dams.
- Secure a taut rope across the top of each dam segment using 2.54-cm PVC conduit driven into sediments on each side of dam. Vertical movement (i.e., inflation) of the dam can be monitored by measuring the distance between the rope and top of the dam.
- Carefully monitor the presence of small holes in the seamless cofferdam liner daily. The holes can develop at weak spots in the material or where the liner is punctured by a sharp object. Until patched, water will leak out of the holes and the volume of water inside the dams will decrease enough to change their shape significantly and compromise connections with other dams or the dams' tight seals on the marsh bottom.
- Install signs and fences to educate the public and deter vandalism.
- Install fencing in the water on the uncontrolled side of the dam to prevent damage from fish spines.

Appendix B. List of items that were or could have been useful during cofferdam installation (I) and maintenance (M).

- Aluminum trash pumps with 7.62-cm diameter light duty discharge hoses (I,M)
- PVC intake hoses (7.62-cm dia) with screens and buckets to limit sediment intake (I,M)
- Onsite fuel and oil supply (I,M)
- Spools of twine and heavy nylon rope (I,M)
- Neoprene and leather gloves (I)
- Professional grade duct tape (I,M)
- Bird deterrents on dams to prevent damage to fill tubes (M)
- Excavator or other heavy equipment to move large dams on land (I)
- High quality radios and cell phones in waterproof sleeves (I,M)
- Automotive tires and axles to allow rolled cofferdams to be transported over inflated cofferdams (I)
- Small portable boat (I,M)
- Sheets of plywood to serve as rigid platforms on dams (I,M)
- 5.08 cm x 30.48 cm treated boards to allow access to top of dam from land (I,M)
- Wagon or cart with pneumatic tires to carry equipment (e.g., water pumps) (I,M)