EFFECTS OF IMPOUNDMENT OF COASTAL WETLANDS IN GEORGIAN BAY

LONG-TERM EFFECTS OF IMPOUNDMENT ON ECOSYSTEM FUNCTIONS OF COASTAL WETLANDS IN GEORGIAN BAY

By

DALLAS TAYLOR, B.Sc.

A Thesis

Submitted to the School of Graduate Studies

In Partial Fulfillments of the Requirements

For the Degree

Masters of Science

McMaster University

McMaster University MASTER OF SCIENCE (2014) Hamilton, Ontario (Biology)

TITLE:	Long-term Effects of Impoundment on Ecosystem Functions of Coastal Wetlands in Georgian Bay
AUTHOR:	Dallas Taylor, B.Sc. (McMaster University)
SUPERVISOR:	Dr. Patricia Chow-Fraser

NUMBER OF PAGES: xii, 100

PREFACE

This Master of Science thesis is composed of two chapters formatted as individual manuscripts to be submitted for publication in peer-reviewed journals. These chapters have been put into context with a unifying theme discussed in a general introduction and conclusion. The first chapter has been submitted to the journal Wetlands, Ecology and Management and we are in the preliminary stages of considering a journal for the second chapter.

As the author of this thesis, and under the supervision of Dr. Patricia Chow-Fraser, I collected and analyzed all data presented that are not otherwise referenced and wrote both chapters presented. All field work and data were collected with the help of dedicated field technicians.

Taylor, D.R. & Chow-Fraser, P. Comparison of long-term changes in wetland communities of a diked and undiked wetland in southern Georgian Bay

Taylor, D.R. & Chow-Fraser, P. Comparison of ecosystems in a chain of beaver impounded wetlands in eastern Georgian Bay: a glimpse of the future in a predicted low water-level crisis

GENERAL ABSTRACT

Seasonal and annual water-level fluctuation is a primary mechanism that maintains high aquatic biodiversity in coastal marshes of the Laurentian Great Lakes by preventing formation of dense mono-cultures of emergent or submergent plants. The past 14 years of sustained low water levels in Georgian Bay (Lake Huron) have prompted proposals to impound coastal wetlands as a way to mitigate against desiccation. We hypothesize that a loss of hydrological connection with the Great Lakes would lead to altered water chemistry and an expansion of emergent vegetation at the expense of aquatic habitat. This could result in a fish community with reduced diversity which may positively impact the larval amphibian community. On the other hand, bird communities may respond with only short-term gains, and may not demonstrate long-term benefits without active management. We investigate how long-term severance of hydrological connection between coastal marshes and Georgian Bay affect ecosystem functions. We compare food web components of a chain of three coastal wetlands in Tadenac Bay that were sequentially impounded by beaver activities over the past 5 decades. We also surveyed the food web of Wye Marsh, which was impounded over 80 years ago, and compare it with that of Matchedash Bay, which had never been impounded. In both cases, there were negative impacts of impoundment on water chemistry, vegetation and fish communities. Bird communities showed no significant differences but larval amphibian diversity was greater in the beaver impoundments than in the hydrologically connected coastal wetland. Based on our results, we do not recommend impounding Georgian Bay wetlands as a means to restore and maintain water levels.

ACKNOWLEDGEMENTS

I would like to thank everyone who has been so supportive in my work over the past two years. To Pat Chow-Fraser, you are an incredible mentor and I appreciate all of the guidance and support you have provided me in my Graduate and Undergraduate degrees. I count myself lucky to have been able to take part in the Biodiversity program and a lab of people so close to one another. Thank you, Mike Waddington, for your help in the design of my second year of field sampling and the written components of my thesis. You and Pat truly make a dynamic duo. Thank you, Rosa da Silva, for being on my examination committee and your enthusiasm surrounding my defence.

I could not have completed this research without the continued financial support by the Sierra Club Canada Foundation and their members, including Mary Muter, who is not only an inspiration to the Georgian Bay community but provided my team accommodations, a boat and endless knowledge of the area.

Thank you to the Wye Marsh Wildlife Centre, Friends of Wye Marsh and the Tadenac Club for access to your properties and for supporting this lab's research. Kind support was provided by Mark Trudeau, Sara Street and Paul Beachamp who were there to assist us on numerous occasions in these locations. We appreciate the accommodations put up by Amy Beament and her family and support from the Ontario Ministry of Natural Resources, as well.

Amanda Fracz was an endless support system throughout this entire experience. She provided love, help and support that I never could have expected. I would also like to extend my appreciation for the help and support of my fellow students and colleagues, starting with Chantel Markle, Chris Biberhofer and Rachel Kelly; you are all one of a kind. Thank you to Julia Rutledge, John Paul LeBlanc, Dan Weller, James Marcaccio, Lyndsay Smith-Cartwright and all undergraduate students that helped me in lab meetings and in the field.

Finally, I would like to thank my parents Dianne and Brett Taylor and my brother Jeff Taylor for having gotten me to this point and for their continued love and support.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

GENERAL INTRODUCTION

Wetlands are characterized as ecosystems that are either permanently or temporarily (seasonally or annually) saturated with water (Keddy and Reznicek 1986), at least periodically support hydrophytes (Keough et al. 1999) and are ecologically and economically important landscape components (Mitsch and Gooselink 2000). As links between terrestrial and aquatic ecosystems, wetlands are some of the most diverse habitats in the world. They act as natural water purifiers, erosion barriers, breeding grounds and foraging habitat for numerous taxa above and below the surface of the water. Wetlands store and recycle nutrients, remove toxins from the water, and provide vegetative structure to hold sediments and reduce erosion (Mitsch 1992; Mitsch and Gooselink 2000). The general flow and focus of this thesis are outlined in Figure 0.1. This thesis is set up in a manner that introduces coastal wetland ecology and how sustained low-water levels in Georgian Bay may affect the natural cycling of wetlands. We have discussed one proposed management solution and our two research studies (chapters 1 and 2) on how that management action may affect Georgian Bay coastal wetland ecosystem function. We bring this research full circle by discussing how our results relate back to those original factors that make coastal wetlands so biologically diverse.

Wetland Classes

According to the National Wetlands Working Group (1997), there are five classes of wetlands: bogs, fens, swamps, marshes and shallow water. Each wetland class provides unique physical characteristics that ultimately drive the vegetative community. Bogs are raised lands primarily composed of sphagnum moss underlain by peat. Unlike other types of wetlands, water is received solely from precipitation and without any influx from runoff or groundwater. Their unique hydrological regimes produce highly acidic water that is low in minerals and nutrients. Fens are peatlands that are rich in dissolved minerals that obtain water from a fluctuating water table fed by ground and surface water. Fens are typically dominated by graminoid vegetation and some bryophytes but vegetation structure is determined by the hydrological patterns of the wetland. Swamps are wooded wetlands underlain by highly decomposed woody peat. Water is typically below the soil surface and rich in dissolved minerals. Marshes are wetlands with shallow water that is highly variable in depth. Often found along coasts of larger bodies of water, marshes are wetted from many sources including surface runoff, groundwater, lake or bay water and precipitation. They can vary in the amount of organic matter buildup and acidity based on hydrological processes. Dominant vegetation in marshes includes emergent, floating-leaved and submerged aquatic macrophytes, as well as non-vascular plants such as mosses. Shallow water wetlands are found at depths of up to 2 m (Keough et al. 1999), between deeper open water and shores. They are predominantly structured by submerged aquatic vegetation. They are highly influenced by limnetic and hydrological processes of the deeper water and will vary depending on the region in which they are found (National Wetlands Working Group 1997).

Laurentian Great Lakes Coastal Wetlands

The Laurentian Great Lakes are the largest freshwater source in the world (Herdendorf 2004). With an area of 244,160 km^2 and volume of 22,700 km^3 (Herdendorf

2004), the Great Lakes are bordered by 8 states and 1 province. The coastline of these lakes spans 17,017 km (Herdendorf 2004), which naturally provides optimal conditions for biologically rich and diverse coastal wetlands. These coastal wetlands, usually marshland or shallow water wetlands are either permanently or seasonally connected to the Great Lakes and act as transition zones between the water and terrestrial ecosystems (Keough et al. 1999). These wetlands are often subject to high disturbance regimes as lake water levels can vary daily, seasonally and inter-annually (Keddy and Reznicek 1986) and a reduction in the annual lake level variation can lead to a loss of areal extent, biodiversity and resilience in wetlands (Keough et al. 1999).

Coastal wetlands have been further classed based on hydrological connectivity and geomorphic position (Keough et al. 1999; Albert et al. 2005). Coastal wetlands can either be lacustrine, riverine or barrier protected as characterized by Keough et al. (1999). Lacustrine wetlands are permanently connected to the lake, and their primary hydrological influence is lake water. Riverine systems are predominantly found along rivers that feed into the Great Lakes. These systems are hydrologically dominated by the rivers themselves, and lake water has less of a dominant role. Barrier protected wetlands have a geologic feature forming a barrier around the wetland. The wetland can be connected to the lake by a channel or high water levels and therefore be influenced by lake water or, if disconnected from the lake, can be predominantly fed by groundwater and surface runoff.

Lake Huron was noted to contain the greatest wetland area on the Canadian side of the Great Lakes, with 16,179 ha (Ingram et al. 2004). However, this was a vast underestimate as Midwood et al. (2012) inventoried Georgian Bay much more extensively showing that in Georgian Bay alone wetland area is an estimated 17,350 ha. Following Lake Huron, Canadian wetland area is next largest in Lake Ontario (11,777.2 ha), Lake Erie (11,417.5 ha), and finally Lake Superior (2,235.8 ha)(Ingram et al. 2004).

Threats and Conservation of Coastal Wetlands

Coastal wetlands have been subject to many direct and indirect threats including human development (Jude and Pappas 1992; Keough et al. 1999), invasive species (Chow-Fraser 2005) and global climate change (Mortsch and Quinn 1996; Angel and Kunkel 2010). Highly productive wetland soils and their proximity to the coast have made wetland infilling for agriculture and development a common practise. Human development has been especially impactful in the southern Great Lakes. In southern Ontario, approximately 75% of wetlands have been lost (Findlay and Houlahan 1997) and the western end of Lake Erie alone has seen a 95% reduction in coastal wetlands since European settlement (Mitsch and Wang 2000). Furthermore, those remaining wetlands are often subject to human-induced stressors such as pollution, increased runoff and accelerated eutrophication (Chow-Fraser 2006; Cvetkovic and Chow-Fraser 2011) as well as bioturbation from the invasive common carp (Chow-Fraser 2005). Furthermore, global climate change has been noted to have an impact on coastal wetlands and is projected to lead to further changes in air temperature, precipitation, surface runoff, snow cover, length of freezing season, ground water storage and evapotranspiration (Mortsch 1998). This in turn can affect the natural balance in these semi-aquatic ecosystems. For example, water levels are expected to decline (Mortsch and Quinn 1996; Mortsch 1998;

Sellinger et al. 2008; Angel and Kunkel 2010), which can cause coastal wetlands to dry up or migrate lakewards and lead to decreased structural habitat and biodiversity (Midwood and Chow-Fraser 2012).

Georgian Bay's Coastal Wetlands

Georgian Bay's coastal zone is among the most pristine coasts in the Great Lakes (Cvetkovic and Chow-Fraser 2011). With 12,629 identified wetland units comprising 17,350 ha of marshland and upstream habitat (Midwood et al. 2012), it is a significant contribution to Great Lakes wetlands and wetland processes. Numerous studies have demonstrated that coastal wetlands in Georgian Bay have remained relatively pristine compared to other Great Lake coastal wetlands (Chow-Fraser 2006; Seilheimer and Chow-Fraser 2006; Croft and Chow-Fraser 2007; Seilheimer and Chow-Fraser 2007). This provides us with reference conditions from which we can draw information without major human disturbances. Major threats to Georgian Bay coastal wetlands include climate change and road and cottage development (deCatanzaro et al. 2009; Fracz and Chow-Fraser 2012b). Currently, water levels in Lakes Michigan-Huron are well below their long term mean and have remained in such a state for the past 14 years. These low water levels and dampened annual fluctuations risk damaging coastal wetland ecosystems by removing the hydrological variability known to maintain high wetland structure and biodiversity (Keddy and Reznicek 1986). Lakes Michigan and Huron are the only two Laurentian Great Lakes that do not have water levels controlled in some way, and dredging of the St. Claire River along with climate change risk further water level declines and degradation of these pristine wetlands.

Importance of Hydrological Connection

Increased evaporation and changing precipitation patterns associated with global climate change are expected to lead to long-term reductions in water levels and reduced hydrological connection between coastal wetlands and Georgian Bay (Fracz and Chow-Fraser 2012a). However, hydrological connection (hereafter meaning a connection with a larger body of water such as Georgian Bay and not referring to the connection to the watershed) is an important means of energy transfer between these ecosystems (Pringle 2003). It allows for transfer of water, organics and aquatic species migration (Jude and Pappas 1992; Pringle 2003; Wei et al. 2004). Pringle (2003) noted that the understanding of effects of hydrological connectivity on ecosystems at the landscape level is lacking, but is a fundamental consideration for land-use decisions. We need to understand how losses of hydrological connection affect ecosystem components before we can consider how to mitigate the effects.

Thesis Objectives

One proposed solution to water level declines and wetland loss in Georgian Bay is to impound coastal wetlands, thereby removing hydrological connection to the bay and maintaining water within the wetlands (McCutcheon 2011; Sierra Club Canada 2012). Our main objective in the following study was to assess the effects of impoundment in Georgian Bay coastal wetlands to understand how ecosystem function may change. To our knowledge, no other literature has investigated how impoundment of coastal wetlands in Georgian Bay affects the ecosystem function as a whole and few studies have investigated the long-term effects of this proposed hydrological change. I used research conducted in the lower Great Lakes to guide my hypotheses and to interpret my findings. I investigated the long-term effects of impoundments in Georgian Bay by looking at various ecosystem factors in two chapters; including effects on aquatic habitat availability, water chemistry, water level regimes, and community structure of aquatic macrophyte, fish, birds, and anurans. To do so, I compared impounded wetlands to sites that have remained hydrologically connected to Georgian Bay, by (1) comparing a wetland diked for over 8 decades to one which has remained hydrologically connected to the bay and by (2) comparing a chain of three beaver impounded wetlands in a reference site.

Literature Cited

Albert DA, Wilcox DA, Ingram JW, Thompson TA (2005) Hydrogeomorphic Classification for Great Lakes Coastal Wetlands. J Great Lakes Res 31 (Supplement 1):129-146.

Angel JR, Kunkel KE (2010) The Response of Great Lakes Water Levels to Future Climate Scenarios with an Emphasis on Lake Michigan-Huron. J Great Lakes Res 36:51-58.

Chow-Fraser P (2005) Ecosystem Response to Changes in Water Level of Lake Ontario Marshes: Lessons from the Restoration of Cootes Paradise Marsh. Hydrobiologia 539:189-204.

Chow-Fraser P (ed) (2006) Development of the Water Quality Index (WQI) to Assess Effects of Basin-Wide Land-Use Alteration on Coastal Marshes of the Laurentian Great Lakes. Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Indiana Biological Survey, Bloomington, IN.

Croft MV, Chow-Fraser P (2007) Use and Development of the Wetland Macrophyte Index to Detect Water Quality Impairment in Fish Habitat of Great Lakes Coastal Marshed. J Great Lakes Res 33 (Special Issue 3):172-197.

Cvetkovic M, Chow-Fraser P (2011) Use of Ecological Indicators to Assess the Quality of Great Lakes Coastal Wetlands. Ecol Indic 11 (6):1609-1622.

deCatanzaro R, Cvetkovic M, Chow-Fraser P (2009) The Relative Importance of Road Density and Physical Watershed Features in Determining Coastal Marsh Water Quality in Georgian Bay. Environ Manag 44:456-467.

Findlay CS, Houlahan J (1997) Anthropogenic Correlates of Species Richness in Southeastern Ontario Wetlands. 11 (4):1000-1009.

Fracz A, Chow-Fraser P (2012a) Impacts of Declining Water Levels on the Quantity of Fish Habitat in Coastal Wetlands of Eastern Georgian Bay, Lake Huron. Hydrobiologia 702 (1):151-169.

Fracz A, Chow-Fraser P (2012b) Importance of Hydrologic Connectivity for Coastal Wetlands to Open Water of Eastern Georgian Bay. Master's Thesis, McMaster University, Digital Commons Open Access Dissertations and Theses

Herdendorf CE (2004) Morphometric Factors in the Formation of Great Lakes Coastal Wetlands. Aquat Ecosyst Health and Manag 7 (2):179-197.

Ingram J, Holmes K, Grabas G, Watton P, Potter B, Gomer T, Stow N (2004) Development of a Coastal Wetland Database for the Great Lakes Canadian Shoreline. Final Report To: The Great Lakes Commission. Ann Arbor, MI.

Jude DJ, Pappas J (1992) Fish Utilization of Great-Lakes Coastal Wetlands. J Great Lakes Res 18 (4):651-672.

Keddy PA, Reznicek AA (1986) Great Lakes Vegetation Dynamics: The Role of Fluctuationg Water Levels and Buried Seeds. J Great Lakes Res 12 (1):25-36.

Keough JR, Thompson TA, Guntenspergen GR, Wilcox DA (1999) Hydrogeomorphic Factors and Ecosystem Responses in Coastal Wetlands of the Great Lakes. Wetlands 19 (4):821-834.

McCutcheon A (2011) Georgian Bay Advocate Nixes Dike Proposal. The Manitoulin Expositor.

Midwood JD, Chow-Fraser P (2012) Changes in Aquatic Vegetation and Fish Communities Following 5 years of Sustained Low Water Levels in Coastal Marshes of Eastern Georgian Bay, Lake Huron. Global Change Biol 18 (1):93-105.

Midwood JD, Rokinicki-Wojcik D, Chow-Fraser P (2012) Development of an Inventory of Coastal Wetlands for Eastern Georgian Bay, Lake Huron. ISRN Ecology 2012:13.

Mitsch WJ (1992) Combining Ecosystem and Landscape Approaches to Great-Lakes Wetlands. J Great Lakes Res 18 (4):552-570.

Mitsch WJ, Gooselink JG (2000) The Value of Wetlands: Importance of Scale and Landscape Settings. Ecol Econ 35 (200):25-33.

Mitsch WJ, Wang N (2000) Large-Scale Coastal Wetland Restoration on the Laurentian Great Lakes: Determining the Potential for Water Quality Improvement. Ecol Eng 15:267-282.

Mortsch LD (1998) Assessing the Impact of Climate Change on the Great Lakes Shoreline Wetlands. Clim Change 40 (2):391-416.

Mortsch LD, Quinn FH (1996) Climate Change Scenarios for Great Lakes Basin Ecosystem Studies. Limnol Oceanogr 41 (5):903-911.

National Wetlands Working Group (1997) The Canadian Wetland Classification System (Second Edition). Wetlands Research Centre, University of Waterloo.

Pringle C (2003) What Is Hydrologic Connectivity and Why Is It Ecologically Important. Hydrological Processes 17:2685-2689.

Seilheimer TS, Chow-Fraser P (2006) Development and Use of the Wetland Fish Index to Assess the Quality of Coastal Wetlands in the Laurentian Great Lakes. Can J Fish Aquat Sci 63 (2):354-366.

Seilheimer TS, Chow-Fraser P (2007) Application of the Wetland Fish Index to Northern Great Lakes Marshes with Emphasis on Georgian Bay Coastal Wetlands. J Great Lakes Res 33 (3):154-171.

Sellinger CE, Stow CA, Lamon EC, Qian SS (2008) Recent Water Level Declines in the Lake Michigan-Huron System. Environmental Science & Technology 42 (2):367-373.

Sierra Club Canada (2012) The International Upper Great Lakes Study Board Released Their Final Report Today. <http://www.sierraclub.ca/en/media/release/international-upper-great-lakes-study-board-released-their-final-report-today>.

Wei A, Albert D, Chow-Fraser P (2004) Influence of Shoreline Features on Fish Distribution in the Laurentian Great Lakes. Can J Fish Aquat Sci 61 (7):1113-1123.

