IMPORTANCE OF HYDROLOGIC CONNECTIVITY FOR COASTAL WETLANDS TO OPEN WATER OF EASTERN GEORGIAN BAY

IMPORTANCE OF HYDROLOGIC CONNECTIVITY FOR COASTAL WETLANDS TO OPEN WATER OF EASTERN GEORGIAN BAY

By

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PREFACE

This M.Sc. Thesis is composed of two chapters that are separate manuscripts that have been created for publication in peer-reviewed journals. These have been put into context with a general introduction and conclusion. The status of both chapters is in the preliminary stages and journals are being considered for publication. Under the supervision of Pat Chow-Fraser, I, as first author in both chapters, analyzed all data and wrote both manuscripts. I also completed all field and laboratory work with assistance from field technicians, with the exception of bathymetric data for seven wetlands collected in 2009. This was provided by my supervisor, but I used the data and performed all the analyses for this thesis.

Fracz, A., & Chow-Fraser, P. Regional and site-specific impacts of declining water levels on quantity of fish habitat in coastal wetlands in eastern Georgian Bay, Lake Huron

Fracz, A., & Chow-Fraser, P**.** Hydrologic connectivity of coastal marshes and their associated ecological and chemical alterations in wetlands of eastern Georgian Bay

GENERAL ABSTRACT

Coastal wetlands are hydrologically connected to their watershed and the lake. Water levels in Georgian Bay have been at a sustained low for thirteen years and thus connectivity of wetlands to the lake is being threatened as water levels decline. Decreased connectivity has likely caused changes in ecological and chemical characteristics. Future climate change models predict further water declines and potentially increasing the number of wetlands that will be hydrologically disconnected. The over-arching goal of this thesis is to investigate the role of connectivity between the lake and coastal marshes in eastern Georgian Bay on the amount of potential fish habitat, water chemistry and larval amphibian habitat.

Bathymetric information is needed in order to estimate fish habitat and two approaches were utilized in order to collect these data. A site-specific method completed in 2009 used an intensive field survey in seven wetlands to create a digital elevation model and calculated the amount of fish habitat at 10 cm increments. A second, regional method, selected 103 sites by using a stratified random sample in 18 quaternary watersheds. In both methods, changes in water levels between 173 and 176 m asl resulted in the most drastic loss of habitat. Approximately 24% of the current fish habitat has already been lost due to low water levels.

Water chemistry in coastal marshes is influenced by hydrologic connection. In the summers of 2010 and 2011, 35 coastal marshes were sampled,

17 of which had been impounded and disconnected by a beaver dam. Beaver-impounded marshes resulted in significantly lower pH, conductivity, dissolved oxygen and sulphate concentrations, but had significantly higher soluble reactive phosphors concentrations. These conditions are indicative of the lack of connection and reduced mixing with lake water. This altered habitat was shown to support breeding area for 7 species of amphibians, the most common being green frogs and the least common being American Toads and chorus frogs.

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I would like to thank my family and friends. I would especially like to thank my parents, Karen and Steve, who have supported and encouraged me to do what makes me truly happy, even if this means playing in the dirt. And lastly my good friends Michelle and Anna.

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GENERAL INTRODUCTION

The Canadian definition of a wetland as described by Mitsch & Gosselink (2000) is "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment." Wetlands are found at the interface of aquatic and terrestrial ecosystems providing distinctive habitat and supporting a rich biodiversity of species.

Wetlands provide a variety of ecological, hydrologic, and recreational functions. They purify polluted water, provide shoreline protection against wave action and erosion, and aid in flood reduction. Ecologically, wetlands provide important habitat for a variety of species, such as amphibians (Price et al, 2007), migratory birds and waterfowl (Smith et al, 1991), fish (Jude & Pappas, 1992), flora (Croft & Chow-Fraser, 2007) and refuge for rare and endangered species (deCatanzaro & Chow-Fraser, 2010). They also contain socio-economic functions such as hunting, trapping, fishing, and attract tourism and recreation. Financially, wetlands in Canada have been estimated to have a value of billions of dollars (Government of Canada, 1991).