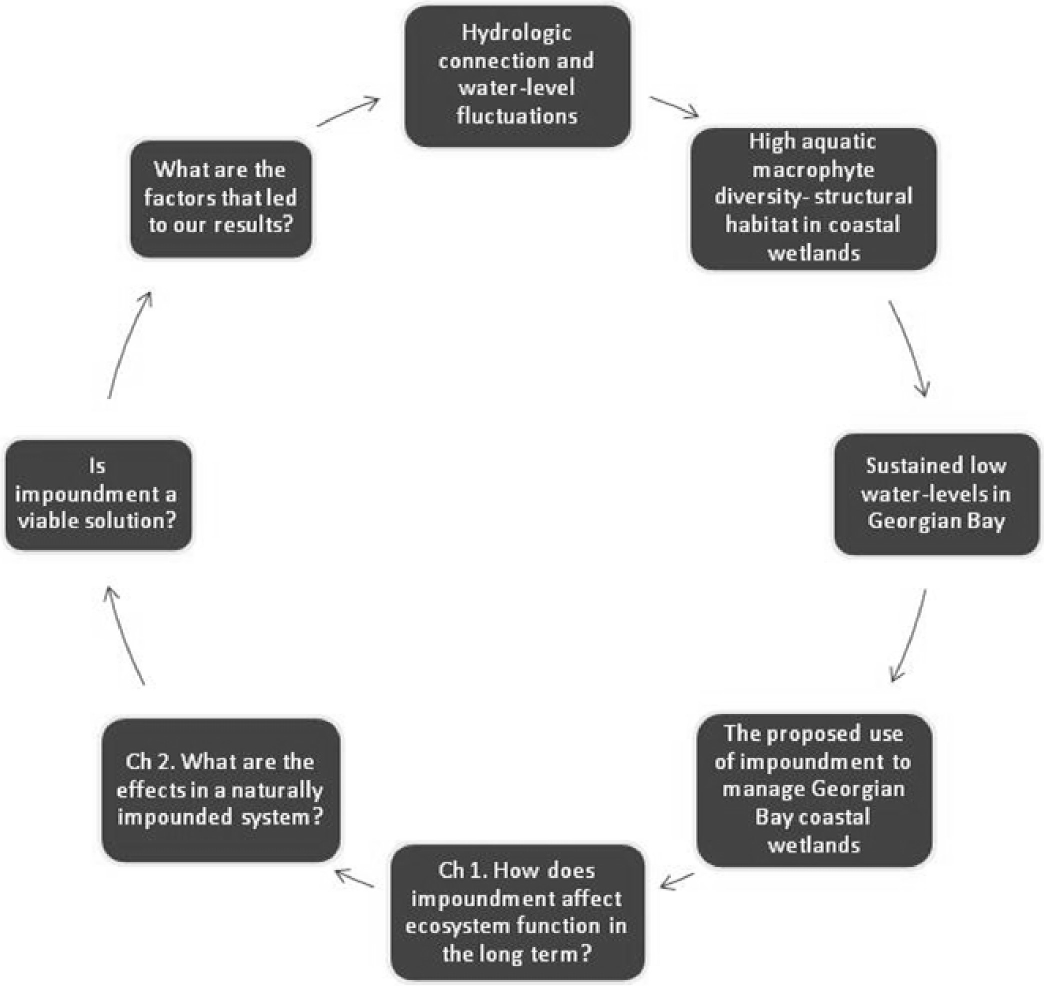

Figure 0.1 Flow chart demonstrating the linkage between published research and chapters in this thesis. We start at the top with hydrologic connection and follow this cyclical format into two chapters of research before concluding about the factors that led to the results we observed.

CHAPTER 1: Comparison of long-term changes in wetland communities of a diked and undiked wetland in southern Georgian Bay

By

Dallas R. Taylor
and
Patricia Chow-Fraser

McMaster University
Department of Biology, 1280 Main St. West
Hamilton, ON, L8S 4K1, Canada
Corresponding author telephone: 905-525-9140 ext. 27461
Corresponding author e-mail: taylod7@mcmaster.ca

Keywords: Laurentian Great Lakes, impounded wetlands, long-term, emergent vegetation, fish, birds

Abstract

Hydrological connection between coastal wetlands and the Laurentian Great Lakes plays an important role in maintaining aquatic biodiversity in the wetlands by preventing monocultures of emergent vegetation from forming, by facilitating frequent exchange of chemical constituents between the wetlands and lakes, and by allowing daily and seasonal migration of fish in and out of the wetlands. We hypothesize that when a wetland becomes impounded or diked, the emergent vegetation will expand at the expense of aquatic habitat (open water and submersed aquatic vegetation), the water chemistry in the marsh will become altered, and the diversity of the fish community will become reduced. Conversely, there should be no long-term impact on avian diversity if water levels are not actively manipulated to maintain bird habitat. We tested these hypotheses by comparing changes in the wetland communities of Wye Marsh, a diked wetland in Georgian Bay, with that of a nearby undiked wetland, Matchedash Bay. We used available historic air photos (from 1930 to 2008) to quantify the amount of aquatic habitat in both wetlands. Consistent with our prediction, the amount of aquatic habitat decreased significantly through time in Wye Marsh, but not in Matchedash Bay; instead, area of open-water in the undiked wetland varied directly with mean water levels of Georgian Bay. Water chemistry in both wetlands reflected surrounding agricultural land-use and exhibited differences that could be predicted on the basis of the hydrological connection with Georgian Bay. Whereas diversity of the fish community in Wye Marsh was significantly lower than that in Matchedash Bay, the avian diversity showed no significant differences.

Introduction

Water level fluctuations in large lakes, such as the Laurentian Great Lakes, are an important driver for many nearshore habitats. They provide a disturbance regime at the land-water interface by fluctuating at seasonal, annual and inter-annual cycles (Keddy and Reznicek 1986; Jude and Pappas 1992; Wilcox 2004). Coastal wetlands, which are found at that interface, experience cyclical succession following these physical disturbances (Quinlan and Mulamoottil 1987; Wilcox and Meeker 1992; Chow-Fraser et al. 1998; Wilcox 2004). Since the fluctuations are so prevalent, secondary succession of aquatic vegetation communities is constantly being restarted at new water-levels, where different species hold a competitive advantage (Keddy and Reznicek 1986; Quinlan and Mulamoottil 1987). The succession within the coastal wetlands is continuously being cycled through, preventing the wetlands from ever reaching climax communities (Keddy and Reznicek 1986; Jude and Pappas 1992; Wilcox 2004). At high water-levels, submergent and floating vegetation dominate and at low water-levels, emergent vegetation dominates (Chow-Fraser et al. 1998). As the water-levels rise, emergent vegetation is drowned out and space is opened up for submergent and floating species. When water levels decline, seeds of emergent vegetation germinate from the seed bank and grow in the shallow reaches of the wetlands (Keddy and Reznicek 1986; Wilcox and Nichols 2008).

Given that water-level fluctuation is a key determinant of macrophyte growth and that macrophytes provide the basic structural habitat of coastal wetlands, a loss of natural fluctuation through impoundment should lead to substantial alteration of ecosystem processes. The extent of emergent vegetation cover tends to be greater in diked than in undiked wetlands (Mitsch 1992; Sherman et al. 1996; Johnson et al. 1997; Galloway et al. 2006). In addition, Gottgens et al. (1998) found that the ratio of emergent vegetation to open water in an impounded marsh was more constant through time compared to a similar system that was hydrologically connected to the lake. Disruption of hydrological connection between a wetland and the lake can also interfere with exchange of chemical constituents such as nutrients, sediment and algae (Mitsch 1992; Fracz and Chow-Fraser 2013). In Georgian Bay, Fracz and Chow-Fraser (2013) found that the water chemistry in beaver-impounded wetlands changed significantly because of a lack of mixing with the open water of the bay. Whereas nutrients and sediments accumulated behind beaver dams and created a more turbid and nutrient-rich environment, similar wetlands that were hydrologically connected with Georgian Bay had more clear water, with water chemistry that reflected the geology of the region (deCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013). Severance of hydrological connection may also affect the movement of biota into and out of wetlands. Keast and Fox (1990) found a lower species richness of fish within a system that was naturally impounded by beavers in Ontario, Canada. This is consistent with other studies that identified hydrological connection as the major driver for the presence of top predators (Snodgrass et al. 1996; Barber et al. 2002; Bouvier et al. 2009) and may explain why invertebrate diversity has been shown to increase in impounded ecosystems (McLaughlin and Harris 1990). Within the Great Lakes context, there are many fish species that migrate into coastal wetlands to spawn and feed (Jude and Pappas 1992; Wei et al. 2004), and an impoundment or dike should similarly disrupt such migrations and lead to cascading effects on the food web.

It is equally important to determine those species that are unaffected or positively affected by diking and impoundments. Galloway et al. (2006) found that impounded wetlands had higher abundance and species richness of marsh-nesting bird species as well as overall bird abundance. Few statistically significant differences were found, however, once they controlled for differences in geographic location and wetland characteristics. Increased waterfowl use was another benefit, but this was dependent on the type of vegetation cover in the diked marsh, which varied according to site characteristics, time since impoundment and local management strategies. Monfils et al. (2014) showed similar results, with few differences between bird communities when directly comparing diked and undiked wetlands. Kadlec (1962) found that impoundments had only short-term benefits for waterfowl because the vegetation structure required continuous water-level manipulations that are expensive to maintain. Therefore, after 5 years of impoundment without active water-level manipulation to encourage cyclic succession patterns in the plant community, waterfowl habitat began to decline in Michigan wetlands.

Within the Great Lakes basin, there have been relatively few published studies on the short-term or long-term effects of impoundment on wetlands (e.g. Johnson et al. 1997; Gottgens et al. 1998; Galloway et al. 2006). In their preliminary assessment of wetland diking as an adaptation strategy to long-term water level reductions brought on by global climate change, Galloway et al. (2006) noted that we must have a broader understanding of how dikes and dams affect ecosystem functioning before we promote their use as an adaptation strategy. Given that most of the studies in the past have focused on coastal marshes in Lakes Erie and Ontario, there is an obvious need to expand the geographic focus to include the other Great Lakes. A case in point is Georgian Bay, the eastern arm of Lake Huron, where sustained low water-levels over the past 14 years (Figure 1.1) have resulted in major losses of wetland habitat (Fracz and Chow-Fraser 2012). No literature exists that documents the long-term or even short-term impact of diking on wetlands that occur along the eastern and southern shoreline of Georgian Bay. Hence, no data are available to guide decisions regarding the appropriate use of dikes or dams as an adaptation strategy. We should be cautious when extrapolating results from studies of wetlands in Lakes Erie and Ontario directly to those in eastern Georgian Bay since Georgian Bay wetlands are geologically unique (deCatanzaro and Chow-Fraser 2011), have vastly better water-quality scores and more diverse biotic communities compared with those in the lower Great Lakes (Cvetkovic and Chow-Fraser 2011). Similarly, Lakes Huron and Michigan experience hydrological regimes with water-level maxima and minima that are quite different from those in the other Great Lakes (see Figure 1.2). Water level peaks are a month later than the lower Great Lakes and two months earlier than Lake Superior.

The primary goal of this study was to document the long-term effects of diking and damming on Wye Marsh, a wetland located in southeastern Georgian Bay that has been impounded at least since the 1930s. We quantified the amount of open aquatic habitat in Wye Marsh using air photos that span eight decades (1930 to 2008) and compare them with similar measurements of Matchedash Bay, a wetland located 20 km east of Wye Marsh (see Figure 1.3) which has been hydrologically connected with waters of Georgian Bay during the past 8 decades. Both marshes are large, provincially significant wetlands with flat bathymetry and cattails (*Typha spp.*) as the dominant emergent vegetation. Both wetlands are also impacted by surrounding agricultural land uses. These similarities enabled us to make valid comparisons since both wetland size and basin morphometry, as well as watershed characteristics can influence how the respective ecosystems will respond to impoundments (see Galloway et al. 2006). We hypothesized that the amount of aquatic habitat in Wye Marsh would decrease through time because of succession of the emergent community whereas aquatic habitat in Matchedash Bay would be significantly and positively related to water levels in Georgian Bay in a manner consistent with coastal marshes elsewhere (see Chow-Fraser et al. 1998). Since Wye Marsh water levels are not actively managed, we did not expect the marsh bird community to show any significant benefits from impoundment. We did expect the fish species diversity in Wye Marsh to be lower than that in Matchedash Bay due to lack of access for migratory species. This study is the first to document the impacts of impoundment on a Georgian Bay wetland, and will provide a basis by which managers can judge the long-term implications of loss of hydrological connection on ecosystem functions in one of the most biologically diverse ecosystems of the Great Lakes.

Methods

Site descriptions

Wye Marsh is an approximately 600 ha provincially significant wetland that has been deemed an Important Bird Area and an Area of National Scientific Interest with sections designated as both Provincial Wildlife Areas and National Wildlife Areas. It is found 1.8 km upstream of Georgian Bay, Lake Huron in Tay, Ontario, Canada. A 19,600 ha agriculturally impacted watershed drains into the marsh which is underlain primarily by limestone (Maccrimmon 1980). The primary input of the marsh is the Wye River (Bufo Inc. 1978). It is unique for a wetland of its size and geographic location with respect to its management history as it has been impounded for more than 8 decades at the northern outlet where it would otherwise be connected to Georgian Bay. There is no record of the construction of the 70-m wide St. Marie Dam. A historical 1930 aerial image of the region confirms the dam's existence to before that date. The dam did, however, break once in 1972, returning hydrological connection of the marsh to the bay until the dam was rebuilt shortly thereafter by the Ontario Ministry of Natural Resources (Ducks Unlimited Canada 1995).

Matchedash Bay is an approximately 900 ha provincially significant wetland and, similar to Wye Marsh, is also considered an Important Bird Area (Wilson and Cheskey 2001). It is hydrologically connected to southeastern Georgian Bay, found approximately 20 km East of Wye Marsh (Figure 1.3). The Matchedash Bay watershed is also heavily impacted by agricultural practices and the perimeter of the marsh has been modified with some docks and cottages. The wetland is primarily underlain by limestone bedrock with some Precambrian Shield rock outlets and has two river inputs; the North River and Coldwater River (IBA Canada 2012).

Data collection and analysis

We collected and analyzed all chemical parameters according to methods described in Chow-Fraser (2006). We analyzed total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), total ammonia-nitrogen (TAN), total nitrate-nitrogen (TNN), pH, specific conductance (COND) and total suspended solids (TSS). Water samples for nutrients were collected near the middle of each wetland, in open water void of submerged aquatic vegetation (Figures 1.4 and 1.5).

To measure daily changes in water levels during the 2013 growing season, we installed a manual stage gauge in Wye Marsh and took readings each morning. Since Matchedash Bay is connected hydrologically to Georgian Bay, we used daily water levels measured at the Midland Canadian Hydrographic Service Station (Department of Fisheries and Oceans Canada), which is located approximately 16 km linear distance away. The lowest recorded water level was subtracted from all entries at each site so that we could compare the relative change in water levels through the season. All mean annual water levels for Georgian Bay were calculated from data provided by the Canadian Hydrographic Services.

To evaluate the effect of hydrological connection on water chemistry, we assembled published data from studies that documented the conditions of wetlands with varying degrees of hydrological connection to Georgian Bay (see Table 1.1). This dataset included mean values for 17 beaver-impounded marshes, 18 hydrologically connected marshes (Fracz and Chow-Fraser 2013) and 15 open-water samples collected along eastern Georgian Bay (deCatanzaro and Chow-Fraser 2011). These data were then compared to corresponding values measured in Wye Marsh and Matchedash Bay in this study.

To assess the historical aquatic habitat availability, we obtained aerial imagery from the National Air Photo Library of Canada (1930, 1931, 1965, 1973, 1976, 1987 and 1995) and the Ontario Ministry of Natural Resources (2002 and 2008). All images that were not already in digital format were scanned as JPEG files (600 by 600 dpi, greyscale) and georeferenced into ArcGIS version 10 (ESRI, Redlands, California, U.S.A.) for analysis. A boundary was digitized around the marsh border for each site and year according to the vegetation zones. The area within each border was then digitized and tabulated as either aquatic habitat or emergent vegetation. This allowed for the quantification of change in habitat through time and as it related to mean annual water levels in Georgian Bay. We completed all statistical analyses in JMP version 10 (SAS Institute Inc., Toronto, Ontario, Canada).

To enable valid comparisons, we wanted to use the same gear type to sample the fish communities in both wetlands. Even though fyke nets and boat electrofishing have been used successfully to survey the fish communities in wetlands of eastern Georgian Bay (see Cvetkovic et al. 2012), neither method could be used in Wye Marsh because an electrofishing boat could not enter the marsh, and all accessible areas were either too shallow or too deep for fyke nets. We therefore chose to use modified Windermere traps (Edwards et al. 1998), which could be set up on the soft, mucky sediment without use of poles. On two occasions in 2012 (May 8-10 and July 5-7) we surveyed the fish communities using traps that were paired 15 m apart, with one net having two conical openings and the other having only one. Net openings were placed parallel to the shoreline in areas that had low-density, patchy, submerged aquatic vegetation. These nets were set between 5 and 6 hours. When the nets were pulled, fish were identified to species, measured and released unharmed to the water where they were caught. We selected 5 sites in Wye Marsh, based on accessibility (see Figure 1.4), to conduct the fish surveys and 3 sites in Matchedash that had similar site characteristics (see Figure 1.5).

To determine how the fish communities in Wye Marsh and Matchedash Bay compared with those found in wetlands elsewhere throughout eastern Georgian Bay, we accessed data from Cvetkovic and Chow-Fraser (2011), which included fish information collected from 116 wetlands collected over a 10-year period (2001 to 2011; Table 1.2). Since these fish had been caught with a different gear type (24-h paired fyke nets), we calculated relative abundances and only compared the most common fish taxa.

For both Wye Marsh and Matchedash Bay, point counts of marsh birds were conducted twice at each of three sites during May and June in 2012. Each point count was 25 minutes in duration, including 10 minutes of passive listening, 10 minutes of active playbacks of secretive marsh species songs, followed by a final 5 minute passive listening period. During the 10-minute playback period, 30-second songs of 10 focal secretive marsh species were broadcast from hand-held speakers, with 30 seconds of silence between playbacks. Playbacks of these secretive marsh birds included the Least bittern, American bittern, Yellow rail, Pied-billed grebe, American coot, Common moorhen, Sora, Virginia rail, King rail and Black rail. Locations of point counts at Wye Marsh were chosen to represent different habitat types, and situated at least 380 m apart (Figure 1.4). We surveyed close to the dam to account for potentially different habitat that may arise due to the construction of the impoundment. The middle of the marsh was chosen to account for any species that may prefer open water surrounded by emergent vegetation and/or interior marsh habitat. The third station was at the boardwalk, near the edge of the marsh and close to the visitor centre, where we observed a structurally diverse habitat. Sites in Matchedash Bay were chosen to best reflect similar attributes of the habitat surveyed in Wye Marsh (Figure 1.5).

Mean scores of the Shannon diversity index were used to assess the overall diversity of the bird and fish communities in both wetlands. Using all of the data collected, we also calculated scores of the Sorensen similarity index for both birds and fish (McCune et al. 2002). For the bird communities, we calculated scores of the Index of Marsh Bird Community Integrity of each site (DeLuca et al. 2004). Unidentifiable bird species were excluded from all index score calculations (Table 1.3). All calculations and statistical analyses were carried out with JMP version 10 (SAS Institute Inc., Toronto, Ontario, Canada).

Results

Water-chemistry variables available for Wye Marsh and Matchedash Bay have been assembled and presented in Table 1.1. Phosphorus concentrations in Wye Marsh were uniformly higher than those in Matchedash Bay, with almost double the TP concentrations (63.0 vs 34.2 µg·L^{-1}, respectively) and 1.5 times higher SRP concentrations (19.4 vs 13.4 µg·L^{-1}). By contrast, inorganic forms of nitrogen were lower in Wye Marsh than in Matchedash Bay (0.04 vs 0.07 mg·L^{-1} for TAN and 0.09 vs 0.27 mg·L^{-1} for TNN, respectively) and no differences were found for TN (both 1.04 mg·L^{-1}). Water in Wye Marsh was more acidic (pH 6.38) than that in Matchedash Bay (pH of 8.0), and had higher specific conductance (307 vs 221 µS·cm^{-1}). The concentration of total suspended solids in Wye Marsh was much lower than that in Matchedash Bay (4.40 vs 14.83 mg·L^{-1}).

Hydrographs for both study sites were strikingly different through the 2013 growing season (Figure 1.6). Water levels in Wye Marsh (impounded wetland) were highest in late April and continued to decline throughout the study period, whereas those in Matchedash Bay (hydrologically connected with Georgian Bay) increased from late April to early August and then began to decline. The magnitude of change in water levels through the season in Wye Marsh (0.34 m) was about 35% lower than that for Matchedash Bay (0.53 m).

The amount of emergent and aquatic habitat (open water and submerged aquatic vegetation) in Wye Marsh (Figure 1.7) and Matchedash Bay (Figure 1.8) has fluctuated over the past eight decades. In this study, we were most interested in how availability of aquatic habitat has changed through time since this is an essential habitat class for both fish and bird species. Between 1930 and 2008, the amount of aquatic habitat in Wye Marsh changed from 153.4 ha in 1930 to 57.38 ha in 2008, a corresponding decrease in percentage marsh habitat from 32% to 12% (Table 1.4). There was a general decline in this habitat category through the 78 years (Figure 1.9A), and even though there was a notable resurgence in aquatic habitat in 1976 (to 31%), we found a statistically significant negative relationship with time (F ratio = 11.32, p = 0.020, r^2 = 0.69). It is important to note that the 1976 air photo was taken shortly after the St. Marie Dam had been rebuilt and therefore reflects a short duration when the marsh had been hydrologically reconnected with Georgian Bay. To remove the confounding effect of the dam failure, we excluded data prior to 1972 and ran a second the regression analysis for Wye Marsh. This time, we found a much stronger negative relationship with time (F ratio = 16.92, p = 0.026, r^2 = 0.85) and a steeper slope that is more representative of the long-term effect of the impoundment on the plant community. By comparison, the amount of aquatic habitat in Matchedash Bay fluctuated from 409.44 ha in 1931 to 330.21 ha in 2008, with the highest value in 1973, at a time when water levels were near record highs for Lake Huron (Table 1.4). Even though there was a corresponding drop in percentage wetland habitat from 50% to 37% between the 77 years, we did not find any statistically significant decline with time (F ratio = 0.13, p = 0.733, r^2 = 0.03; Figure 1.9A).

We hypothesized that the amount of aquatic habitat in coastal wetlands should vary as a function of water levels in Lake Huron, as long as they are hydrologically connected with Georgian Bay. We tested this hypothesis by regressing the amount of aquatic habitat against mean annual water levels of Lake Huron, and found a significant positive relationship between these variables in the hydrologically connected Matchedash Bay (F ratio = 12.20, p = 0.017, r^2 = 0.71; Figure 1.9B). By contrast, we found no statistically significant relationship between these variables for Wye Marsh when we used all data from 1930 to 2008 (F ratio = 2.2656, p = 0.1926, r^2 = 0.31) or when we only used data from 1976 to 2008 (F ratio = 3.7211, p = 0.1493, r^2 = 0.55; Figure 1.9B).

The fish communities in the two wetlands differed with respect to species richness (4 vs 5 for Wye Marsh and Matchedash Bay, respectively; Table 1.2). Of the species caught, only the yellow perch (*Perca flavescens*) and pumpkinseed (*Lepomis gibbosus*) were common. The shorthead redhorse (*Maxostoma macrolepidotum*) and brown bullhead (*Ameiurus nebulosus*) were only caught in Wye Marsh whereas the longear sunfish (*Lepomis megalotis),* largemouth bass (*Micropterus salmoides*) and rock bass (*Ambloplites rupestris)* were only found in Matchedash Bay. Each wetland had 2 species of migratory fish; Wye Marsh had the yellow perch and shorthead redhorse, whereas Matchedash had yellow perch and longear sunfish. Mean Shannon diversity score for Wye Marsh was significantly lower than that for Matchedash Bay (Wilcoxon rank sums Z = 2.22, p = 0.0262; Table 1.5). The lower score associated with Wye Marsh reflected the number of nets that were empty or had only one species present. The Sorensen similarity index was 0.44 (Table 1.5).

The two wetlands had similar bird community composition and diversity, especially with respect to presence of marsh-dependent species (Table 1.3). In total, 22 bird species were identified in Wye Marsh, including 4 marsh-dependent species (swamp sparrow; *Melospiza georgiana*, American bittern; *Botaurus lentiginosus*, marsh wren; *Cistothorus palustris*, and Virginia rail; *Rallus limicola*). By comparison, we found 17 species in Matchedash Bay, which also included 4 marsh dependent species. Three marsh-dependent species were found in both wetlands; unique species were the American bittern in Wye Marsh and the great blue heron (*Ardea herodias*) in Matchedash Bay. The mean Shannon diversity score was numerically higher for Wye Marsh (2.02 ± 0.19) compared with Matchedash Bay (1.97 ± 0.10) but the difference was not statistically significant (Wilcoxon rank sums Z = 0.44, p = 0.6625; Table 1.5). The Sorensen similarity index score was 0.72 (Table 1.5). The Index of Marsh Bird Community Integrity score for Wye Marsh (5.65 ± 0.83) was only slightly lower than that for Matchedash Bay (5.79 ± 0.40; Table 1.5), but again, we found no significant differences between wetlands (Wilcoxon rank sums Z = -0.00, p = 1.0000).

Discussion

It is helpful to interpret differences and similarities between Wye Marsh and Matchedash Bay in light of what we know about the limnology of Georgian Bay and associated coastal marshes. Previous studies have shown that water chemistry in offshore waters of Georgian Bay is generally alkaline and has a high specific conductance (see deCatanzaro and Chow-Fraser 2011) that reflect influences of the limestone bedrock from the Bruce Peninsula and the Niagara Escarpment to the South and West (Table 1; Weiler 1988). Open water also has high concentration of nitrates, but relatively low concentrations of total and soluble phosphorus and total suspended solids (Table 1.1). By contrast, water in the coastal marshes (deCatanzaro and Chow-Fraser 2011) and beaver impoundments (Fracz and Chow-Fraser 2013) has reduced alkalinity, lower ionic strength and lower concentration of nitrates. The coastal water also has higher concentrations of suspended solids and phosphorus that reflects more heavy influences from watershed runoff, particularly when a system is completely disconnected from Georgian Bay (i.e. beaver impoundments; Table 1.1).

The location of Wye Marsh and Matchedash Bay on primarily limestone bedrock distinguishes them from the other coastal wetlands and beaver impoundments that have been studied in eastern Georgian Bay (i.e. deCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013). The relatively high specific conductance in these marshes can be attributed to this difference in bedrock, and also to the agriculturally dominant land use in their watershed (Table 1.1). Associated with farming activities, there are higher concentrations of soluble and total phosphorus as well as higher nitrates in the water. Consistent with other studies, however, the impounded wetland had higher concentrations of phosphorus (63.0 and 19.4 $\mu g \bullet L^{-1}$ of TP and SRP, respectively) compared to the hydrologically connected marsh (34.2 and 13.4 $\mu g \bullet L^{-1}$, respectively), and this may due to differences in connectivity. These results differ from nutrients in Lake Erie, where higher TP was found in hydrologically connected wetlands but are consistent with SRP concentrations being higher in diked wetlands (Mitsch 1992). Ammonia concentrations in both wetlands were much higher than the 0.008 $mg \bullet L^{-1}$ measured in open waters of Georgian Bay, but the concentration of 0.07 $mg \bullet L^{-1}$ in Matchedash Bay is higher than that in the beaver-impounded wetlands (0.03 $mg \bullet L^{-1}$) and Wye Marsh (0.04 $mg \bullet L^{-1}$). This is inconsistent with predicted effects of hydrological disconnection.

The lower pH in Wye Marsh (6.38) relative to Matchedash Bay (8.00) may also be attributed to the effect of hydrological severance between Wye Marsh and Georgian Bay, similar to the lower pH in a beaver-impounded wetland when compared to hydrologically connected coastal wetlands in eastern Georgian Bay (5.57 vs 6.95; Table 1.1). The fact that Matchedash has a more alkaline pH compared to the coastal wetlands sampled by deCatanzaro and Chow-Fraser (2011) is because the former receives H^+ and phosphorus-enriched fertilizer runoff from agricultural land use and also drains limestone bedrock, whereas the coastal wetlands of eastern Georgian Bay receive primarily dystrophic runoff from the Precambrian Shield.

The open waters of Georgian Bay are naturally low in total suspended solids (0.8 $mg•L^{-1}$ in Table 1.1). At the land-water interface, however, human activities, carp bioturbation and watershed runoff can all contribute to higher levels of suspended solids (Chow-Fraser et al. 1998). In eastern Georgian Bay, TSS values measured within beaver-impounded wetlands (15.5 $mg•L^{-1}$) were consistently higher than those in hydrologically connected coastal marshes (2.1 $mg•L^{-1}$), and likely reflected the lack of mixing with the dilute water of Georgian Bay. When we compare the situation between Wye Marsh and Matchedash Bay, however, differences in TSS concentrations could not be explained by the effect of impoundment. In fact, results were opposite to what we had expected with three times lower TSS values within Wye Marsh compared with Matchedash Bay (4.4 vs 14.83 $mg•L^{-1}$). We attribute the higher turbidity in Matchedash Bay to boat traffic (which is not allowed in Wye Marsh) and to a very large population of common carp that can keep sediment suspended by their spawning and feeding activities (Pers. obs.; Mitsch 1992; Lougheed et al. 1998; Chow-Fraser 2005).

Along with a shift in water chemistry, we also found very different trends in water-level regimes during the growing season (Figure 1.6). Water levels of Georgian

Bay at Midland, and therefore that of Matchedash Bay, peaked in late summer (early August), which is consistent with the general pattern exhibited by Lake Huron (Figure 1.2). This reflected a slow recharge through the summer when snow across the large Georgian Bay watershed melts and gradually makes its way into the bay. Wye Marsh, on the other hand, demonstrated a steady drop in water level from May to August. We suggest that Wye Marsh rapidly filled up in early May from snowmelt in the relatively small watershed, and gradually drained during the summer months. We speculated that differences in the hydrological regimes of these two wetlands were due mainly to the loss of connection between Georgian Bay and the coastal marsh, which could have led to drastic physical alterations of aquatic habitat and impact the biotic communities (Keddy and Reznicek 1986).

Throughout the period of study, the proportion of aquatic habitat in Wye Marsh (12-32%) was always lower than that in Matchedash Bay (37-63%). Whereas aquatic habitat in Matchedash Bay varied as a function of the water level in Georgian Bay, that in Wye Marsh was only significantly related to years following impoundment. Though there are inconsistencies in the literature with respect to the extent of vegetation expansion in diked wetlands, studies conducted in the lower Great Lakes agree that mean vegetative coverage of diked wetlands is greater than that of adjacent undiked wetlands (Mitsch 1992; Sherman et al. 1996; Johnson et al. 1997; Monfils et al. 2014). We suggest that the stabilized water level in Wye Marsh over these many decades has allowed the emergent vegetation community of mainly cattails to expand and form dense persistent floating mats irrespective of water depth, a situation that runs counter to past studies in which emergent vegetation is governed by changes in water level (Keddy and Reznicek 1986; Lyon et al. 1986; Chow-Fraser 2005; Wei and Chow-Fraser 2008). These persistent mats can become uprooted from the sediment and float, providing structure for further growth while being tolerant of water level fluctuations or deeper reaches of the marsh (Galloway et al. 2006). In 1995, a prescribed drawdown of Wye Marsh was attempted to curtail the growth of cattails. One of the reasons for the failure to successfully create a structurally diverse habitat was that these floating mats were able to withstand the drawdown (Ducks Unlimited Canada 1995). Despite the obvious tenacity of cattails, when the dam broke during the 1970s, much of the sediment and cattail community was flushed out of Wye Marsh and a larger proportion of the marsh in 1976 was identified as aquatic habitat (Figure 1.7). This caused the system to be reset, but when the dam was restored, the cattails began to expand as aggressively as before, so that by the late 1980s, floating mats of cattails were once again the dominant feature. By comparison, the proportion of aquatic habitat in Matchedash Bay did not vary significantly with time, and only with water level of Georgian Bay.

We predicted that the dam in Wye Marsh would restrict the movement of migratory fish species and lead to lower diversity index scores compared with Matchedash Bay. Although we could not discriminate our two sites on the basis of the number of migratory fish species (Table 1.2), we did find a lower diversity associated with Wye Marsh compared with Matchedash Bay. This is consistent with studies conducted elsewhere in the Great Lakes (Brazner 1997; Johnson et al. 1997; Bouvier et al. 2009) where reduced hydrological connectivity with the lake appeared to affect the distribution of fish species in coastal marshes and led to reduced Sorensen similarity scores between isolated sites (Table 1.5). Keast and Fox (1990) also found a similar reduction in species richness of fish in beaver dammed wetlands elsewhere in Ontario, Canada. As wetlands became more and more isolated from the main stream, species richness became further reduced. The two species that are considered migratory (yellow perch and shorthead redhorse; Jude and Pappas 1992; Wei et al. 2004) in Wye Marsh, could be a remnant population that survived the impoundment because they were uniformly small compared to those in Matchedash Bay and appeared to be stunted, possibly because of competition for food and the absence of a large predator (Johnson et al. 1997; Markham et al. 1997).

All fish species found were among the most commonly caught species along the eastern shores of Georgian Bay from 2001 to 2011. The two species that are common to both Wye Marsh and Matchedash Bay (pumpkinseed and yellow perch) are known to have moderate niche breadths and are tolerant of disturbance in other coastal marshes of the Great Lakes (Seilheimer and Chow-Fraser 2007). Absent from Wye Marsh were large piscivores such as largemouth bass. Bouvier et al. (2009) and Keast and Fox (1990) also found that impounded wetlands tended to lack large piscivores, and past investigators have attributed this to differences in environmental conditions or to hydrological connectivity (Snodgrass et al. 1996; Taylor 1997; Bouvier et al. 2009). For Wye Marsh, we suggest that both factors may have played a role in the long term but regardless of the mechanism, we attribute the lower fish biodiversity compared with Matchedash Bay to the presence of the St. Marie Dam.

As hypothesized, the bird communities sampled at both sites were very similar. Of the 25 species identified, 14 were common between sites including the marsh dependent Virginia rail, marsh wren and swamp sparrow. We did not find any significant differences between mean Shannon's *H* or IMBCI scores of these wetlands. Accordingly, the Sorensen similarity score between sites (0.72) was relatively high, indicating a large overlap in marsh birds. These scores also provide information with respect to the quality of the marsh habitat, which are both similarly high.

Our results are consistent with those reported in Monfils et al. (2014) who showed that 67% of the bird species were common between diked and undiked wetlands of Lake Michigan. Galloway et al. (2006) also found that paired wetlands along the shores of the lower Great Lakes showed few differences in bird communities. In contrast, Nummi (1992) found greater waterfowl use after 3 years of wetland impoundment by beaver damming, but did not look into the long-term impacts. Our results emphasize the findings of Kadlec (1962) and Harris and Marshall (1963) who showed only short-term gains from diking in situations where there was no active management of water levels.

Management implications

Impoundment structures such as dams and dikes are generally permanent, and are designed to manipulate/maintain water levels over the long term. In most instances, they have been implemented to maintain or improve plant and bird communities (Galloway et al. 2006). It is widely discussed, however, that active water-level management is required to maintain diverse plant and bird communities and that impoundment as a management strategy does not adequately consider aquatic species. In Georgian Bay, one of the most biologically diverse stretches of the Great Lakes, we observed overall negative implications of diking wetlands over the long term, consistent with past studies. We saw infilling of the diked wetland through time, where we did not in the hydrologically connected wetland. In turn, we also saw patterns of altered water chemistry due to a loss of hydrological connection similar to what has been observed by deCatanzaro and Chow-Fraser (2011) and Fracz and Chow-Fraser (2013). Impacts to the fish community may also be attributed to changes in water chemistry, structure of the plant communities, and a barrier to migration of top predators. Our data also support past literature that bird and plant communities gain no apparent benefit from long-term impoundment without active water-level manipulation. Coastal wetlands along the eastern shoreline of Georgian Bay are relatively inaccessible and this inaccessibility would make active management costly and difficult to undertake. Even if they were accessible, these coastal wetlands are unlikely to be managed beyond their initial installation. Given these concerns, we do not believe that diking and damming are suitable strategies to combat the loss of wetland habitat due to declining water levels in Georgian Bay.

Acknowledgements

First, we thank Sierra Club Canada Foundation for providing partial funding for the project, and Friends of Wye Marsh for providing logistical and in-kind support as well as encouragement. This project was initiated after connecting with Friends of Wye Marsh and would not have continued without the assistance of the Wye Marsh Wildlife Centre. We appreciate the invaluable help from Sara Street and Paul Beachamp for providing information about Wye Marsh, and advice and assistance on how to sample within the marsh. We thank the Ontario Ministry of Natural Resources who provided the fishing gear. Dr. James Michael Waddington provided helpful advice on water-level recording and analyses. Last but not least, we wish to thank all those who assisted in the field and lab, including Chris Biberhofer, Amanda Fracz, Chantel Markle, Lucy Meng, Mary Muter, Julia Rutledge, and Caitlin Sommers.

Literature Cited

Barber MJ, Childers DL, Babbitt KJ, Anderson DH (2002) Controls on Fish Distribution and Abundance in Temporary Wetlands. Can J Fish Aquat Sci 59:1441-1450.

Bouvier LD, Cottenie K, Doka SE (2009) Aquatic Connectivity and Fish Metacommunities in Wetlands of the Lower Great Lakes. Can J Fish Aquat Sci 66 (6):933-948.

Brazner JC (1997) Regional, Habitat, and Human Development Influences on Coastal Wetlands and Beach Fish Assemblages in Green Bay, Lake Michigan. J Great Lakes Res 23 (1):36-51.

Bufo Inc. (1978) Natural History of Wye Marsh. vol 2. Canadian Wildlife Service.

Chow-Fraser P (2005) Ecosystem Response to Changes in Water Level of Lake Ontario Marshes: Lessons from the Restoration of Cootes Paradise Marsh. Hydrobiologia 539:189-204.

Chow-Fraser P (ed) (2006) Development of the Water Quality Index (WQI) to Assess Effects of Basin-Wide Land-Use Alteration on Coastal Marshes of the Laurentian Great Lakes. Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Indiana Biological Survey, Bloomington, IN.

Chow-Fraser P, Lougheed V, Le Thiec V, Crosbie B, Simser L, Lord J (1998) Long-Term Response of the Biotic Community to Fluctuating Water Levels and Changes in Water Quality in Cootes Paradise Marsh, a Degraded Coastal Wetland of Lake Ontario. Wetl Ecol Manag 6:19-42.

Cvetkovic M, Chow-Fraser P (2011) Use of Ecological Indicators to Assess the Quality of Great Lakes Coastal Wetlands. Ecol Indic 11 (6):1609-1622.

Cvetkovic M, Kostuk K, Chow-Fraser P (2012) Gear-Type Influences on Fish Catch and a Wetland Fish Index in Georgian Bay Wetlands. N Am J Fish Manag 32:313-324.

deCatanzaro R, Chow-Fraser P (2011) Effects of Landscape Variables and Season on Reference Water Chemistry of Coastal Marshes in Eastern Georgian Bay. Can J Fish Aquat Sci 68 (6):1009-1023.

DeLuca WV, Studds CE, Rockwood LL, Marra PP (2004) Influence of Land Use on the Intregrity of Marsh Bird Communities of Chesapeake Bay, USA. Wetlands 24 (4):837-847.

Ducks Unlimited Canada (1995) Wye Marsh: An Historical Perspective, Review of Habitat Management and Recommendations for Future Action.

Edwards AJ, Czypinski GD, Selgeby JH (1998) A Collapsible Trap for Capturing Ruffe. N Am J Fish Manag 18:465-469.

Fracz A, Chow-Fraser P (2012) Impacts of Declining Water Levels on the Quantity of Fish Habitat in Coastal Wetlands of Eastern Georgian Bay, Lake Huron. Hydrobiologia 702 (1):151-169.

Fracz A, Chow-Fraser P (2013) Changes in Water Chemistry Associated with Beaver-Impounded Coastal Marshes of Eastern Georgian Bay. Can J Fish Aquat Sci 70 (6):834-840.

Galloway M, Bouvier L, Meyer S, Ingram J, Doka S, Grabas G, Holmes K, Mandrak N (2006) Chapter 8. Evaluation of Current Wetland Dyking Effects on Coastal Wetlands and Biota. In: Mortsch LD, Ingram J, Hebb A, Doka SE (eds) Great Lakes Coastal Wetland Communities: Vulnerability to Climate Change and Response to Adaptation Strategies. Environment Canada and the Department of Fisheries and Oceans, pp 187-229.

Gottgens JF, Swartz BP, Kroll RW (1998) Long-Term GIS-Based Records of Habitat Changes in a Lake Erie Coastal Marsh. Wetl Ecol Manag 6:5-17.

Harris SW, Marshall WH (1963) Ecology of Water-Level Manipulations on a Northern Marsh. Ecology 44 (2):331-343.

IBA Canada (2012) IBA Canada Important Bird Areas; Matchedash Bay. Accessed 10/10/2013 2013

Johnson DL, Lynch J, W.E., Morrison TW (1997) Fish Communities in a Diked Lake Erie Wetland and an Adjacent Undiked Area. Wetlands 17 (1):43-54.

Jude DJ, Pappas J (1992) Fish Utilization of Great-Lakes Coastal Wetlands. J Great Lakes Res 18 (4):651-672.

Kadlec JA (1962) Effects of a Drawdown on a Waterfowl Impoundment. Ecology 43 (2):267-281.

Keast A, Fox MG (1990) Fish Community Structure, Spatial Distribution and Feeding Ecology in a Beaver Pond. Environ Biol Fishes 27:201-214.

Keddy PA, Reznicek AA (1986) Great Lakes Vegetation Dynamics: The Role of Fluctuationg Water Levels and Buried Seeds. J Great Lakes Res 12 (1):25-36.

Lougheed VL, Chow-Fraser P (2002) Development and Use of a Zooplankton Index of Wetlands Quality in the Laurentian Great Lakes Basin. Ecol Appl 12 (2):474-486.

Lougheed VL, Crosbie B, Chow-Fraser P (1998) Predictions on the Effect of Common Carp (Cyprinus Carpio) Exclusion on Water Quality, Zooplankton, and Submergent Macrophytes in a Great Lakes Wetland. Can J Fish Aquat Sci 55 (5):1189-1197.

Lyon JG, Drobney RD, Olson CE (1986) Effects of Lake-Michigan Water Levels on Wetland Soil Chemistry and Distribution of Plants in the Straits of Mackinac. J Great Lakes Res 12 (3):175-183.

Maccrimmon HR (1980) Nutrient and Sediment Retention in a Temperate Marsh Ecosystem. Int Rev Ges Hydrobio 65 (5):719-744.

Markham CA, Lynch J, W.E., Johnson DL, Petering RW (1997) Comparison of White Crappie Populations in Diked and Undiked Lake Erie Wetlands. Ohio J Sci 97 (4):72-77.

McCune B, Grace JB, Urban DL (2002) Analysis of Ecological Communities. MjM software, Gleneden Beach, Oregon, USA.

McLaughlin DB, Harris HJ (1990) Aquatic Insect Emergence in Two Great Lakes Marshes. Wetl Ecol Manag 1 (2):111-121.

Mitsch WJ (1992) Combining Ecosystem and Landscape Approaches to Great-Lakes Wetlands. J Great Lakes Res 18 (4):552-570.

Monfils MJ, Brown PW, Hayes DB, Soulliere GJ, Kafcas EN (2014) Breeding Bird Use and Wetlands Characteristics of Diked and Undiked Coastal Marshes in Michigan. J Wildl Manag 78 (1):79-92.

Nummi P (1992) The Importance of Beaver Ponds to Waterfowl Broods - an Experiment and Natural Tests. Ann Zool Fenn 29 (1):47-55.

Quinlan C, Mulamoottil G (1987) The Effects of Water Level Fluctuations on Three Lake Ontario Shoreline Marshes. Can Wat Resour J 12 (1):64-77.

Seilheimer TS, Chow-Fraser P (2007) Application of the Wetland Fish Index to Northern Great Lakes Marshes with Emphasis on Georgian Bay Coastal Wetlands. J Great Lakes Res 33 (3):154-171.

Sherman DE, Kroll RW, Engle TL (1996) Flora of a Diked and an Undiked Southwestern Lake Erie Wetland. Ohio J Sci 96 (1):4-8.