Wetland Classes

Not all wetlands are considered the same and there are five wetland classes in Canada. These are defined as swamps, marshes, bogs, fens or open water. Bogs experience low amount of nutrients, are peat covered and are purely influenced by precipitation. The dominate form of vegetation is *Sphagnum*. Fens have a very low gradient which leads to slow hydrologic drainage and higher mineralized water chemistry. They have a high water table and are dominated by vegetation that is composed of graminoid species and brown mosses. Swamps are nutrient rich, productive systems where slowly moving water can persist for long periods of time or seasonally. Aerated conditions can occur at the surface if the water table drops seasonally below it and trees, shrubs and forbs comprise its vegetation. Marshes are rich in nutrients, mainly wet and are periodically inundated with standing or slowly moving water. Vegetation shows a distinct zonation and is subject to water level changes seasonally or even daily if in a coastal environment. The vegetation present is graminoids, shrubs, forbs or emergent plants. Open water is standing water in the transitional phase between open water and a marsh. There is no emergent vegetation, but floating and submergent vegetation may be present (NNWG, 1997).

Coastal Wetlands and the Laurentian Great Lakes

Keough et al (1999) define coastal wetlands as the above and add that connection with the Great Lakes is a key feature distinguishing coastal wetlands from other inland wetlands. Coastal wetlands may be considered to extend to the water depth of two meters, using the historic low and high water levels or the greatest extent of wetland vegetation. Hydrologic connections may extend upstream along rivers since exchanges caused by seiches and longer-period lake-levels fluctuations influence riverine wetlands. Wetlands under substantial hydrologic influence from Great Lakes waters may be considered coastal wetlands (Keough et al, 1999).

The Laurentian Great Lakes basin is 521,830 km^2 and is composed of five lakes, which includes Lake Superior, Huron, Michigan, Erie and Ontario. It is the world's largest freshwater ecosystem with one –fifth of the world's freshwater (Mayer et al, 2004). With a total shoreline length of 17,071 km, there is great potential for coastal wetland habitat (Herdendorf, 2004). All of these lakes, with the exception of Lake Michigan, have shoreline in Canada with the total amount of coastal wetland area in the Canadian Great Lakes Basin equalling 63,706.6

hectares (GLWC, 2004). The largest amount of coastal wetland area in Canada is in Lake Huron with 16,179 hectares. It is not surprising that this lake has the greatest area given its size, geology, morphology and lesser degree of anthropogenic impact. The Great Lakes Wetland Consortium (GLWC) recognizes that the wetland area identified for Lake Huron is an underestimation as there was limited data availability at the time of the study and since then the McMaster Coastal Wetland Inventory (MCWI) has been developed which completed a thorough wetland inventory evaluation in Georgian Bay, an extension of Lake Huron (Midwood et al, in press). Therefore, the estimate of coastal wetland area for Lake Huron is greater than what is reported. Lake Ontario and Lake Erie have similar amounts of coastal wetland area containing approximately 11,777.2 ha and 11,417.5 ha while, Lake Superior has the smallest of all the Canadian Lakes with a mere 2235.8 ha (GLWC, 2004).

Classification of Great Lakes Coastal Wetlands and Hydrologic Influences

Coastal wetlands can be further classified based on geomorphic position, dominant hydrologic sources and current hydrologic connectivity to the lake. Albert et al (2005) defines three hydrogeomorphic classification systems which included lacustrine, riverine and barrier protected systems. This classification scheme builds on that which was developed by Keough et al (1999), but divides the wetlands into further subdivisions based on physical and florisitic differences. The broadest categories are defined below.

Lacustrine coastal wetlands are controlled directly by water with its connecting lake, water levels, nearshore currents and ice scour. In comparison, riverine wetlands occur along or within rivers that flow into the Great Lakes. The river is the dominant factor influencing hydrology, but it is also influenced by lake processes as the lake water may flood back into the wetland. Riverine wetlands tend to be somewhat sheltered from wave action as the bars and river channel act as protection. The third system is barrier protected wetlands. These wetlands are separated by a barrier feature and may be isolated or connected to the lake through a channel crossing the barrier. If this is the case, lake water levels will have a dominant effect. If isolated, groundwater and surface water would be the more dominant source of input (Albert et al, 2005).