Snodgrass J, W., Bryan AL, Jr., Lide R, F., Smith G, M. (1996) Factors Affecting the Occurrence and Structure of Fish Assemblages in Isolated Wetlands of the Upper Coastal Plan, U.S.A. Can J Fish Aquat Sci 53 (443-454).

Taylor CM (1997) Fish Species Richness and Incidence Patterns in Isolated and Connected Stream Pools: Effects of Pool Volume and Spatial Position. Oecol 110:560-566.

Wei A, Albert D, Chow-Fraser P (2004) Influence of Shoreline Features on Fish Distribution in the Laurentian Great Lakes. Can J Fish Aquat Sci 61 (7):1113-1123.

Wei A, Chow-Fraser P (2008) Testing the Transferability of a Marsh-Inundation Model across Two Landscapes. Hydrobiologia 600:41-47.

Weiler RR (1988) Chemical Limnology of Georgian Bay and the North Channel between 1974 and 1980. Hydrobiologia 163 (1):77-83.

Wilcox DA (2004) Implications of Hydrologic Variability on Succession of Plants in Great Lakes Wetlands. Aquat Ecosyst Health and Manag 7 (2):223-231.

Wilcox DA, Meeker JE (1992) Disturbance Effects on Aquatic Vegetation in Regulated and Unregulated Lakes in Northern Minnesota. Can J Bot 69:1542-1551.

Wilcox DA, Nichols SJ (2008) The Effects of Water-Level Fluctuations on Vegetation in a Lake Huron Wetland. Wetlands 28 (2):487-501.

Wilson WG, Cheskey ED (2001) Matchedash Bay Important Bird Area Conservation Action Plan.

Table 1.1 Comparison of water chemistry data for Wye Marsh, Matchedash Bay, and mean values for 17 beaver-impounded wetlands (Fracz and Chow-Fraser 2013), 18 hydrologically connected coastal wetlands (Fracz and Chow-Fraser 2013) and 11 open water sites (deCatanzaro and Chow-Fraser 2011) in Georgian Bay. Values in brackets are the range. N/A = data not available. * indicates published data collected in Wye Marsh in 1998 (Lougheed and Chow-Fraser 2002) and unpublished data collected in Matchedash Bay in 2010.

Parameter	Wye Marsh	Matchedash Bay	Beaver-impounded wetland	Hydrologically connected coastal wetland	Open Georgian Bay water
Impounded	**Yes**	**No**	**Yes**	**No**	**No**
Agricultural land-use in watershed	**Yes**	**Yes**	**No**	**No**	**No**
Total P ($\mu g \cdot L^{-1}$)	**63.0**	**34.2**	**30.2** (4.5-55.6)	**15.3** (4.7-29.7)	**5.5** (4.0-7.8)
Soluble Reactive P ($\mu g \cdot L^{-1}$)	**19.4**	**13.4**	**13.3** (6.2-30.1)	**3.7** (0.6-10.8)	**0.6** (0.5-1.0)
Total Nitrogen ($mg \cdot L^{-1}$)	**1.04**	**1.04**	N/A	N/A	N/A
Total Ammonia N ($mg \cdot L^{-1}$)	**0.04**	**0.07**	**0.03** (0.00-0.23)	**0.02** (0.00-0.07)	**0.008** (0.004-0.012)
Total Nitrate N ($mg \cdot L^{-1}$)	**0.09**	**0.27**	**0.03** (0.005-0.09)	**0.04** (0.01-0.10)	**0.235** (0.19-0.26)
pH	**6.38**	**8.00**	**5.57** (4.76-7.52)	**6.95** (6.19-8.97)	**8.1** (8.0-8.2)
Specific Conductance ($\mu S \cdot cm^{-1}$)	**307**	**221**	**47** (14-131)	**134** (54-207)	**180** (159-196)
Total Suspended Solids ($mg \cdot L^{-1}$)	**4.40*** (N/A)	**14.83*** (N/A)	**15.5** (2.01-32.75)	**2.1** (0.25-7.0)	**0.8** (0.6-1.3)

Table 1.2 Comparison of fish catch per unit effort (CPUE) in Wye Marsh and Matchedash Bay during 2012. CPUE were calculated from fish surveyed at 5 sites in Wye Marsh and 3 sites in Matchedash with Windermere traps set for 6 hours. % catch in Georgian Bay wetlands indicates the average proportion of each species caught using fyke nets in each wetland from 116 wetlands in Georgian Bay between 2001 and 2011 (Cvetkovic and Chow-Fraser 2011). Species are ordered base on residence status in wetlands based on Jude and Papas (1992), with wetland dependency decreasing as you move down the list.

Common name	Scientific name	Catch per unit effort		% catch in Georgian Bay wetlands 2001-2011
		Wye Marsh	Matchedash Bay	
Bluntnose minnow	*Pimephales notatus*	0.00	0.00	7.18
Brown Bullhead	*Ameiurus nebulosus*	0.40	0.00	8.33
Largemouth Bass	*Micropterus salmoides*	0.00	0.67	5.91
Golden Shiner	*Notemigonus crysoleucas*	0.00	0.00	1.21
Pumpkinseed	*Lepomis gibbosus*	2.00	27.33	38.94
Rock Bass	*Ambloplites rupestris*	0.00	0.33	5.61
Longear Sunfish	*Lepomis megalotis*	0.00	28.33	1.29
Yellow Perch	*Perca flavescens*	0.60	25.33	5.47
Shorthead Redhorse	*Maxostoma macrolepidotum*	1.20	0.00	0.17

Table 1.3 Comparison of marsh bird abundances in Wye Marsh and Matchedash Bay during 2012. Data are sum of all birds surveyed in three 25-minute point counts in each wetland during May and June. Species are ordered based on their IMBCI scores with scores decreasing as you move down the list (DeLuca et al. 2004).

Common name	Scientific name	Abundance	
		Wye Marsh	Matchedash Bay
Virginia Rail	*Rallus limicola*	1	1
Marsh Wren	*Cistothorus palustris*	15	27
American Bittern	*Botaurus lentiginosus*	1	0
Great-Blue Heron	*Ardea herodias*	0	1
Swamp Sparrow	*Melospiza georgiana*	16	15
Trumpeter Swan	*Cygnus buccinator*	5	3
Common Tern	*Sterna hirundo*	2	0
Common Yellowthroat	*Geothlypis trichas*	3	3
Osprey	*Pandion haliaetus*	2	0
Red-Winged Blackbird	*Agelaius phoeniceus*	24	23
Tree Swallow	*Tachycineta bicolor*	18	0
Yellow Warbler	*Setophaga petechia*	4	4
Willow Fly-Catcher	*Empidonax traillii*	2	2
Caspian Tern	*Hydroprogne caspia*	5	8
Belted King Fisher	*Megaceryle alcyon*	0	1
Barn Swallow	*Hirundo rustica*	3	5
Song Sparrow	*Melospiza melodia*	6	4
Canada Goose	*Branta canadensis*	6	1
Wood Duck	*Aix sponsa*	4	0
Mallard	*Anas platyrhynchos*	1	2
Double Crested Cormorant	*Phalacrocorax auritus*	1	0
Bufflehead	*Bucephala albeola*	1	0
Mourning Dove	*Zenaida macroura*	1	0
American Crow	*Corvus brachyrhynchos*	1	2
American Robin	*Turdus migratorius*	0	1
Duck *spp*.	N/A	4	0

Table 1.4 Changes, in hectares, of aquatic habitat and emergent vegetation in Wye Marsh and Matchedash Bay over 8 decades. All areas were calculated from digitized aerial images of the wetlands. Mean annual Georgian Bay water levels were obtained from the Canadian Hydrographic Service.

	Year	Aquatic Habitat (ha)	Dense Emergent Vegetation (ha)	Total area (ha)	% Aquatic Habitat	Mean Annual Georgian Bay Water Level (m a.s.l.)
Wye Marsh	1930	153.44	332.97	486.41	31.55%	176.65
	1965	95.58	372.48	468.06	20.42%	175.92
	1976	151.11	336.56	487.66	30.99%	176.90
	1987	86.83	392.51	479.35	18.11%	176.97
	1995	66.25	425.49	491.74	13.47%	176.53
	2002	53.20	434.61	487.81	10.91%	176.12
	2008	57.38	430.44	487.83	11.76%	176.01
Matchedash Bay	1931	409.44	411.21	820.65	49.89%	176.12
	1965	356.58	532.75	889.34	40.10%	175.92
	1973	562.18	330.46	892.63	62.98%	177.12
	1987	426.81	457.06	883.87	48.29%	176.97
	1995	483.56	413.01	896.57	53.93%	176.53
	2002	375.02	534.41	909.44	41.24%	176.12
	2008	330.21	557.83	888.04	37.18%	176.01

Table 1.5 Comparison of diversity index scores for the avian and fish communities of Wye Marsh and Matchedash Bay. *p*-values correspond to the probability that the index scores are significantly different between sites (α = 0.05). Bolded *p*-values indicate significantly different means between sites. IMBCI=Index of Marsh Bird Community Integrity (Deluca et al. 2004).

Community	Diversity Index	Mean values		*p*-value
		Wye Marsh	Matchedash Bay	
Marsh birds	Shannon's H'	2.02 ± 0.19	1.97 ± 0.10	0.6625
Marsh birds	E (IMBCI)	5.65 ± 0.83	5.79 ± 0.40	1.0000
Wetland fish	Shannon's H'	0.16 ± 0.15	1.02 ± 0.11	**0.0262**
Marsh Birds	Sorensen similarity	0.72		N/A
Wetland fish	Sorensen similarity	0.44		N/A

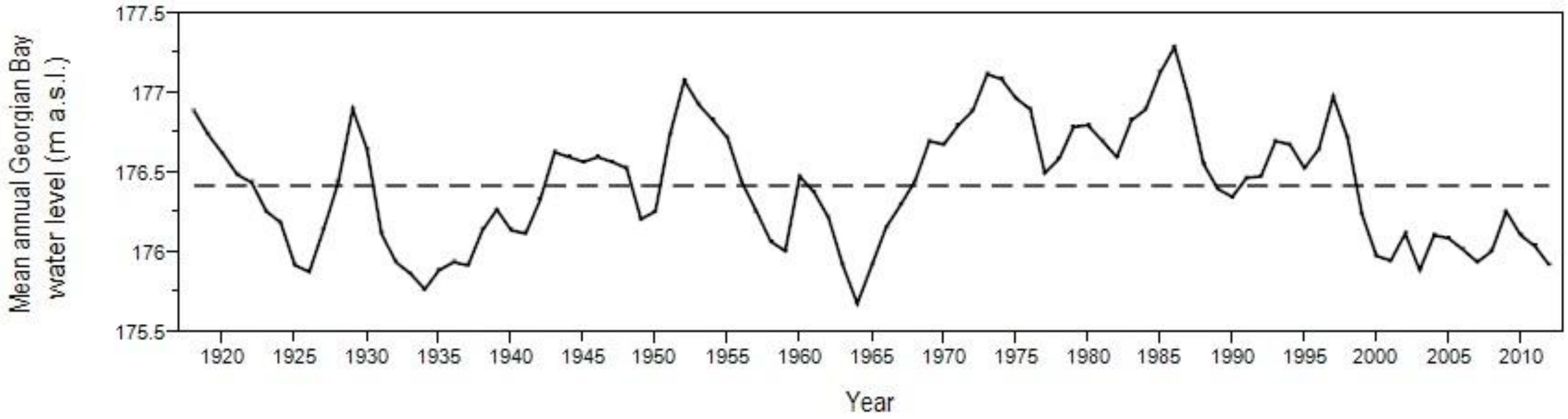

Figure 1.1 Mean annual Georgian Bay water levels through time (1918-2012). Dashed line indicates the long-term mean.

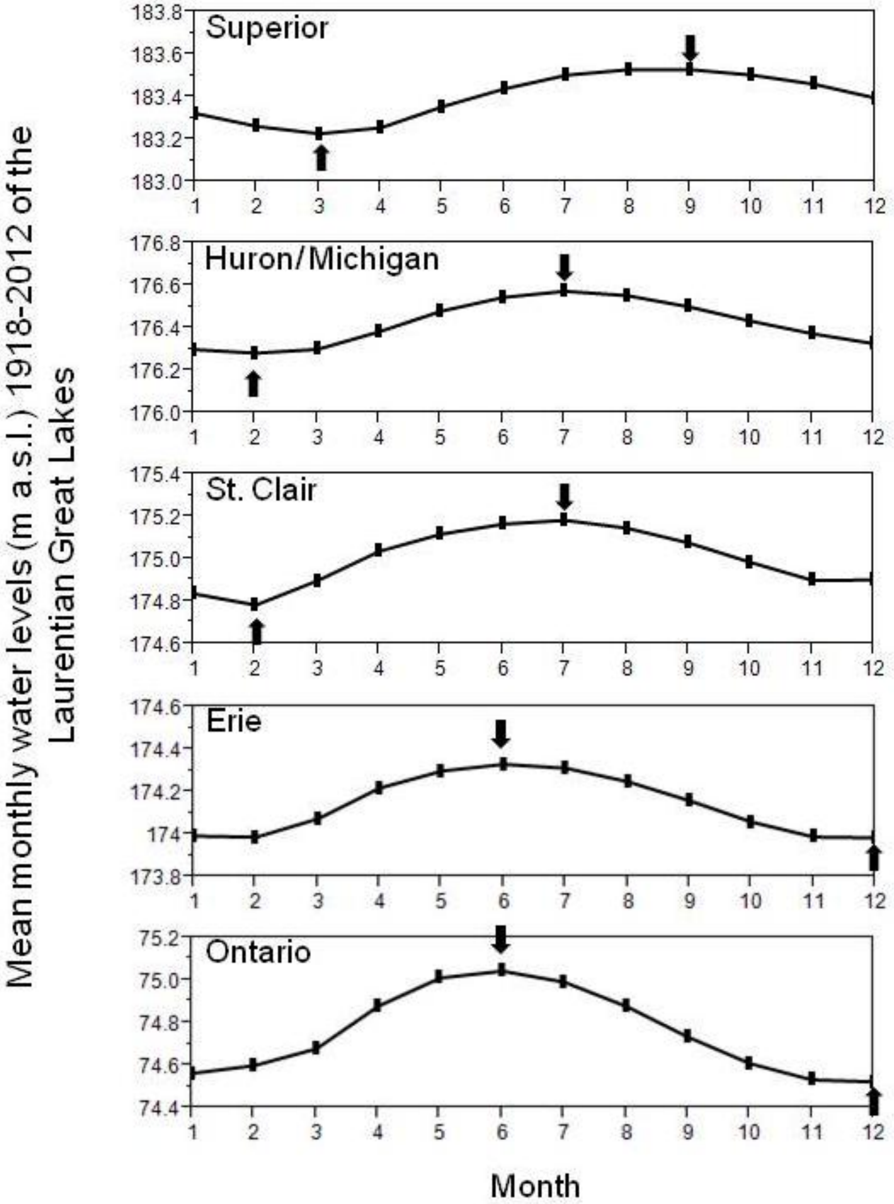

Figure 1.2 Hydrograph of mean monthly water levels in the Laurentian Great Lakes. Arrows indicate water level maxima and minima for each lake. Where available, data cover the period from 1918 to 2012. Obtained from the Canadian Hydrographic Service.

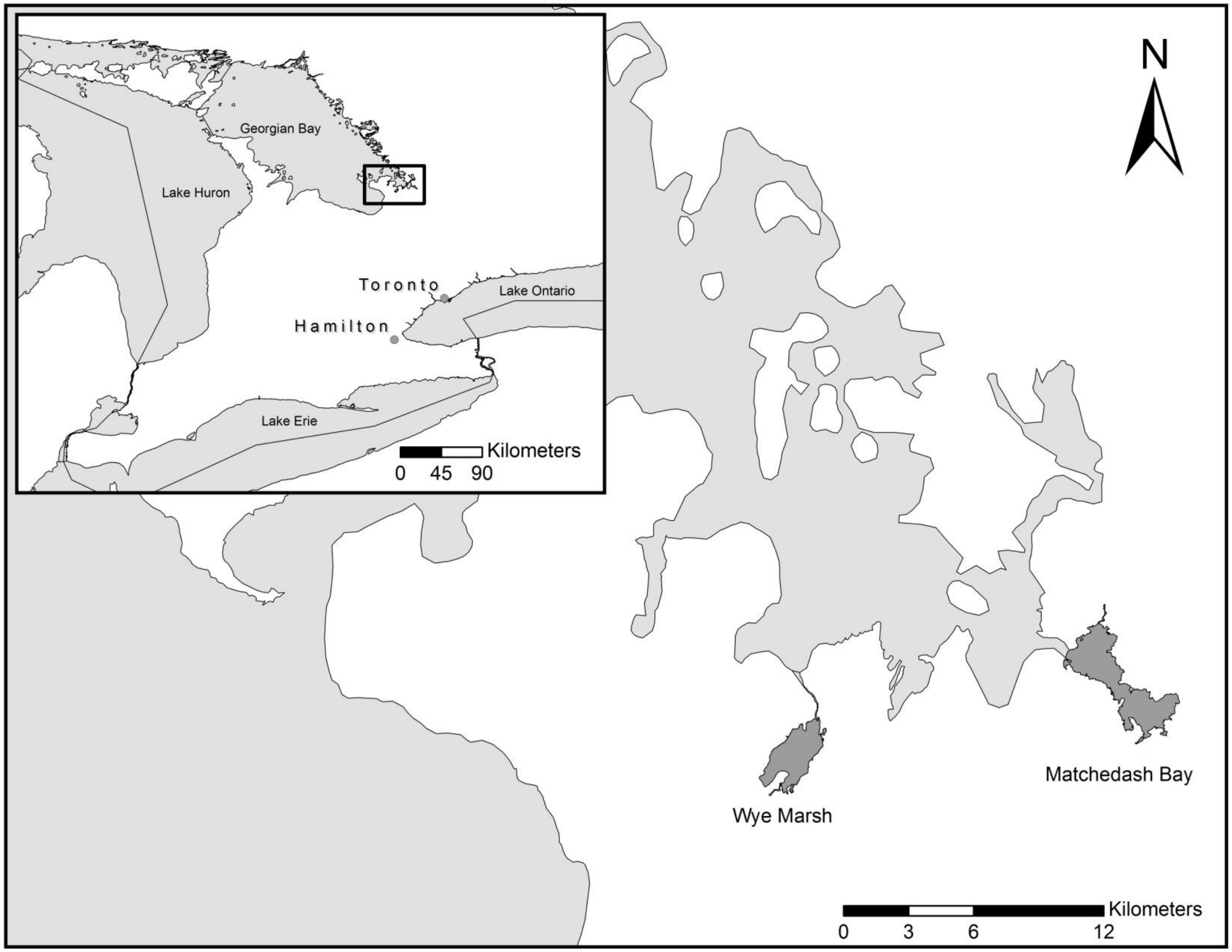

Figure 1.3 Location of Wye Marsh and Matchedash Bay in Southeastern Georgian Bay, Ontario.

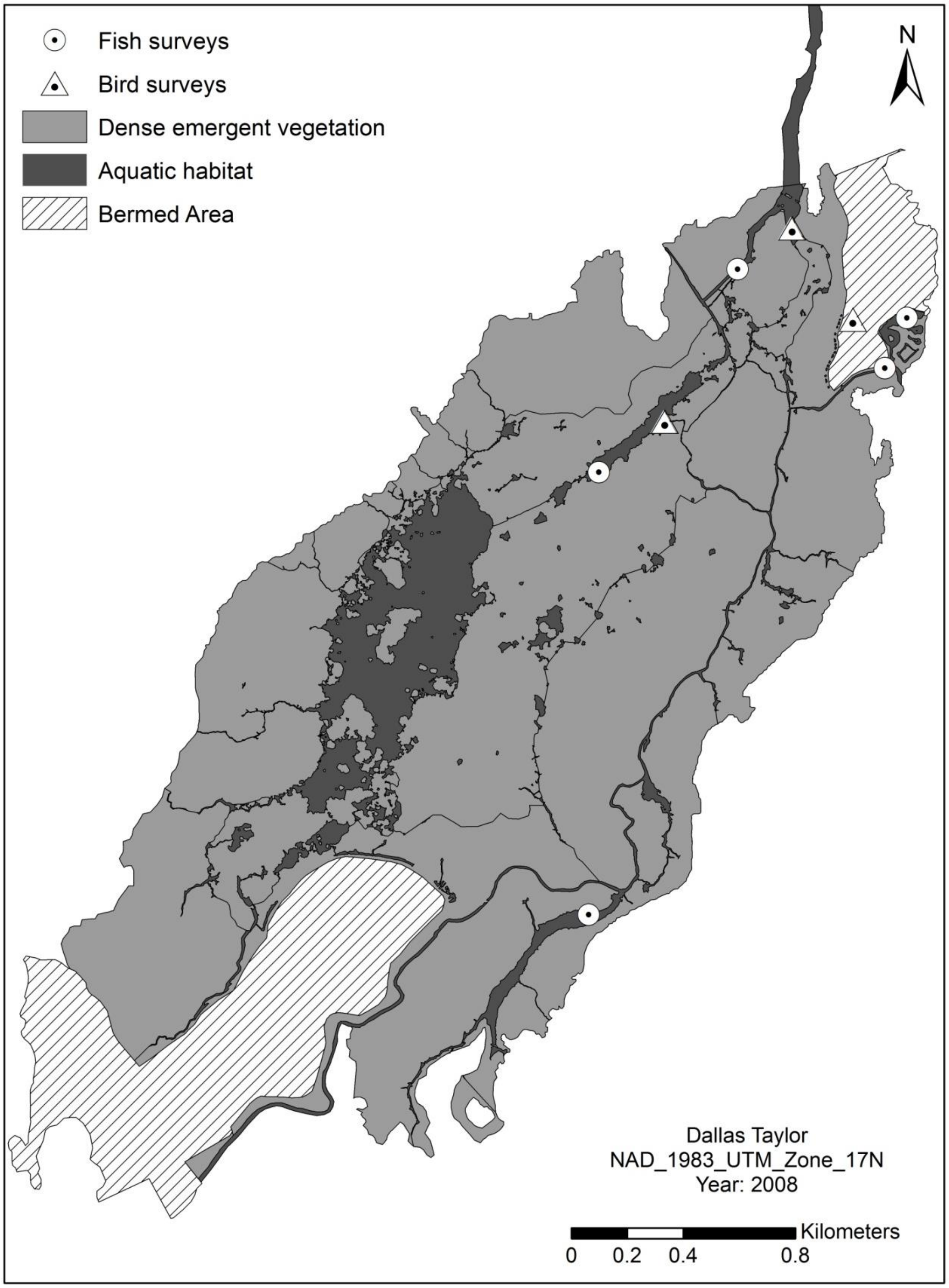

Figure 1.4 Map of Wye Marsh digitized from an aerial photo taken in 2008. Locations where fish and bird surveys were conducted in 2012, the bermed area (created in early 1990s), areas with dense stands of emergent vegetation and aquatic habitat are indicated.

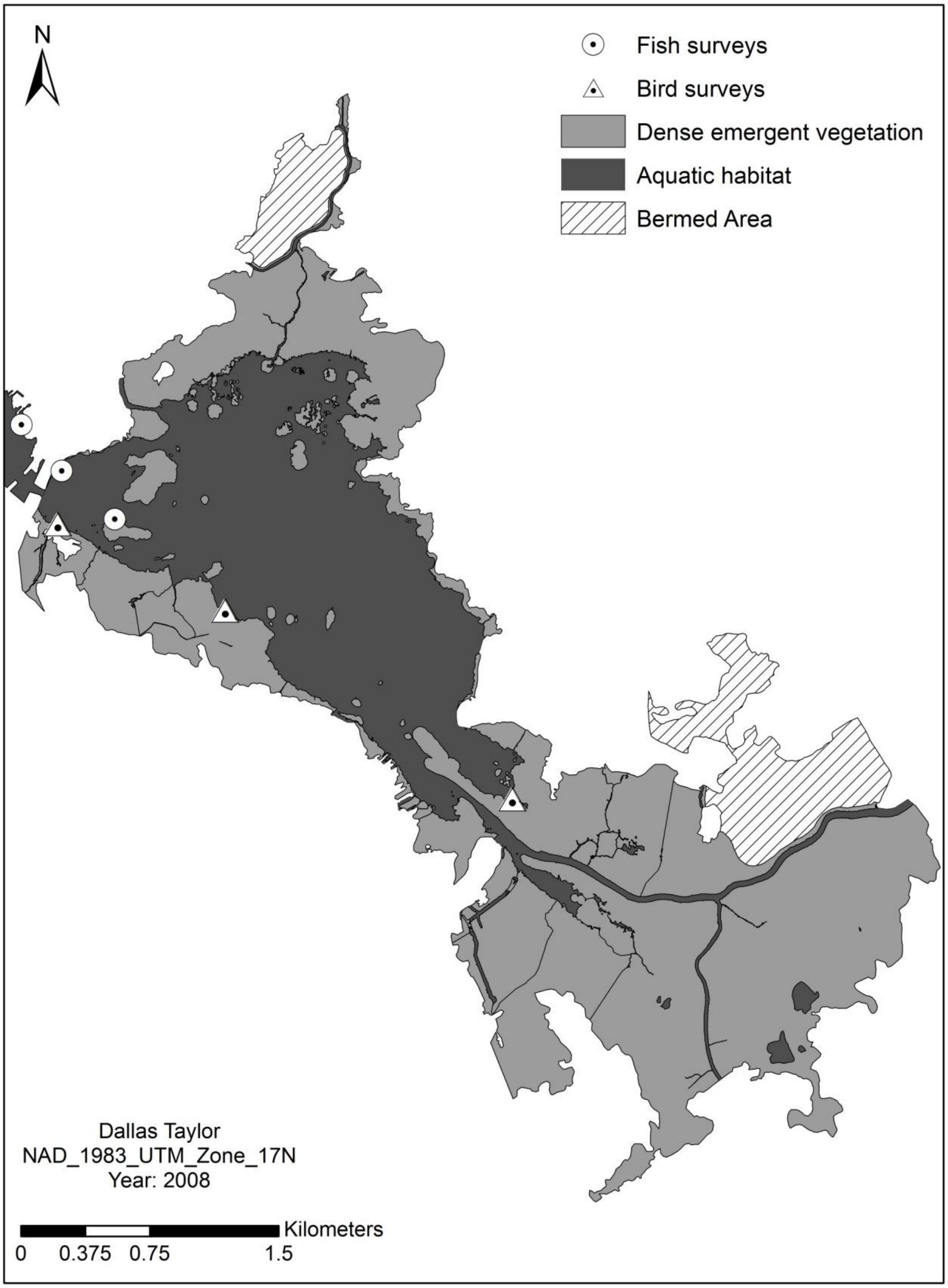

Figure 1.5 Map of Matchedash Bay digitized from an aerial photo taken in 2008. Locations where fish and bird surveys were conducted in 2012, the bermed area (created in early 1990s), and areas with dense stands of emergent vegetation and aquatic habitat are indicated.

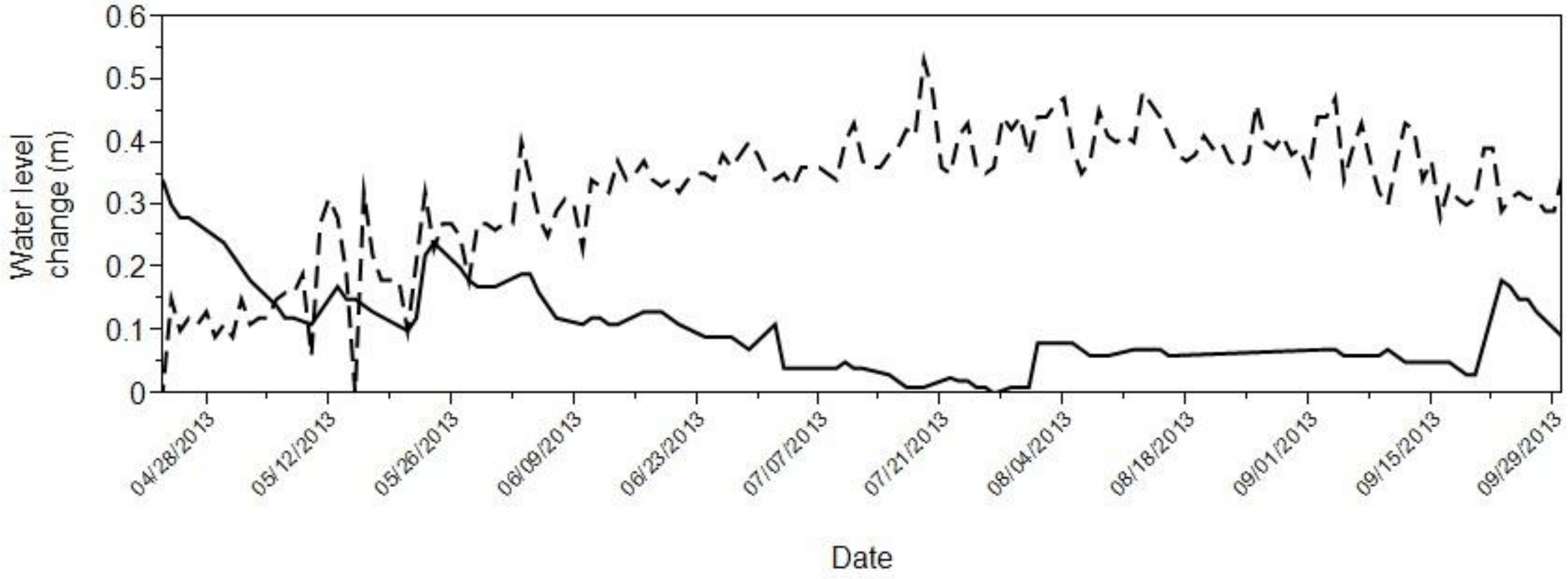

Figure 1.6 Relative change in water levels during the 2013 growing season between Wye Marsh (solid line) and Matchedash Bay/ Georgian Bay (broken line). Data were collected in the morning at Wye Marsh and 07:00 for Matchedash Bay, from April 23, 2013 to September 30, 2013.

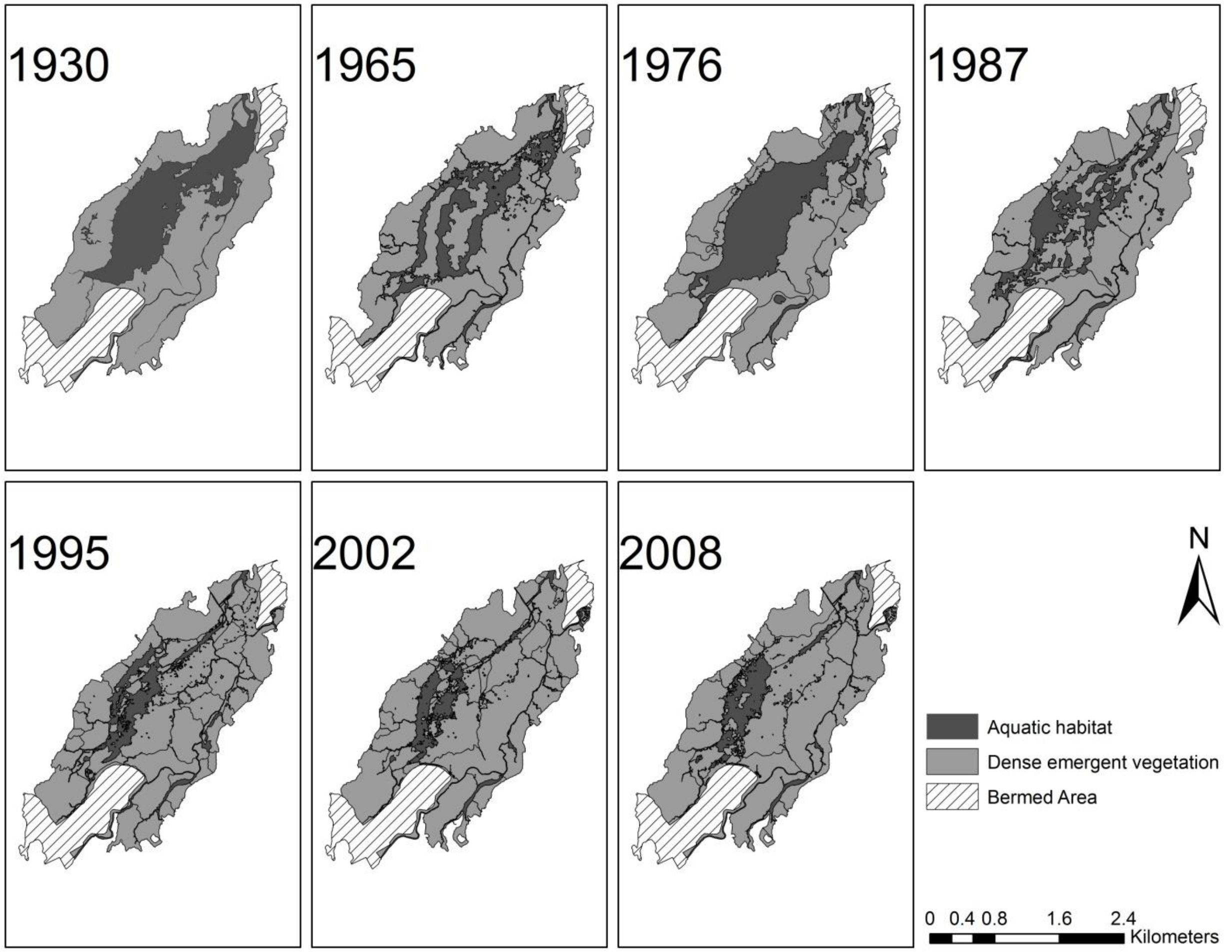

Figure 1.7 Changes in areal cover of dense emergent vegetation and aquatic habitat in Wye Marsh from 1930 to 2008. All maps were digitized from aerial photos acquired from April to October (National Air Photo Library, Canada and Ontario Ministry of Natural Resources).

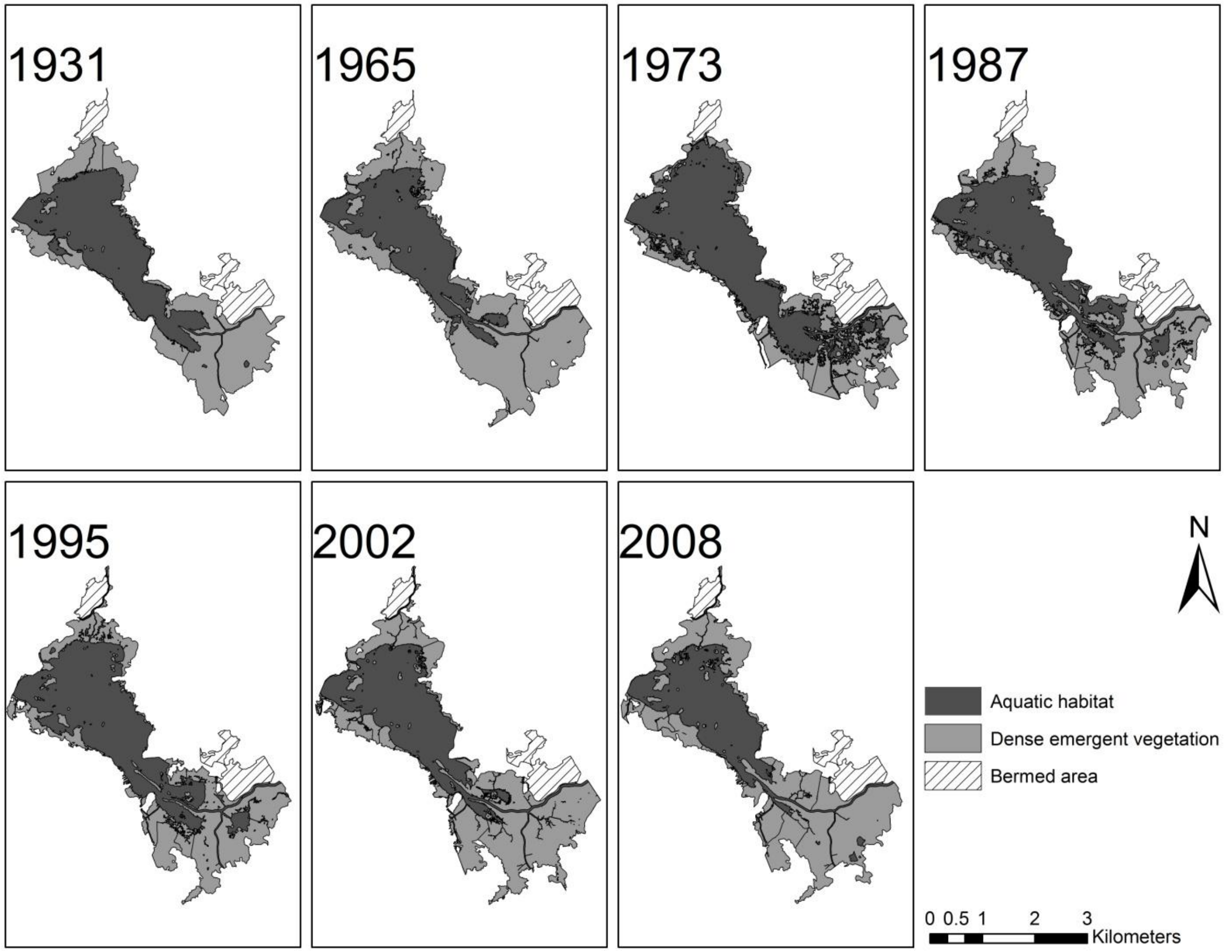

Figure 1.8 Changes in areal cover of dense emergent vegetation and open-water areas in Matchedash Bay from 1931 to 2008. All maps were digitized from aerial photos acquired from April to October (National Air Photo Library, Canada and Ontario Ministry of Natural Resources).

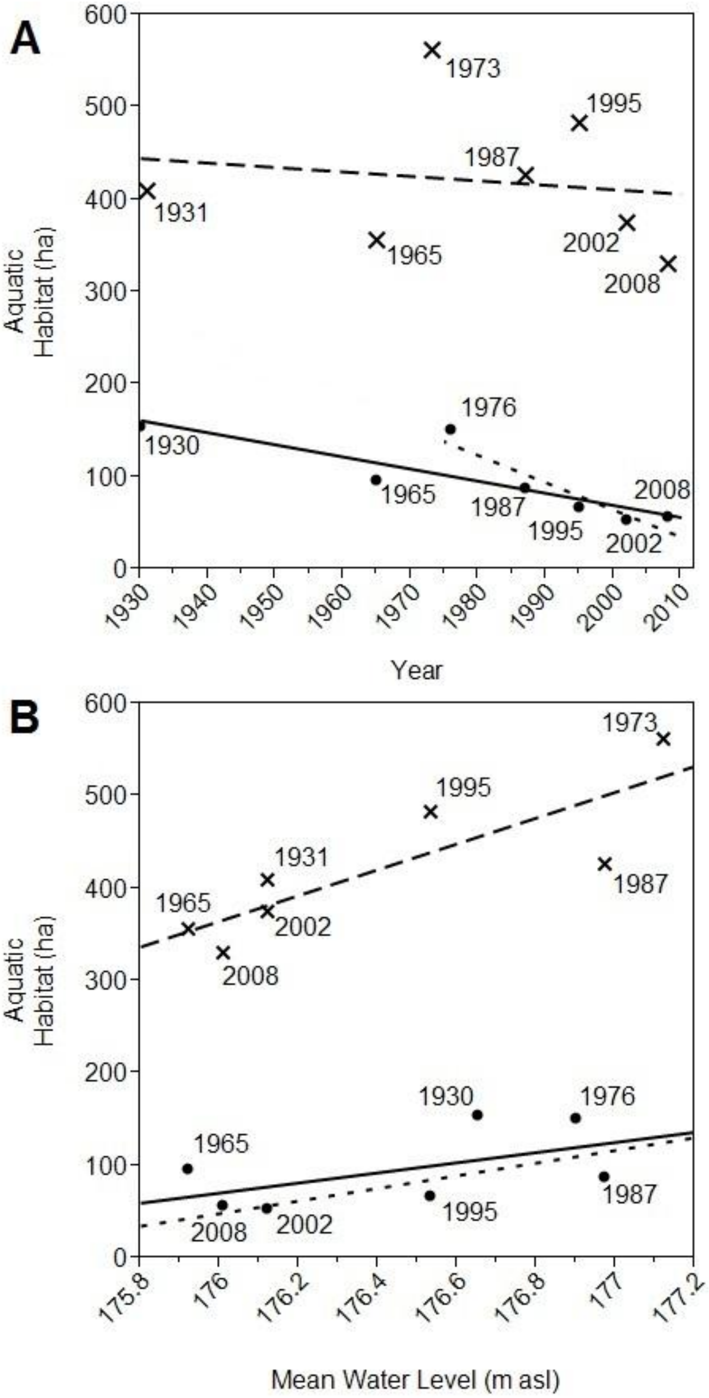

Figure 1.9 Plot of available aquatic habitat in Matchedash Bay (crosses) and Wye Marsh (circles) versus a) time and b) mean annual water level of Georgian Bay. Broken line is the least-squares regression line through Matchedash Bay data (1931 to 2008), while the solid line is a regression through Wye Marsh data (1930 to 2008). Also shown is a regression line through Wye Marsh data that exclude the 1930 and 1965 data points (dotted line).

CHAPTER 2: Comparison of ecosystems in a chain of beaver impounded wetlands in eastern Georgian Bay: a glimpse of the future in a predicted low water-level crisis

By

Dallas Taylor*
and
Patricia Chow-Fraser

McMaster University
Department of Biology, 1280 Main St. West
Hamilton, ON, L8S 4K1, Canada
Telephone: 905-525-9140 ext. 27461
*Corresponding author e-mail: taylod7@mcmaster.ca

Keywords: Laurentian Great Lakes, Georgian Bay, water level, impounded wetlands, fish, larval amphibians

Abstract

Water level fluctuation in the Laurentian Great Lakes is a primary mechanism that helps maintain high aquatic biodiversity in coastal marshes. Increased evaporation and changing precipitation patterns associated with global climate change is expected to lead to long-term reductions in water levels and severely disrupt these water-level fluctuations in the Great Lakes. Human adaptations to these changes could take the form of wetland diking, but since there are often unintended consequences of such management actions, it is prudent to conduct a study within Georgian Bay to better understand the effects of impoundment on a large range of ecosystem components. Here, we use data collected in a chain of three coastal wetlands that are located in a minimally disturbed embayment of eastern Georgian Bay, to test hypotheses regarding the effect of impoundment. The two wetlands located at higher elevations have been impounded successively by beaver activities over the past four decades, and therefore, have been disconnected from waters of Georgian Bay for 10 to 30 years. By contrast, the coastal marsh located at the lowest elevation has not been disconnected from the bay in recent times. Consistent with previous studies, we found significant differences between the impounded and un-impounded wetlands with respect to seasonal trends in water levels, water chemistry, and species composition of the aquatic macrophyte community, as well as differences in the abundance and diversity of the fish and anuran communities. Diking and damming eliminate the natural variability of water levels that maintain biodiversity of coastal wetlands and have a net negative impact on aquatic biota.