Recently, Morrice et al (2011) have tested the assumptions of hydrogeomorphic classification by identifying the sources of hydrology and determined the relative importance of each source to a coastal wetland to develop a hydrology based classification scheme. The three classes were compared to Albert et al (2005) classification and included tributary dominated, mixed influence and lake dominated. They found that the hydrogeomorphic classification was adequate at separating wetland classes, but could be improved upon with if it could account for wetland hydrodynamics. They found that riverine wetlands tended to have the largest watershed and were dominantly influenced by tributaries. Lacustrine wetlands tended to have larger seiche influences and small watersheds. Lastly, barrier protected wetlands tended to have minor seiche influences and small watersheds (Morrice et al, 2011).

Threats to Wetlands

Over the past 200 years coastal wetlands have been destroyed due to the expansion of urbanization and the conversion of land into agriculture (Mayer et al, 2004). More than two-thirds of Great Lakes wetlands have already been lost and where wetland loss has not occurred, there has often been degradation (Mayer et al, 2004). A considerable number of coastal wetlands within the Laurentian Great Lakes basin have been shown to be impacted, especially in the lower Great Lakes of Erie and Ontario (Cvetkovic & Chow-Fraser, 2011). Great Lakes coastal wetlands water chemistry on both the Canadian and American coastlines have been found to be strongly and positively related to the amount of agricultural practices present in the surrounding watersheds (Crosbie & Chow-Fraser, 1999; Morrice et al, 2008). Coastal zones are also among the most populated regions with a total of 9.2 million people living near them in Canada (Mayer et al, 2004). Urbanization can lead to increased sediment and nutrient loading in coastal watersheds causing degradation in water quality (Chow-Fraser, 2006; Morrice et al, 2008). Pollution from industrial waste treatment facilities and carp turbation has also been found to cause water quality impairment (Chow-Fraser, 2006).

Climate change also poses a threat to coastal wetlands. The strongest climate-wetland link for Great Lakes coastal wetlands is through changes in water level (Mortsch, 1998). Wetlands are maintained by natural fluctuations. The timing, amplitude and duration of water levels as well as the mean and seasonal cycles are expected to be altered with climate change (Mortsch, 1998). Future water level fluctuations are also expected to be outside of the historical range (Sellinger et al, 2008). These alterations would undoubtedly cause shifts and changes to species diversity and assemblages.

Wetland Conservation

Given the many important services that wetlands provide, protection of these ecosystems is of critical importance. Canada currently does not have specific wetland legislation in place, but can receive protection through a variety of provincial and federal acts. For example, the *Fisheries Act* states that fish habitat must be protected from alteration, destruction and conversion. Over 80 species of fish use coastal wetland habitat at some point in their life cycle (Jude & Pappas, 1992), and thus coastal wetlands are indirectly protected. Other federal acts, such as the *Canadian Wildlife Act*, *Migratory Bird Convention Act* and *Species at Risk Act* can provide additional indirect wetland protection (Environment Canada, 2012). Provincial acts such as the *Planning and*

Development Act and *Conservation Land Act* can offer additional protection within Ontario (Environmental Canada, 2012).

Canada has also demonstrated its commitment to wetland conservation through various agreements, treaties and action plans, both nationally and within Canada. Canada is a part of the intergovernmental treaty RAMSAR where 37 sites have been designated as wetlands of international importance. Currently, the Great Lakes Water Quality Agreement (GLWQA) is being amended from its last draft in 1987. This is a bi-national agreement which in previous versions has identified that significant wetland areas that are threatened by development, dredging or pollution be identified, preserved and rehabilitated. A strictly Canadian initiative is the Great Lakes Wetlands Conservation Action Plan (GLWACAP) which was started in 1994 with an objective to conserve and rehabilitate wetlands through existing programs.

A common theme in wetland policy is the identification of wetlands that are of a significant value. The Ontario Ministry of Natural Resources (OMNR) has an evaluation system in southern and northern Ontario used to identify wetlands at a provincial scale (OWES; OMNR, 1993). It was developed to aid the planning process based on the value of wetlands biological, hydrologic, and social factors. Therefore, its designation and evaluation has important implications for the types of land uses that can occur in the vicinity of a wetland and thus the quality of habitat that it provides.

Coastal Wetlands of Georgian Bay

Georgian Bay is located within Lake Huron of the Laurentian Great Lakes. It lies on the boundary of two unique geologic settings; one of granitic Precambrian shield rock at the north and limestone bedrock at the south. Its 4,500 km of highly convoluted shoreline is conductive to the formation of coastal wetlands. The MCWI has identified 12629 distinct wetland units which comprised of 17,350 hectares of low and high marsh as well as upstream habitat area (Midwood et al, in press). Therefore, this area contains a large amount of wetland habitat.

Not only are there are large amount of wetlands in Georgian Bay, but the quality of the habitat is also very high. Wetlands have been determined to have very high water quality comparatively to other regions within the Great Lakes basin (Chow-Fraser, 2006; Cvetkovic & Chow-Fraser, 2011). This is due to the relatively little amounts of human disturbance present. Development is this area is mainly recreational, made of cottages and roads. Despite the low degree of human influence, there has been a strong correlation between water chemistry impairment in coastal marshes and the amount of roads within its watershed (deCatanzaro et al, 2009).

The quality, quantity and uniqueness of Georgian Bay is why it is designated a world biosphere reserve by UNESCO and in Georgian Bay, wetlands are classified as embayments or fringing which would correspond with Albert et al (2005) designation of lacustrine coastal wetlands. The main source of water is highly influenced by lake processes. Fringing wetlands would be exposed to high amount of wave energy and experience little to no protection. This often results in a narrow fringe of emergent vegetation. Embayment wetlands may experience moderate amounts of wave energy and contain a greater diversity of vegetation. The shape of these wetlands highly influences the degree of connection with open water and the amounts of wave energy experienced.

Hydrologic Connectivity

In landscape ecology, connectivity is the degree to which a landscape facilitates or impedes movement of organisms among resources patches.

Hydrologic connectivity is defined as "water mediated transfer of matter, energy and/or organisms within or between elements of the hydrologic cycle". Current knowledge on hydrologic connectivity is lacking due to the complexity of water movement and lack of understanding of human influences on it. A lack of connectivity will inevitably lead to losses in ecological integrity, declines in aquatic biodiversity and major environmental consequences (Pringle, 2003).

A coastal wetland is hydrologically connected to both its watershed and the lake and thus, the hydrologic interaction between the wetland, watershed and lake determines sources of water, residence times and nutrient loading and cycling (Morrice et al, 2011). A hydrologic disconnection of a wetland from one of these sources has effects on water chemistry and habitat availability. The connection most threatened for coastal wetlands is that to the lake, the consequences which are largely unknown.

Thesis objectives

Water level changes, influenced by both lake-level regulation and climate change, has emerged as a major threat to wetlands, and understanding the response of wetlands to reduced hydrologic connections will have increasingly important implications. The first objective of this thesis is to quantify the amount of fish habitat that would be inaccessible to fish with further declines in water levels. I will also estimate the amount of habitat that has already been lost and estimate the amount that is predicted to be lost based on climate change forecasts. By quantifying the amount of wetland habitat that is dependent on hydrologic connection to Georgian Bay, I will provide essential information to aid decisions regarding the ecological effects of water level regulations.

My second objective is to examine the effects of decreased hydrologic connection on coastal wetland water chemistry and larval amphibian breeding habitat. The decreased hydrologic connection is due to beaver impoundments rather than to low water levels. Besides examining changes in water chemistry, I will also examine the types of larval amphibians that are supported in these beaver impoundments. This is an area that has received very little attention to date and I hope my results will give some insight into how natural water chemistry will vary when wetlands become hydrologically disconnected from Georgian Bay.