Introduction

One of the many consequences of global climate change in North America is lower than normal water levels in the Great Lakes (Mortsch and Quinn 1996; Angel and Kunkel 2010). For lakes Huron and Michigan, which share the same hydrological regime, water levels have been at or below the long-term average since 1999 (see Figure 2.1). The mean water level during the period of decline (1999 to 2013) is 176.04 m above sea level (a.s.l.), and compared with the long-term mean of 176.57 m a.s.l. (see Table 2.1), water level has been 53 cm below average over the past 14 years. No other Great Lake (Superior, Erie or Ontario) is associated with as large or long of a deviation below their long-term mean (see Table 2.1; Gronewold et al. 2013). Though there are many other consequences, these sustained low water levels have had immediate and devastating effects on the quantity and quality of fish habitat in coastal wetlands of eastern Georgian Bay (Lake Huron), an area known for some of the highest quality coastal wetlands along the Great Lakes shoreline (Cvetkovic and Chow-Fraser 2011). Some of these negative effects include loss of up to 24% of breeding and nursery habitat because access to migratory fish is prevented (Fracz and Chow-Fraser 2012a), deterioration in habitat structure related to disappearance of canopy-type submergent vegetation (Midwood and Chow-Fraser 2012; Leblanc et al. in sub.) and a reduction in species richness of the fish and plant communities because of increased patches of high-density floating vegetation (Midwood and Chow-Fraser 2012).

One possible solution to prevent wetland loss is to create embankments or dikes around the perimeter of a wetland (or sections of the wetland) so that water can be manipulated behind the impoundment. Diking has been used in the Great Lakes, particularly in Lake Erie, to create waterfowl habitat for hunting (Herdendorf 1992) and to protect coastal marshes from erosion by wind and wave action of the open lake, especially during periods of high water levels (Herdendorf 1992; Gottgens et al. 1998). Although many support the use of impoundments to improve recreational opportunities for waterfowl hunting, there are also criticisms to this approach which point out that dikes are barriers to aquatic species that migrate between the lake and coastal wetlands (Jude and Pappas 1992; Johnson et al. 1997; Markham et al. 1997). Impoundments can also significantly alter nutrient chemistry (Mitsch 1992; Fracz and Chow-Fraser 2013), the species composition of aquatic vegetation within wetlands (Keddy and Reznicek 1986; Mitsch 1992; Sherman et al. 1996; Johnson et al. 1997; Gottgens et al. 1998), as well as the fish (Keast and Fox 1990; Johnson et al. 1997; Markham et al. 1997), bird (Kadlec 1962; Galloway et al. 2006; Monfils et al. 2014) and invertebrate (McLaughlin and Harris 1990) communities.

When evaluating the cost-benefits of using dikes or dams to deal with problems associated with declining water levels in Georgian Bay, managers must also keep in mind that almost all published studies conducted to understand the effects of diking on wetlands in the Great Lakes basin, have focused on wetlands of the lower Great Lakes (e.g. Mortsch et al. 2006), and in particular Lake Erie (e.g. Robb 1989). The lower Great Lake wetlands have very different bedrock and surrounding land uses compared with those in eastern Georgian Bay. The granitic bedrock of the Canadian Shield, underlying wetlands of eastern and northern Georgian Bay (deCatanzaro and Chow-Fraser 2011), is not easily weathered and is therefore very different from the limestone, shale and sandy sediment that can be easily eroded in Lake Erie wetlands (Herdendorf 1992). Such differences influence how wetlands develop and are modified by physical forces, and these may dictate what species are found within them. Because many of the Georgian Bay wetlands are located on Precambrian Shield, they are highly coloured with low nutrient and low ionic content, and aquatic species that dominate in Georgian Bay are different from those in Lake Erie wetlands that are nutrient-rich, alkaline and have relatively high sediment loads from agricultural activities in their watersheds (Chow-Fraser 2006; Cvetkovic and Chow-Fraser 2011). Therefore, a proper study must be conducted first to determine if results from Lake Erie can be directly transferred to Georgian Bay with respect to effects of impoundment on biotic and abiotic components of coastal wetland ecosystems.

The goal of our paper is to investigate how impoundment may affect the water chemistry and food webs in a chain of wetlands, two of which have become sequentially disconnected from waters of Georgian Bay by beaver activities over the past four decades. To examine these effects in the absence of human activities, we chose sites that have been managed by a private fishing and hunting club in a relatively pristine condition for over 100 years. We compare differences in aquatic habitat between the two beaver impounded wetlands and the one coastal marsh that is still hydrologically connected with waters of Georgian Bay, focusing on water chemistry, seasonal changes in water levels and the aquatic macrophyte community. We also investigate how the impounded and un-impounded wetlands differ with respect to the fish and larval amphibian communities.

We hypothesize that impoundment would eliminate the erratic seasonal fluctuations in water level that characterize coastal marshes, as reported in Chapter 1. Consistent with previous studies, we predict that the water chemistry in the beaver impounded wetlands would be more heavily influenced by runoff from the watershed, whereas water chemistry in the coastal marsh would reflect properties of open water in Georgian Bay (deCatanzaro and Chow-Fraser 2011). We predict that beaver impoundments would change the structural habitat of the aquatic system by leading to a more homogenized vegetation structure (Keddy and Reznicek 1986; Midwood and Chow-Fraser 2012) and subsequently affect the fish (Johnson et al. 1997; Markham et al. 1997; Kowalski 2010) and larval amphibian communities (Cunningham et al. 2007). We also expect hydrological disconnection to reduce species diversity of the fish community in impounded wetlands because species that migrate in and out of coastal wetlands (such as the Northern pike and Smallmouth bass) would be eliminated. In contrast, we expect that the larval amphibian community should benefit from lower predation pressure in the impounded wetlands. By conducting this study under pristine conditions, our results will be directly applicable to the many undisturbed wetlands of eastern Georgian Bay without the need for further validation. Our study is timely and fills a void in the literature to help resource agencies determine how best to manage coastal wetlands under a low water-level crisis.

Methods

Site description

The three wetlands in this study were once part of a single wetland unit (15.33 ha) that we call the Black Rock Wetland Complex (referred to as Black Rock subsequently), which is located in Tadenac Bay, a relatively large embayment of eastern Georgian Bay (Figure 2.2). From an aerial photo taken in 1987 (National Air Photo Library of Canada, Ottawa), we identified the presence of a beaver dam, which cut off approximately a third of the wetland complex (5.39 ha) to form the oldest beaver pond (BLR30). Based on water level measurements and historical mean monthly water levels of Georgian Bay (Figure 2.1), BLR30 may have been reconnected with Tadenac Bay in 1997 due to higher water levels. Between the time that 2002 and 2008 IKONOS satellite images were taken, a second beaver dam was built creating a second beaver pond (BLR10; 5.21 ha). This pond is therefore between 6 and 12 years old. The rest of the 4.73 ha of wetland remain hydrologically connected to Tadenac Bay as a coastal marsh with a rock sill at its outlet (BLR00). There is very little elevation difference among the three wetlands, with only 2 cm separating BLR30 from BLR10 and 78 cm separating BLR10 from BLR00. Based only on visual observations and placement of fyke nets in these wetlands, the maximum depth of all wetlands is approximately 2 metres. There is also a shallow layer of sand and fine-particle sediment over the bedrock (Weiler 1988). The surrounding upland consists mainly of mixed coniferous forest (deCatanzaro et al. 2009).

The Tadenac Club owns and manages most of the land and water in the Tadenac Bay watershed and has done so for over 100 years. Tadenac Bay is managed as a wilderness area mostly for hunting and fishing and the Club limits the number of members and motorized boats in the property at one time. Consequently, in multiple surveys of Tadenac Bay between 2004 and 2012, it has been identified as one of the least human-impacted embayments in Georgian Bay based on abundance of pollution intolerant flora and fauna and exceptionally clear and low-nutrient water (Chow-Fraser 2006; Croft and Chow-Fraser 2007; Cvetkovic and Chow-Fraser 2011; deCatanzaro and Chow-Fraser 2011; Midwood and Chow-Fraser 2012).

Field sampling

We collected water chemistry data on July 16, 2013 and processed and analyzed according to the methods described in Chow-Fraser (2006). We analyzed specific conductance (COND), pH, sulphates (SULPH), true colour (COL), total phosphorus (TP), soluble reactive phosphorus (SRP), total ammonia-nitrogen (TAN), total nitrate-nitrogen (TNN), and total suspended solids (TSS; Table 2.2). Water samples were collected in the middle of the water column at deep pockets of the wetland in areas devoid of submerged aquatic vegetation (to prevent contamination by epiphyton). For the two beaver ponds, the deepest part was immediately behind each beaver dam, whereas for the coastal marsh, this was in the middle, approximately 100 metres from the outlet (see Figure 2.2).

To measure daily changes in water level in the beaver ponds during the 2013 growing season, two Levelogger Junior Edge water level loggers (Model 3001, Solinst Canada Ltd., Georgetown, ON), which measure absolute pressure, were hung within perforated PVC piping and set within an open pool of water directly behind each beaver dam. The PCV piping was used to hold the loggers off the sediment surface and to inhibit interference by wildlife. To prevent buildup of sediment in the piping that may negatively affect the performance of the logger, we covered the flow-through holes in the piping with nylon stockings. To compensate for barometric pressure changes, a Barologger (Model 3001, Solinst Canada Ltd., Georgetown, ON) was set approximately

50 cm above the top of the dam in BLR10 within the Levelogger housing. We downloaded data and used the proprietary software provided by the company (Solinst Canada Ltd., Georgetown, ON) for processing. The processed data were then exported into JMP version 10 (SAS Institute Inc., Toronto, Ontario, Canada) for further analysis. Water-level data measured in Severn Sound (obtained from Fisheries and Oceans Canada) were assumed to be representative of water levels in BLR00.

Aquatic macrophytes were sampled in August 2013 in both BLR00 and BLR10. Surveys were conducted from canoes in deep water (>1 m) or by walking in waders in shallow water. All aquatic vegetation occurring in 0.5 m by 0.5 m quadrats were identified to species whenever possible. Areas too deep to reach were sampled by dragging a metal rake along the bottom and pulling up macrophytes to identify. Each site was sampled repeatedly until no new species was identified in three consecutive quadrats. Unfortunately, a thunderstorm prevented us from conducting a proper vegetation survey of BLR30; however, the time spent collecting water samples and sampling for fish afforded us an opportunity to verify that the plant community was more similar to that of BLR10 than of BLR00.

Surveys of the fish community were conducted in 2012 and 2013 with three sets of paired fyke nets in each wetland (following protocols described in Seilheimer and Chow-Fraser (2006)). One pair of large nets with wings (4 mm bar mesh, 4.25 m lengths, 1.0 m x 1.25 m front opening), one pair of large nets with no wings (4 mm bar mesh, 4.25 m lengths, 1.0 m x 1.25 m front opening) and one pair of small nets with wings (4 mm bar mesh, 2.1 m length, 0.5 m x 1.0 m front opening) were used. Each net was set parallel to the emergent zone at the 1 m depth for the large nets and 0.5 m depth for the small nets. Nets were connected with a 7.0 m lead with wings set at 45° angles and were left for approximately 24 hours. Fish were then collected, identified to species, measured (total length) and released unharmed. Net locations were chosen based on accessibility within the wetland since both beaver ponds fill in with dense floating mats of mud and vegetation during the summer months. Data were classified as sport fish or prey fish, and whether they were resident or migratory (Jude and Pappas 1992; Wei et al. 2004).

We followed the standardized protocol used by Fracz and Chow-Fraser (2012b) to survey the larval amphibian community once each month between June and August during 2012, and again in mid-July, 2013. In the 60-minute surveys, we walked along the perimeter of each wetland, sweeping a D-frame Amphibian net (Wildlife Supply Company, Yulee, Florida, USA) rapidly through ankle-deep water perpendicular to the shoreline (maximum depth of 5-10 cm). Surveys always started at a location to the right of the wetland opening and proceeded counter clockwise around the perimeter. Captured larval amphibians were stored in a bucket of water until the sampling hour was completed. Water in the bucket was constantly replenished with pond water to prevent water temperature from rising in the bucket during the survey. At the end of the sampling hour, larval amphibians were identified to species and released unharmed into the wetland where they were caught.

Analyses

All data were compiled and analyzed in JMP 10 (SAS Institute Inc., Toronto, Ontario, Canada). A least squares regression line was calculated for water levels at each site to compare general trends. We determined day-to-day variation in water level by calculating the absolute difference between water level of a given day and the water level of the previous day. Since our hypothesis was that day-to-day variation would differ between hydrologically connected and impounded sites, we used a non-parametric Kruskal-Wallis test to compare means and a Steel-Dwass post-hoc test to compare individual site differences. Alpha (α) and Gamma (γ) Diversity scores were calculated for each wetland according to Veech et al. (2002). In addition to Alpha and Gamma Diversity, Shannon's diversity index scores were also calculated and compared among sites and months. Where assumptions were met, an ANOVA was used to compare means; otherwise we used a non-parametric Kruskal-Wallis test and Steel-Dwass post-hoc test ($\alpha = 0.05$). We used the Sorensen similarity index to quantify the degree of overlap between two given fish communities or the same community over two different time periods (McCune et al. 2002).

Results

Specific conductance and pH values generally decreased as sites became progressively more disconnected with Georgian Bay; the highest values were associated with the coastal marsh (BLR00), intermediate values with the younger beaver pond (BLR10) and lowest values with the older beaver pond (BLR30) (Table 2.2). We observed similar trends for sulphate concentrations, which decreased from 4.0 $mg \cdot L^{-1}$ in BLR00 to concentrations below detection in both impounded wetlands. Both of the impounded wetlands had much higher water colour compared with that in the coastal marsh, and this was mirrored in the TP concentrations (Table 2.2). The soluble fraction of phosphorus (SRP), however, was all at background level. There was no consistent trend with respect to levels of inorganic nitrogen (either TAN or TNN) and degree of hydrological connectivity, but the concentrations were all relatively low. The TSS concentrations were uniformly low in BLR00 and BLR30, but quite elevated in BLR10 (Table 2.2).

Both the magnitude (range in values of 14 and 15 cm, respectively) and shape of the hydrographs for BLR10 and BLR30 were very similar. After a slight increase from mid-May to early June, water levels in both beaver ponds declined gradually over remainder of the season (Figure 2.3). By contrast, water levels in the coastal marsh varied much more erratically from day to day and there was an overall increase from May to early August (range in values of 51 cm), after which levels trended gradually downwards. We fitted a least squares linear regression through both datasets to contrast the opposing trends in the hydrographs; the impounded wetlands clearly showed a downward trend after June while the coastal marsh showed an upward trend until August. We quantified the variability of daily water-level fluctuations by calculating the absolute difference between two consecutive days. Mean daily change in water levels was statistically different among sites (Kruskal-Wallis, Chi-square = 143.24, *p* = <0.0001); the average daily change in water level for BLR00 was 4 cm, which was significantly larger than 0.7 cm for BLR10 (Steel-Dwass, Z = -10.41, *p* = <0.0001) and 0.6 cm for BLR30 (Steel-Dwass, Z = -10.29, *p* = <0.0001). No statistical difference was found between means of BLR10 and BLR30 (Steel-Dwass, Z = 0.75, *p* = 0.73).

In addition to differences in hydrographs, we also noted qualitative differences in the species composition of the aquatic macrophyte community between the beaver ponds and the coastal marsh (Table 2.3). There were only 12 aquatic plant taxa in the younger impoundment (BLR10), compared with 23 in the coastal marsh (BLR00). Nine species were common in both wetlands, with 14 species unique to the hydrologically connected wetland and 3 unique to the impoundment. Both wetlands contained commonly sampled and uncommon species found in Georgian Bay wetlands (Table 2.3) and at least one species from each of three major categories: emergent, floating aquatic vegetation and submerged aquatic vegetation.

We expected the severed connection with Georgian Bay to lead to quantifiable differences in diversity and structure of the fish community in the two beaver ponds compared with the coastal marsh. The monthly α-Diversity for each site, however, did not differ significantly (ANOVA, F = 0.22, $p = 0.95$); monthly values for BLR00 were 10 in June, 9 in July, and 7 in August, while corresponding scores for BLR10 were 10, 7 and 8 during the same time period. Similarly, mean Shannon H' scores did not vary significantly among months (Kruskal-Wallis, Chi-square = 8.72, $p = 0.12$), ranging from 0.97 to 1.41 in BLR00 and from 1.04 to 1.41 in BLR10 (Table 2.4).

When we examined γ-Diversity for the pooled monthly samples, however, we did find differences between sites, with 16 fish species found in the coastal marsh and only 11 for the beaver pond (Table 2.4). We also found qualitative differences with respect to the species composition. The large sport fish species such as Northern pike, Yellow perch, Smallmouth and Largemouth bass were almost exclusively found in BLR00, the coastal marsh. On the other hand, the unique species found in BLR10 were Brook stickleback, Brassy minnow, and Iowa darter. Although Largemouth bass was found in both wetlands, there was only a single individual in the beaver pond which had been caught in August. In the coastal marsh, this species was found in relatively high numbers throughout the summer (Table 2.4). Distribution of Blackchin shiner, Central mudminnow, Golden shiner, and Bowfin, was skewed in the opposite direction, with these being found mostly in the beaver pond rather than in the coastal marsh. With respect to migratory guild, we found 2 to 4 migratory species in the hydrologically connected wetland, and at least half of them were sport fish. In contrast, we found no migratory species in the impounded wetland.

We calculated Sorensen similarity index scores to compare degree of overlap in fish assemblages for the two wetlands over the course of the summer (Table 2.5). The monthly data for BLR10 produced the highest similarity scores, ranging from 0.78 to 0.93. By contrast, similarity scores comparing monthly data for the coastal marsh were much lower, ranging from 0.35 to 0.63, and reflected the gradual species replacement through the season. When comparing similarity between sites, we found low values ranging from 0.30 to 0.67. It was interesting to note that all the wetlands shared a high overlap in species during August (similarity score of 0.67), whereas they appeared to be quite different during June and July (score of 0.40 and 0.38, respectively).

Next, we examined the year-to-year variation in the fish caught in BLR00 and BLR10. Almost all of the larger sport fish species caught in the coastal marsh during July 2012 were also caught in 2013, and their relative abundances were also very similar between years (Table 2.6). Included in these were Northern pike, Yellow perch, Largemouth bass and a single occurrence of Smallmouth bass that was not replicated in 2013. Of the remaining fish, Pumpkinseeds and Brown bullheads were dominant in both years, but a few individuals of Bowfin and Banded killifish were also caught in 2012. By comparison, no sport fish species was found in the beaver pond in July of either year; instead, the community was dominated by seven species including abundant Pumpkinseed, Brown bullhead, Golden shiner and Blackchin shiner, and less common species such as Brook stickleback, Bowfin, and Central mudminnow.

We investigated how the age of the beaver pond might have affected the fish communities when compared with the coastal marsh (Table 2.7). Fish assemblages in BLR10 and BLR30 were very similar, but we were surprised to see several individuals of Yellow perch and Black crappie in the samples collected in the older beaver impoundment since both are considered migratory species. We did not expect to see them in a beaver pond that had been impounded from Tadenac Bay for over 30 years. Despite these deviations from expectations, sport fish such as Northern pike and Largemouth bass were only found in the coastal marsh, and this is consistent with what was expected for migratory fish.

We investigated if the apparent reduction in migratory sport fish may also have an effect on the size of the fish within the impounded wetlands. Only Pumpkinseeds were sufficiently abundant in all three wetlands to statistically compare the size of the fish. The mean length of individuals (9.60 ± 0.31 cm) was significantly larger in the coastal marsh than in the newer beaver pond (7.96 ± 0.33 cm) (Steel-Dwass, Z = -2.76, *p* =

0.016) although no significant differences were found when lengths of fish from the older beaver pond (8.87 ± 0.16 cm) were compared with those in the coastal marsh (Steel-Dwass, Z = -1.47, *p* = 0.30) (Figure 2.4).

Finally, we examined the larval amphibian community to determine if there were qualitative differences in the species composition. Since adult stages of frogs are semi-aquatic, they can move freely from wetland to wetland, whereas their larval stages are restricted to their natal wetland until they mature. The coastal marsh only supported Green frog and Northern leopard frog and these were only found during the June survey. By contrast, the newer beaver pond supported two additional amphibians, the American bullfrog and the Spring peeper throughout the summer (Table 2.7). The older beaver pond also supported two more species in July than the coastal marsh in 2013.

Discussion

Water in Georgian Bay is known to be oligotrophic, with heavy influences from limestone bedrock in the South and West (Weiler 1988). Previous studies have investigated changes in water chemistry in Georgian Bay as it moves nearshore into coastal wetlands (deCatanzaro and Chow-Fraser 2011) and how water changes when beaver activity removes connection to the open bay (Fracz and Chow-Fraser 2013). We confirmed that water chemistry in Black Rock wetlands are typical of similar wetlands throughout Georgian Bay, but are quite different from that of open water Georgian Bay, which has higher specific conductance (184 $\mu S \cdot cm^{-1}$), alkalinity (pH = 8.18), nitrates (0.18 $mg \cdot L^{-1}$) and sulphates (11.9 $mg \cdot L^{-1}$) (Table 2.2) but low phosphorus levels (TP = 4.9 $\mu g \cdot L^{-1}$ and SRP = 0.7 $\mu g \cdot L^{-1}$) and is very clear (TSS = 1.40 $mg \cdot L^{-1}$ and COL = 2.9 pt-co). As water moves into the coastal marsh of Black Rock, the reduced mixing with the bay is reflected in higher levels of nutrients and sediments and higher colour, all of which are due to a greater watershed influence and internal cycling (deCatanzaro and Chow-Fraser 2011). In the beaver impoundments, specific conductance, pH and sulphates are lowest, and are consistent with the effects of very limited mixing with bay water (Table 2.1; deCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013). We know from Fracz and Chow-Fraser (2013) that dissolved oxygen is lower in beaver impoundments compared to connected wetlands and reducing conditions brought on by the low oxygen may lead to lower concentrations of nitrate and sulphate, as well as lower pH values. These trends, therefore, support the notion that as the wetland loses hydrological connection to the open bay, run-off from the ion- and nutrient-poor Shield rock and geochemical conditions within the wetlands become a more dominant determinant of water chemistry (deCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013).