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Chapter 1:

Regional and site-specific impacts of declining water levels on quantity of fish habitat in coastal wetlands in eastern Georgian Bay, Lake Huron

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Abstract

Coastal wetlands of eastern Georgian Bay (GB) are known to support critical spawning and nursery habitat for the fish community of Lake Huron. Due to the convoluted shorelines of granitic rock where coastal wetlands have formed, a lowering of lake levels below the wetlands' outlet can cause the wetland to become hydrologically disconnected (stranded) from the bay, something that had been observed in 2009 when water levels were at 176.0 m above sea level (asl). Forecasts from climate-change models indicate that lake levels may continue to drop by up to 3 meters, and since loss of connectivity will restrict access of migratory fish from GB into spawning and nursery habitat, there is an urgent need to know how much more critical fish habitat might be lost. Amount of fish habitat can be easily estimated in GIS for a wetland at a given water level as long as there is a suitable digital elevation model (DEM). Unfortunately, bathymetric information does not currently exist for the > 3700 coastal wetlands along the 17,000 km of the GB shoreline. Here, we report the results of two approaches used to address the problem. For the first site-specific method (SM), we collected bathymetric data from seven sites through an intensive field survey and created digital elevation models to calculate the amount of fish habitat at 10-cm decrements of 176. 0 m asl. For the second regional method (RM), we performed a stratified random sampling procedure to identify 103 sites (> 2ha) in 18 quaternary watersheds throughout eastern Georgian Bay between Severn Sound and Key River. We measured the maximum depth of the wetland outlets and

estimated the cumulative area of wetland habitat that would become hydrologically disconnected as water levels decline beyond the maximum depth of outlets, which ranged from 162.94 to 176.06 m asl. For both methods, we found that amount of lost habitat was extremely sensitive to changes in water level between 173 and 176 m asl. Based on the seven sites with DEM, we estimate that 24% of submerged habitat has already been lost in wetlands when water levels dropped from 177.5 to 176.11 m asl. We estimate that if water levels were to drop to a predicted extreme of 175.56 m asl, 13% of the total number of coastal wetlands and 6% of total area in coastal wetlands would disappear in addition to what has already been lost. Result of this study will inform management decisions that pertain to water level regulation the Great Lakes basin.

Introduction

Coastal wetlands provide essential habitat for more than 80 fish species of the Laurentian Great Lakes that use wetlands to feed, spawn and as nursery habitat (Jude & Pappas, 1992; Wei et al, 2004). Of these, approximately 50 fishes are residents and therefore solely dependent on wetlands. The remaining fish are migratory, moving into and out of wetlands during their lifespan (Jude & Pappas, 1992). Use of these wetlands by migratory fish species varies according to season and time of day; therefore, there must be a consistent degree of connection to facilitate fish movement (Jude & Pappas, 1992; Bouvier et al, 2009).

The degree of connection of wetlands to open water has also been shown to influence both species richness and piscivore richness (Jude & Pappas, 1992; Johnson et al, 1997; Bouvier et al, 2009). Snodgrass et al (1996) suggests that frequently disturbed systems depend more on colonization rates to determine richness while environmental characteristics play a more important role when disturbances are rare. They found that the variation in frequency and duration of connectivity between wetlands and permanent aquatic habitats are responsible for the structure of fish assemblages (Snodgrass et al, 1996). Similarly, Baber et al (2002) found that fish assemblage structure in temporary wetlands was influenced dominantly by connectivity to permanent wetlands and secondly by environmental factors. Connectivity has been shown to play a larger role than environmental characteristics in fish richness and abundance even in permanent wetlands that are not frequently disturbed such as the permanent lower Great

Lakes wetlands (Bouvier et al, 2009). Likewise, Johnson et al (1997) found that barriers to fish movement such as dykes reduced fish species richness. This stresses the importance of hydrologic connection on fish species richness and diversity in wetlands that are both frequently and permanently connected.

Hydrologic connectivity of coastal wetlands to open water of a Great Lake is entirely determined by the water level of the lake in question. Water levels in the Laurentian Great Lakes fluctuate naturally across seasonal, annual and multi-year cycles (Quinn, 2002; Sellinger et al, 2008; Hanranhan et al, 2010; Figure 1.1). It is well known that variation in lake elevation will flood different amounts of wetland habitats depending on local site geomorphology; within a site, high water levels tend to produce more submergent vegetation and open water in wetlands, while low water levels produce more wet meadows and emergent vegetation (Keddy & Reznicek, 1986; Chow-Fraser, 2005; Hudon et al, 2006; Wilcox & Nichols 2008). Natural fluctuations in water level support species diversity and alterations in these cycles may cause loss of wetland extent, diversity and resilience to disturbance (Keough et al, 1999; Wilcox & Nichols, 2008). Water levels in Georgian Bay, Lake Huron started to decline in 1999 and have continued to remain at low levels (Assel et al, 2004; Sellinger et al, 2008; Canadian Hydrographic Services, 2012), reducing the extent of hydrologic connectivity to coastal wetlands that serve as critical spawning and nursery habitat to the Lake Huron fish community. In these relatively pristine wetlands, pressure from human development is relatively low (deCatanzaro et al, 2009), and therefore sustained low water levels appear to be the major threat (Midwood et al, in press). Except for the southeastern portion, Georgian Bay wetlands occur on granitic rock of the Canadian Shield (Midwood et al, in press). Declines in water level will affect the distribution and area of wetlands, as the granitic Precambrian Shield rock will prevent erosion at the embayment outlet, often preventing their expansion lake-ward when water levels decline.