The patterns of water-level fluctuations in the two beaver-impounded wetlands were very similar, with respect to direction and amplitude of change and the day-to-day variability (Figure 2.3). In both cases, water levels declined through the season with summer amplitudes of only 15 and 14 cm, respectively for BLR10 and BLR30, and the day-to-day fluctuations were approximately 6 times lower than those in the coastal marsh, which had a summer amplitude of 51 cm. In Chapter 1, we found a similar dampening of water-level fluctuations when we compared the hydrology of an impounded wetland to that of a hydrologically connected coastal wetland in southeastern Georgian Bay.

Our hypothesis was that a change in the hydrological regime would alter the structural habitat used by many aquatic species by inhibiting cyclical succession brought on by water level changes (Keddy and Reznicek 1986). This was in fact the case, where the species richness of aquatic macrophytes in the beaver impounded wetland was approximately half that of the hydrologically connected wetland. Sherman et al. (1996) and Galloway et al. (2006) found opposite results in Lake Erie and across the lower Great Lakes, respectively, where undiked wetlands contained fewer species and less vegetative areal coverage compared to diked wetlands. Robb (1989) and Thiet (2002) also found fewer species in the undiked wetlands of Lake Erie, where differences between diked and undiked wetlands were primarily due to changes in water level regimes. Site-to-site variation within each group was high, making it clear that local processes must be considered before generalizing impacts of diking (Robb 1989; Galloway et al. 2006). These studies attribute the differences between diked and undiked wetlands to much deeper, more turbid water in the undiked wetlands which excludes both emergent and submerged vegetation. In Georgian Bay, however, turbidity is not an issue (Chow-Fraser 2006), leaving water level fluctuation being the major structural driver. It appears that reduced water-level fluctuation has changed the conditions within the wetland, giving a competitive advantage to those species best suited for the more constant water-level conditions (Keddy and Reznicek 1986). To support this, we took the Georgian Bay subset of Croft and Chow-Fraser's (2007) dataset and arranged it by how commonly a species was sampled in hydrologically connected wetlands. We found that both common (e.g. *Pontederia cordata* and *Nymphaea odorata*) and rare (e.g. *Stuckenia pectinatus* and

Nymphoides cordata) species are present in both study sites, again indicating that richness differences are most likely caused by the dampened fluctuations and not by the quality of plant habitat.

The literature indicates that diversity in the fish community decreases with a loss of connection with the larger water body (Keast and Fox 1990; Brazner 1997; Johnson et al. 1997; Taylor 1997; Kowalski 2010). In this study, however, we found no significant differences in species richness and diversity between the beaver ponds and the coastal marsh, but we found an overall decline in the number of sport fish and migratory species in BLR10 and BLR30 compared with BLR00. This is consistent with Keast and Fox (1990), who noted an absence of specialists and large predators in other beaver dammed wetlands in Ontario. Here, we saw a 30% decline in the seasonal γ-Diversity, which may be attributed to the movement of those species in and out of the hydrologically connected wetland (Table 2.4).

The fish species we found exclusively in the impoundments were dark, cool-water species that prefer dense vegetation and are tolerant of a wide variety of habitats (Scott and Crossman 1998). Those that were common in both impounded wetlands were species with large niche breadths and tolerance to degraded conditions, such as Pumpkinseed, Golden shiner and Brown bullhead (Seilheimer and Chow-Fraser 2007). It appears that regardless of the age of the impoundment, the removal of the hydrological connection and changes in conditions within the wetland have led to the elimination of some important fish species such as Northern pike, Largemouth bass and Smallmouth bass. We found a much larger difference between species in June than in August between the coastal wetland and the beaver ponds, and we attribute this difference to the presence of migratory species in coastal marshes during spring (Jude and Pappas 1992; Wei et al. 2004).

The removal of top predators in ecosystems is known to cause a change in the community structure of prey species. Without a top predator, prey populations tend to be controlled by food availability rather than by predation, and this can be reflected in stunting of the population, which occurs because of increased competition for food (Tonn et al. 1992; Markham et al. 1997). Since impoundments are essentially closed systems, dispersal is prevented even when populations become overcrowded. McLaughlin and Harris (1990) showed that impoundments had a lower diversity of invertebrate prey for fish and Fox and Keast (1990) observed that Pumpkinseeds were smaller in beaver impoundments. In our study, the mean size of Pumpkinseeds was smaller in the younger beaver pond than in either the coastal marsh or the older beaver pond (Figure 2.4). This was the only wetland of the three without a top predator, and lends support to the hypothesis that stunting may have occurred in BLR10.

We also found differences in the larval amphibian communities that are consistent with effects of impoundments on the food web. We see both a different richness and abundance of larval amphibians between impounded and un-impounded sites. Our results, however, are inconsistent with Metts et al. (2001) who found that larval amphibian diversity was greater in connected streams than in beaver ponds in South Carolina and with Russell et al. (1999) who found no difference between connected stream diversity and differently aged beaver ponds. One reason for these differences may be due to the type of wetlands being studied (streams vs palustrine wetland and coastal marshes). Cunningham et al. (2007) proposed that connectivity is a major determining factor for breeding amphibians in palustrine wetlands, but these differences are species-specific. In this study, we found a much greater diversity in the beaver ponds. We suggest that the physical habitats in the impounded wetlands are different from the habitat of the connected wetland, with different water chemistry, lower fluctuations in water levels and less heterogeneity in the structural habitat. The dystrophic nature of the beaver ponds may also protect larval anurans from predators that feed visually.

Our original hypothesis was that age of the beaver pond would affect the response of the ecosystem to being hydrologically severed from Tadenac Bay. We did not, however, see many differences between the beaver ponds; both of them responded similarly to loss of hydrological connection with Georgian Bay, and had similar water chemistry and almost identical water-level regimes. They also shared similar food web structures, with higher abundances and more species of larval amphibians, and fewer migratory sport fish than BLR00. It is possible that BLR30 had been re-connected with Georgian Bay over the past 30 years and hence, the beaver pond may be younger than we think it is. At this point, we do not have any evidence that duration of impoundment affects the response of the ecosystem to hydrological severance.

Coastal wetlands are naturally dynamic systems and without the fluctuating water levels, they lose their biodiversity. We have shown that the water-level fluctuations in beaver impoundments are greatly dampened when compared to the day-to-day variations of hydrologically connected coastal marshes, and we attribute the net negative impact on ecosystem components to this change in hydrology. If diking were implemented to benefit waterfowl and marsh birds, water levels would have to be actively manipulated (Kadlec 1962; Monfils et al. 2014). Given that many of the wetlands in eastern Georgian Bay are remote and only accessible by boat, it would be too costly and impractical to manage any impounded wetlands to achieve this goal. Here, we are using beaver impoundments as a proxy for how human-made impoundments without active water level manipulation may change the ecosystem. We acknowledge that beavers are important ecosystem engineers and provide landscape scale diversity. The beaver impoundments in this study demonstrated new ecosystems as opposed to acting as protected coastal wetlands. However, proposed human-made impoundments are not natural and would be used for the purposes of protecting coastal wetlands. Due to the overall negative effects that we noted on coastal ecosystem components, we therefore suggest that managers should not consider using diking or damming as a solution to problems stemming from current low water levels.

Acknowledgements

We would like to thank the Sierra Club Canada Foundation first and foremost for providing funding for the project and the Tadenac Club for allowing us access to their property to conduct this research. We could not have completed our research without the continued support of Mary Muter, who provided accommodations, a boat and vast knowledge of the area, as well as Mark Trudeau, who was there to assist us on numerous occasions. We appreciate the accommodations put up by Amy Beament and her family as well. Thank you to Dr. James Michael Waddington who provided helpful advice in the planning and implementation stages of this project and his continued support in the writing of this manuscript. We would also like to thank all of those dedicated field technicians and graduate students who assisted in the field and in the lab, including Amanda Fracz, Chris Biberhofer, Chantel Markle, Julia Rutledge, Nick Luymes, Caitlin Sommers, Jessica Watts, John Paul LeBlanc, Dan Weller, Brandon Burns and Aman Gill.

Literature Cited

Angel JR, Kunkel KE (2010) The Response of Great Lakes Water Levels to Future Climate Scenarios with an Emphasis on Lake Michigan-Huron. J Great Lakes Res 36:51-58.

Brazner JC (1997) Regional, Habitat, and Human Development Influences on Coastal Wetlands and Beach Fish Assemblages in Green Bay, Lake Michigan. J Great Lakes Res 23 (1):36-51.

Chow-Fraser P (ed) (2006) Development of the Water Quality Index (WQI) to Assess Effects of Basin-Wide Land-Use Alteration on Coastal Marshes of the Laurentian Great Lakes. Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Indiana Biological Survey, Bloomington, IN.

Croft MV, Chow-Fraser P (2007) Use and Development of the Wetland Macrophyte Index to Detect Water Quality Impairment in Fish Habitat of Great Lakes Coastal Marshed. J Great Lakes Res 33 (Special Issue 3):172-197.

Cunningham JM, Calhoun AJK, Glanz WE (2007) Pond-Breeding Amphibian Species Richness and Habitat Selection in a Beaver-Modified Landscape. J Wild Manag 71 (8):2517-2526.

Cvetkovic M, Chow-Fraser P (2011) Use of Ecological Indicators to Assess the Quality of Great Lakes Coastal Wetlands. Ecol Indic 11 (6):1609-1622.

deCatanzaro R, Chow-Fraser P (2011) Effects of Landscape Variables and Season on Reference Water Chemistry of Coastal Marshes in Eastern Georgian Bay. Can J Fish Aquat Sci 68 (6):1009-1023.

deCatanzaro R, Cvetkovic M, Chow-Fraser P (2009) The Relative Importance of Road Density and Physical Watershed Features in Determining Coastal Marsh Water Quality in Georgian Bay. Environ Manag 44:456-467.

Fox MG, Keast A (1990) Effects of Winterkill on Population Structure, Body Size, and Prey Consumption Patterns of Pumpkinseed in Isolated Beaver Ponds. Can J Zool 68 (12):2489-2498.

Fracz A, Chow-Fraser P (2012a) Impacts of Declining Water Levels on the Quantity of Fish Habitat in Coastal Wetlands of Eastern Georgian Bay, Lake Huron. Hydrobiologia 702 (1):151-169.

Fracz A, Chow-Fraser P (2012b) Importance of Hydrologic Connectivity for Coastal Wetlands to Open Water of Eastern Georgian Bay. Master's Thesis, McMaster University, Digital Commons Open Access Dissertations and Theses

Fracz A, Chow-Fraser P (2013) Changes in Water Chemistry Associated with Beaver-Impounded Coastal Marshes of Eastern Georgian Bay. Can J Fish Aquat Sci 70 (6):834-840.

Galloway M, Bouvier L, Meyer S, Ingram J, Doka S, Grabas G, Holmes K, Mandrak N (2006) Chapter 8. Evaluation of Current Wetland Dyking Effects on Coastal Wetlands and Biota. In: Mortsch LD, Ingram J, Hebb A, Doka SE (eds) Great Lakes Coastal Wetland Communities: Vulnerability to Climate Change and Response to Adaptation Strategies. Environment Canada and the Department of Fisheries and Oceans, pp 187-229.

Gottgens JF, Swartz BP, Kroll RW (1998) Long-Term GIS-Based Records of Habitat Changes in a Lake Erie Coastal Marsh. Wetl Ecol Manag 6:5-17.

Gronewold AD, Clites AH, Smith JP, Hunter TS (2013) A Dynamic Graphical Interface for Visualizing Projected, Measured, and Reconstructed Surface Water Elevations on the Earth's Largest Lakes. Environmental Modelling and Software 49 (1):34-39.

Herdendorf CE (1992) Lake Erie Coastal Wetalnds: An Overview. J Great Lakes Res 18 (4):533-551.

Johnson DL, Lynch J, W.E., Morrison TW (1997) Fish Communities in a Diked Lake Erie Wetland and an Adjacent Undiked Area. Wetlands 17 (1):43-54.

Jude DJ, Pappas J (1992) Fish Utilization of Great-Lakes Coastal Wetlands. J Great Lakes Res 18 (4):651-672.

Kadlec JA (1962) Effects of a Drawdown on a Waterfowl Impoundment. Ecology 43 (2):267-281.

Keast A, Fox MG (1990) Fish Community Structure, Spatial Distribution and Feeding Ecology in a Beaver Pond. Environ Biol Fishes 27:201-214.

Keddy PA, Reznicek AA (1986) Great Lakes Vegetation Dynamics: The Role of Fluctuationg Water Levels and Buried Seeds. J Great Lakes Res 12 (1):25-36.

Kowalski KP (2010) Overcoming Barriers to Coastal Wetland Ecosystem Rehabilitation: Strategies for the Great Lakes. Ph. D. thesis, University of Michigan, University of Michigan library, dissertations and theses

Leblanc JP, Weller JD, Chow-Fraser P (in sub.) Thirty-Year Update: Changes in Biological Characteristics of Degraded Muskellunge Nursery Habitat in Southern Georgian Bay, Lake Huron, Canada. McMaster University.

Markham CA, Lynch J, W.E., Johnson DL, Petering RW (1997) Comparison of White Crappie Populations in Diked and Undiked Lake Erie Wetlands. Ohio J Sci 97 (4):72-77.

McCune B, Grace JB, Urban DL (2002) Analysis of Ecological Communities. MjM software, Gleneden Beach, Oregon, USA.

McLaughlin DB, Harris HJ (1990) Aquatic Insect Emergence in Two Great Lakes Marshes. Wetl Ecol manag 1 (2):111-121.

Metts BS, Lanham JD, Russell KR (2001) Evaluation of Herptofaunal Communities on Upland Streams and Beaver-Impounded Streams in the Upper Piedmont of South Carolina. Am Midl Nat 145 (1):54-65.

Midwood JD, Chow-Fraser P (2012) Changes in Aquatic Vegetation and Fish Communities Following 5 years of Sustained Low Water Levels in Coastal Marshes of Eastern Georgian Bay, Lake Huron. Global Change Biol 18 (1):93-105.

Mitsch WJ (1992) Combining Ecosystem and Landscape Approaches to Great-Lakes Wetlands. J Great Lakes Res 18 (4):552-570.

Monfils MJ, Brown PW, Hayes DB, Soulliere GJ, Kafcas EN (2014) Breeding Bird Use and Wetlands Characteristics of Diked and Undiked Coastal Marshes in Michigan. J Wildl Manag 78 (1):79-92.

Mortsch LD, Ingram J, Hebb A, Doka SE (2006) Great Lakes Coastal Wetland Communities: Vulnerability to Climate Change and Response to Adaptation Strategies. Final Report Submitted to the Climate Change Impacts and Adaptation Program. Natural Resources Canada. Environment Canada and the Department of Fisheries and Oceans, Toronto, Ontario.

Mortsch LD, Quinn FH (1996) Climate Change Scenarios for Great Lakes Basin Ecosystem Studies. Limnol Oceanogr 41 (5):903-911.

Robb MD (1989) Diked and Undiked Freshwater Coastal Marshes of Western Lake Erie. Ohio State University, OhioLink

Russell KR, Moorman CE, Edwards JK, Metts BS, Guynn DCJ (1999) Amphibian and Reptile Communities Associated with Beaver (Castor Canadensis) Ponds and Unimpounded Streams in the Piedmont of South Carolina 14 (2):149-158.

Scott WB, Crossman EJ (1998) Freshwater Fishes of Canada, vol 4. Galt House Publications Ltd., Oakville, Ontario, Canada.

Seilheimer TS, Chow-Fraser P (2006) Development and Use of the Wetland Fish Index to Assess the Quality of Coastal Wetlands in the Laurentian Great Lakes. Can J Fish Aquat Sci 63 (2):354-366.

Seilheimer TS, Chow-Fraser P (2007) Application of the Wetland Fish Index to Northern Great Lakes Marshes with Emphasis on Georgian Bay Coastal Wetlands. J Great Lakes Res 33 (3):154-171.

Sherman DE, Kroll RW, Engle TL (1996) Flora of a Diked and an Undiked Southwestern Lake Erie Wetland. Ohio J Sci 96 (1):4-8.

Taylor CM (1997) Fish Species Richness and Incidence Patterns in Isolated and Connected Stream Pools: Effects of Pool Volume and Spatial Position. Oecol 110:560-566.

Thiet RK (2002) Diversity Comparison between Diked and Undiked Coastal Freshwater Marshes on Lake Erie During a High-Water Year. J Great Lakes Res 28 (2):285-298.

Tonn WM, Paszkowski CA, Holopainen IJ (1992) Piscivory and Recruitment: Mechanisms Structuring Prey Populations in Small Lakes. Ecol 73 (3):951-958.

Veech JA, Summerville KS, Crist TO, Gering JC (2002) The Additive Partitioning of Species Diversity: Recent Revival of an Old Idea. Okios 99 (1):3-9.

Wei A, Albert D, Chow-Fraser P (2004) Influence of Shoreline Features on Fish Distribution in the Laurentian Great Lakes. Can J Fish Aquat Sci 61 (7):1113-1123.

Weiler RR (1988) Chemical Limnology of Georgian Bay and the North Channel between 1974 and 1980. Hydrobiologia 163 (1):77-83.

Table 2.1: Comparison of mean ± SE water levels (m, a.s.l.) for each Great Lake calculated for two time periods. Data are monthly means for each year obtained from the Great Lakes Water Level Dashboard (Gronewold et al. 2013). Difference is the mean for 1999-2013 subtracted from the mean for 1860-2013.

Great Lake	1999 - 2013	1860 - 2013	Difference
Superior	183.20±0.020	183.40±0.005	0.20
Huron-Michigan	176.04±0.031	176.57±0.010	0.53
Erie	174.22±0.015	174.22±0.008	0.00
Ontario	74.76±0.019	74.78±0.008	0.02

Table 2.2: Comparison of water chemistry data for the chain of beaver-impounded wetlands (BLR00, BLR10 and BLR30) in the Black Rock wetland complex in Tadenac Bay, Georgian Bay. Except for pH, water chemistry data for the Black Rock wetlands were collected and analyzed in 2013, whereas data for open water of Georgian Bay were collected and analyzed during summer of 2005 outside the inlet of Tadenac Bay in the King Bay marina (Ontario Ministry of Environment, unpub. data). pH for BLR00 was measured in 2012, whereas those for BLR10 and BL30 were measured in 2011.

Parameter	Open water of Georgian Bay	BLR00	BLR10	BLR30
Status of hydrological connection to open waters of Georgian Bay	--	Connected	Not connected	Not connected
Specific conductance ($\mu S \cdot cm^{-1}$)	184	96	31	25
pH	8.18	7.55	5.93	5.90
Sulphates ($mg \cdot L^{-1}$)	11.9	4.0	<0.05	<0.05
True colour (pt-co)	2.9	67	220	194
Total P ($\mu g \cdot L^{-1}$)	4.9	8.76	16.30	16.74
Soluble reactive P ($\mu g \cdot L^{-1}$)	<1	<1	<1	<1
Total Ammonia N ($mg \cdot L^{-1}$)	< 0.05	< 0.05	< 0.05	< 0.05
Total Nitrate N ($mg \cdot L^{-1}$)	0.18	0.05	0.05	0.08
Total suspended solids ($mg \cdot L^{-1}$)	1.40	1.17	5.75	1.17

Table 2.3: Presence (indicated by ✓) of aquatic vegetation in BLR00 and BLR10 identified in surveys conducted in August 2013. Data are presented in descending order of percent occurrence (% occurrence) of the species in question within 119 Georgian Bay coastal wetlands surveyed between 2003 and 2007 (from Cvetkovic and Chow-Fraser 2011). Total species richness of aquatic plants in BLR00 was 23 compared to 12 in BLR10; mean species richness of aquatic plants in the 119 wetlands was 23.4 (range from 0 to 46).

Common name	Scientific name	BLR00	BLR10	% occurrence
Pickerelweed	*Pontederia cordata*	✓	✓	95
Tape grass/ water celery	*Vallisneria americana*	✓		90
Fragrant water lily	*Nymphaea odorata*	✓	✓	83
Muskgrass spp.	*Chara spp.*	✓		79
Common yellow pond lily	*Nuphar variegata*	✓	✓	77
Slender water nymph	*Najas flexilis*	✓		76
Canadian waterweed	*Elodea canadensis*	✓		75
Wild rice spp.	*Zizania palustris*	✓		62
Hardstem bulrush	*Schoenoplectus acutus*		✓	61
Variable pondweed	*Potamogeton gramineus*	✓		60
Pipewort	*Eriocaulon aquaticum*	✓		57
Fern-leaf pondweed	*Potamogeton robbinsii*	✓		55
Beck's marsh marigold	*Bidens beckii*	✓	✓	53
Softstem bulrush	*Schoenoplectus validus*	✓	✓	51
Slender pondweed	*Potamogeton pusillus*	✓	✓	50
Large-leaved pondweed	*Potamogeton amplifolius*		✓	46
Broad-leaved pondweed	*Potamogeton natans*	✓	✓	46
Floating burreed	*Sparganium fluctans*	✓		41
Freshwater sponge	*Freshwater sponge*	✓		40
Water bulrush	*Scirpus subterminalis*	✓		40
Water shield	*Brasenia schreberi*	✓	✓	40
Common bladderwort	*Utricularia vulgaris*	✓	✓	39
Quillwort spp.	*Isoetes spp.*	✓		28
Stonewort spp.	*Nitella spp.*	✓		27
Sago pondweed	*Stuckenia pectinatus*		✓	20
Little floating hearts	*Nymphoides cordata*	✓		13

Table 2.4: Number of fish caught in paired fyke nets during June, July and August in BLR00 and BLR10 during 2012. Fish species are classified as being a sport fish with respect to Georgian Bay; migratory status was determined with Jude and Pappas (1992) and Wei et al (2004).

Common name	Scientific name	Description	Migratory status	BLR00			BLR10		
				Jun	Jul	Aug	Jun	Jul	Aug
Northern Pike	*Esox lucius*	Sport fish	Migratory		2	2			
Yellow Perch	*Perca flavescens*	Sport fish	Migratory		1	2			
Smallmouth Bass	*Micropterus dolomieu*	Sport fish	Migratory	1	1				
Largemouth Bass	*Micropterus salmoides*	Sport fish	Resident	25	59	51			1
Longear Sunfish	*Lepomis megalotis*	Prey fish	Migratory		1				
Black Crappie	*Pomoxis nigromaculatus*	Prey fish	Migratory	1					
Brook Stickleback	*Culaea inconstans*	Prey fish	Resident				5	3	1
Longnose Gar	*Lepisosteus osseus*	Prey fish	Resident	1					
Bowfin	*Amia calva*	Prey fish	Resident		1		6	11	2
Pumpkinseed	*Lepomis gibbosus*	Prey fish	Resident	14	91	75	184	90	84
Golden Shiner	*Notemigonus crysoleucas*	Prey fish	Resident			2	63	40	30
Brassy Minnow	*Hybognathus hankisoni*	Prey fish	Resident				1		
Brown Bullhead	*Ameiurus nebulosus*	Prey fish	Resident	11	23	4	36	62	49
Central Mudminnow	*Umbra limi*	Prey fish	Resident			1	2	6	5
Bluntnose Shiner	*Pimphales notatus*	Prey fish	Resident	7					
Banded Killifish	*Fundulus diaphanus*	Prey fish	Resident	1	3				
Iowa Darter	*Etheostoma exile*	Prey fish	Resident				1		
Blackchin Shiner	*Notropis heterodon*	Prey fish	Resident	23			12	27	6
Blacknose Shiner	*Notropis heterolepis*	Prey fish	Resident	3			10		

Table 2.5: Matrix summarizing Sorensen similarity scores between the fish communities sampled in BLR00 and BLR10 during June, July and August in 2012.

		BLR00			**BLR10**		
		June	July	August	June	July	August
BLR00	June	--					
	July	0.53	--				
	August	0.35	0.63	--			
BLR10	June	0.40	0.32	0.30	--		
	July	0.35	0.38	0.57	0.82	--	
	August	0.44	0.47	0.67	0.78	0.93	--

Table 2.6: Number of fish caught in paired fyke nets during July of 2012 and 2013 in BLR00 and BLR10. Fish species are classified as being a sport fish with respect to Georgian Bay; migratory status was determined with Jude and Pappas (1992) and Wei et al (2004).

Common name	Scientific name	Description	Migratory status		BLR00		BLR10	
					2012	2013	2012	2013
Northern Pike	*Esox lucius*	Sport fish	Migratory		2	2		
Yellow Perch	*Perca flavescens*	Sport fish	Migratory		1	1		
Smallmouth Bass	*Micropterus dolomieu*	Sport fish	Migratory		1			
Largemouth Bass	*Micropterus salmoides*	Sport fish	Resident		59	42		
Longear Sunfish	*Lepomis megalotis*	Prey fish	Migratory		1	2		
Black Crappie	*Pomoxis nigromaculatus*	Prey fish	Migratory			1		
Brook Stickleback	*Culaea inconstans*	Prey fish	Resident				3	5
Bowfin	*Amia calva*	Prey fish	Resident		1		11	2
Pumpkinseed	*Lepomis gibbosus*	Prey fish	Resident		91	42	90	36
Golden Shiner	*Notemigonus crysoleucas*	Prey fish	Resident				40	24
Brown Bullhead	*Ameiurus nebulosus*	Prey fish	Resident		23	137	62	323
Central Mudminnow	*Umbra limi*	Prey fish	Resident				6	17
Banded Killifish	*Fundulus diaphanus*	Prey fish	Resident		3			
Blackchin Shiner	*Notropis heterodon*	Prey fish	Resident				27	116
				α- Diversity	9	7	7	7

Table 2.7: Number of fish caught in paired fyke nets during July 2013 at BLR00, BLR10 and BLR30. Fish species are classified as being a sport fish with respect to Georgian Bay; migratory status was determined with Jude and Pappas (1992) and Wei et al (2004).

Common Name	Scientific Name	Description	Migratory Status	BLR00	BLR10	BLR30
Northern Pike	*Esox lucius*	Sport fish	Migratory	2		
Yellow Perch	*Perca flavescens*	Sport fish	Migratory	1		7
Largemouth Bass	*Micropterus salmoides*	Sport fish	Resident	42		
Longear Sunfish	*Lepomis megalotis*	Prey fish	Migratory	2		
Black Crappie	*Pomoxis nigromaculatus*	Prey fish	Migratory	1		5
Brook Stickleback	*Culaea inconstans*	Prey fish	Resident		5	
Bowfin	*Amia calva*	Prey fish	Resident		2	
Pumpkinseed	*Lepomis gibbosus*	Prey fish	Resident	42	36	146
Golden Shiner	*Notemigonus crysoleucas*	Prey fish	Resident		24	51
Brown Bullhead	*Ameiurus nebulosus*	Prey fish	Resident	137	323	336
Central Mudminnow	*Umbra limi*	Prey fish	Resident		17	2
Blackchin Shiner	*Notropis heterodon*	Prey fish	Resident		116	

Table 2.8: Number of larval amphibians caught in standardized D-net sweeps in BLR00, BLR10 and BLR30 during June, July and August in 2012 and 2013.

			June			July			August		
Year	Common name	Scientific name	BLR00	BLR10	BLR30	BLR00	BLR10	BLR30	BLR00	BLR10	BLR30
2012	American bullfrog	*Lithobates catesbeianus*	0	2	--	0	2	--	0	5	--
	Green frog	*Rana clamitans*	1	2	--	0	6	--	0	7	--
	Northern leopard frog	*Rana pipiens*	8	0	--	0	0	--	0	1	--
	Spring peeper	*Pseudacris crucifer*	0	4	--	0	0	--	0	0	--
2013	American bullfrog	*Lithobates catesbeianus*	--	--	--	0	0	1	--	--	--
	Green frog	*Lithobates clamitans*	--	--	--	0	1	2	--	--	--
	Northern leopard frog	*Lithobates pipiens*	--	--	--	0	3	0	--	--	--
	Spring peeper	*Pseudacris crucifer*	--	--	--	0	0	0	--	--	--

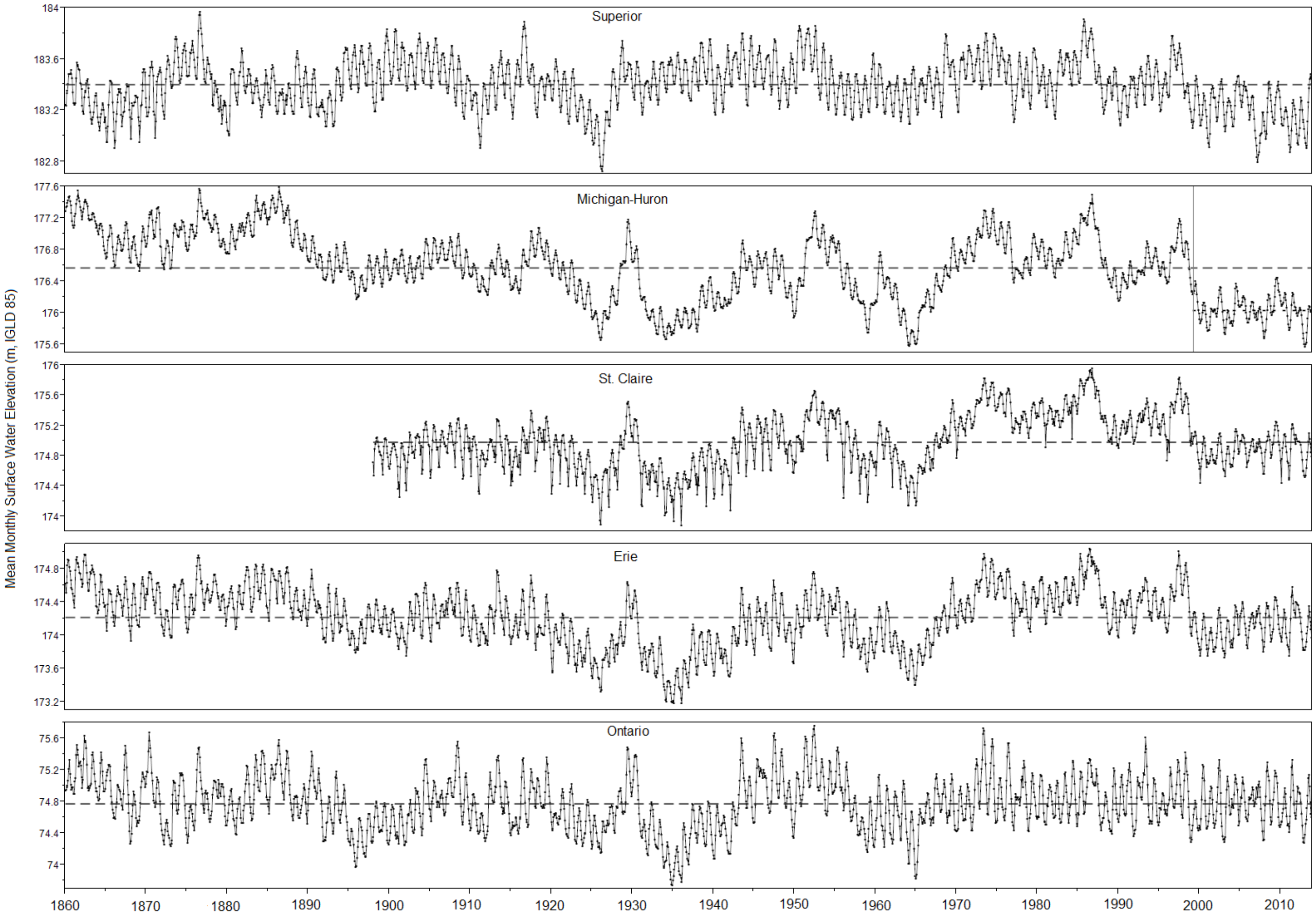

Figure 2.1: Hydrograph of mean monthly surface water elevation for each Laurentian Great Lake from 1860 to 2013 from data obtained in the online dashboard by Gronewold et al. (2013). Dashed lines indicate the long-term mean. Vertical line for Lake Michigan-Huron indicates the start of the sustained low water levels and dotted line indicates the mean water level for those 14 years.

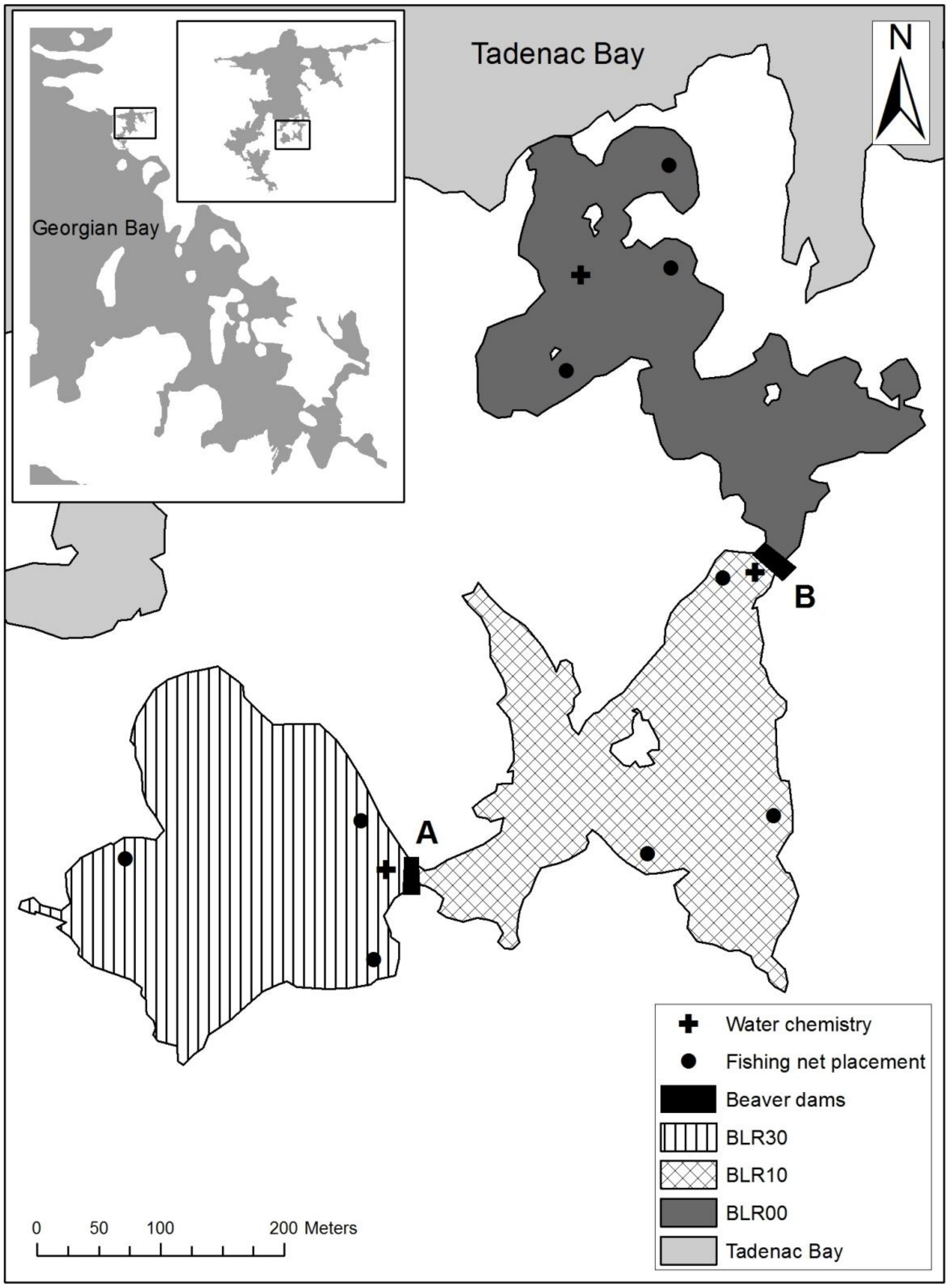

Figure 2.2: Map showing the chain of beaver-impounded wetlands within the Black Rock wetland complex of Tadenac Bay, Georgian Bay. "A" and "B" are locations of the beaver dams found in the 1987 aerial imagery (1:50,000) and 2008 IKONOS satellite imagery. Locations of sampling stations for water chemistry (+) and placement of paired fyke nets (•) are shown.

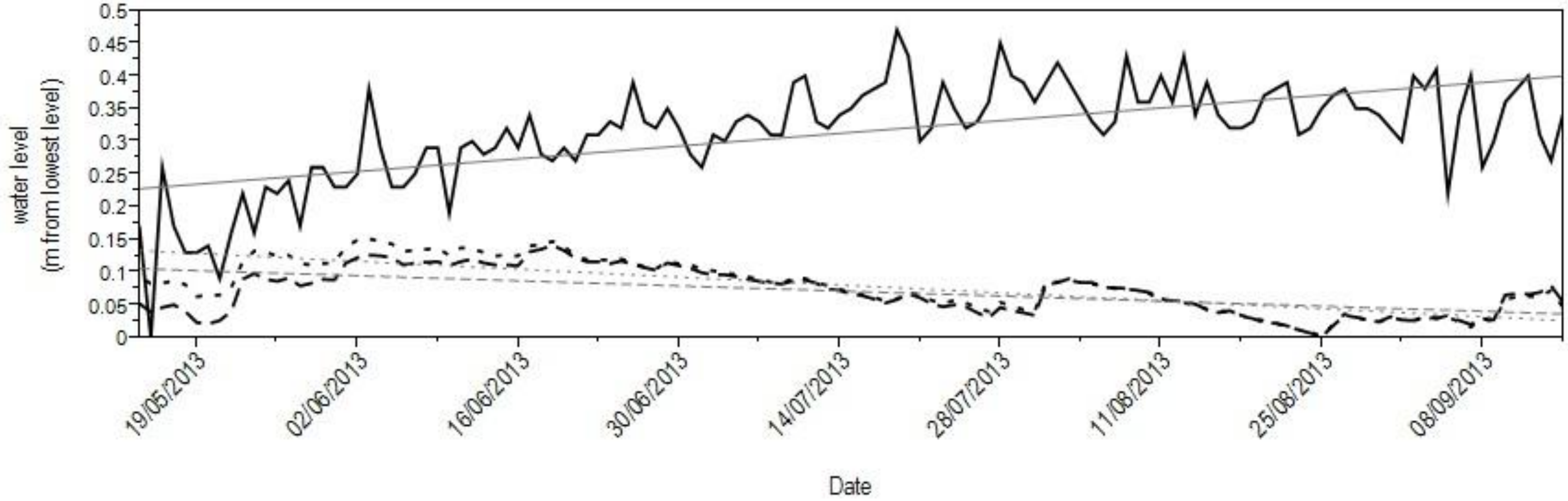

Figure 2.3: Relative change in daily (06:00 hours) water levels among BLR00 (solid line), BLR10 (dashed line) and BLR30 (dotted line) during the 2013 growing season (May 14 to September 15, 2013). Grey lines are the least-squares regression lines through respective datasets.

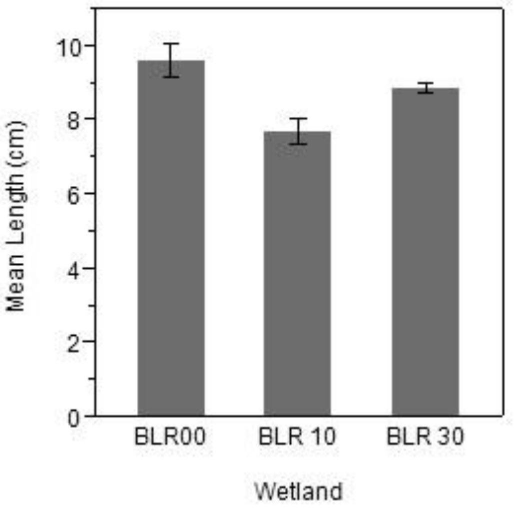

Figure 2.4: Mean (± SE) lengths of Pumpkinseed sunfish sampled at BLR00, BLR10 and BLR30 during the same week in July 2013.

General Conclusion

Coastal wetlands are naturally dynamic systems which create diverse habitats for many taxa. These chapters have shown that with reduced water-level fluctuations and a severed connection to Georgian Bay, whether by man-made diking or beaver impoundments, wetland ecosystems can become significantly altered. Impoundment structures are generally implemented to benefit bird and plant communities; however, we did not find that to be the case in our Georgian Bay studies. We found no significant difference in the marsh bird communities between Wye Marsh and Matchedash Bay and the wetland was gradually filling in and succeeding into a terrestrial ecosystem over the eight decades. Although we did not assess the waterfowl community during the migration season, our assessment of the breeding bird usage in these wetlands, including marsh obligate species, indicated no long-term benefit of the impoundment. Even in Black Rock, only a few decades behind a beaver dam led to greatly reduced vegetation diversity.

With respect to negative impacts, we have also shown that water-level fluctuations are greatly dampened in impounded wetlands when compared to hydrologically-connected coastal marshes, and that the water chemistry in these impoundments appears to follow predictable trends reflecting the loss of connection. The physical barriers negatively impacted the biotic communities by inhibiting migratory access of fish such as large sport fish (e.g. northern pike) that often require migratory access into and out of the wetlands to spawn. Overall diversity was lower in the impounded wetlands and the species were generally more tolerant of degraded conditions. The only group that appeared to benefit from the impoundment was the larval amphibian community, which was associated with a higher species diversity and abundance, and we attributed this to the change in water-quality conditions and the fish community structure.

Coastal wetlands along the eastern shoreline of Georgian Bay are relatively inaccessible and this inaccessibility would make active management costly and difficult to undertake. Our findings suggest that impounded Georgian Bay coastal wetlands that are not actively managed will not derive long-term ecosystem benefits from the impoundment. Even if Georgian Bay's coastal wetlands were easily accessible, these coastal wetlands are unlikely to be managed beyond their initial installation in the long-term due to the high costs of management.

This research demonstrated the negative implications of impoundment as a management solution to low-water levels. Although there is a risk of coastal wetlands losing water to further declines in Georgian Bay water level, leaving them hydrologically connected will allow them to benefit from higher water levels when they return. For example, due to high snow accumulation in the winter of 2013 and 2014, summer water levels in 2014 are expected to rise between 10 and 40 cm to levels just below the long-term mean. Given these concerns and the results from both chapters, we warn against using diking as an adaptation to combat the loss of wetland habitat due to declining water levels in Georgian Bay.

Future research should consider how the coastal wetlands in Georgian Bay are changing under low water-level conditions. We need to identify, through the use of bathymetric surveys and modelling, how and where coastal wetlands may be able to migrate lakewards. If predictions can be made as to where important wetland habitat may shift, adequate protection of these important wetland habitats can be put into practice.