Predictions for future water levels in Georgian Bay vary depending on global circulation models (GCM) and emission scenarios. Studies by Lofgren et al (2002) and Angel and Kunkel (2010) predict both water level increases and decreases while other studies have predicted only declines (see Mortsh & Quinn, 1996; Hanrahan et al, 2010). The majority of these predictions suggest that future water levels in Georgian Bay will be lower than current levels, with some extreme projections suggesting declines by as much as 3 m by 2080 (Angel & Kunkel, 2010). Even though there are disagreements concerning the extent of the extremes, it is generally agreed that future water fluctuations will vary outside the observed historical range (Sellinger et al, 2008). This increases the urgency for managers to understand the ecological consequences of water-level extremes so that appropriate changes can be incorporated into water-level regulation plans (Wilcox & Xie, 2007).

The International Joint Commission is in the process of conducting the International Upper Great Lakes Study (IUGLS) to assess the ecological implications of changing water levels in the Upper Great Lakes (with emphasis on

Lakes Huron-Michigan and Lake Superior). The current focus is to review operations of the current structure controlling Lake Superior outflow to assess whether changes should be made to its regulation plan. The objective of our study was to model the quantity and proportion of area of fish habitat in coastal wetlands corresponding to proposed water-level scenarios, to aid evaluation of ecological response of different regulation plans. Regional and site-specific methods were used to achieve this objective. The site-specific method was more labour intensive and involved complete modeling of an individual wetland with a digital elevation model (DEM), while the regional method was a rapid assessment approach which allowed sampling of a large number of wetlands and modeled the total wetland as a single entity. This is the first study to forecast how quantity of fish habitat in coastal wetlands may change with water level decline at the scale of the entire eastern Georgian Bay region and will provide guidance for managers in other parts of the Laurentian Great Lakes or other large lakes to undertake similar region-wide assessment of declining water levels related to global climate change.

Method

Study Area

Wetlands in the eastern Georgian Bay area are plentiful and relatively pristine, as they have been influenced minimally by wide-scale landscape alterations in comparison to other areas of the Great Lakes (Snell 1987; Cvetkovic

& Chow-Fraser, 2011). Many of these coastal wetlands have very high water quality and are naturally oligotrophic due to the low nutrient input and connection to Georgian Bay (Chow-Fraser, 2006; deCatanzaro & Chow-Fraser, 2011). There is a large quantity of wetlands with an approximate number of 12 629 distinct wetland units covering eastern and northern Georgian Bay (Midwood et al, in press). The dominant floating vegetation types are *Nymphaea, Nuphar, Brasenia,* and *Zizania* species, submerged species of *Potamogeton* and *Myriophyllum,* and emergent species of *Schoenoplectus* and *Eleocharis* (Midwood & Chow-Fraser, 2010). The soils are thin, sandy, acidic, and patchy (Weiler, 1988).

Site Specific Model Site Selection

Seven sites in eastern Georgian Bay were selected in 2009 for the site-specific study (square symbols in Figure 1.2; Table 1.1). These sites were chosen as some bathymetric data had already been collected and permission to access these sites for sampling purposes had already been acquired. Based on the extensive sampling that had been carried out by the Chow-Fraser lab (see Cvetkovic and Chow-Fraser 2011), we determined that these sites are representative of coastal wetlands in eastern Georgian Bay. These sites covered a range of sizes (1.54 ha to 43.03 ha), and some are protected embayments while others are more exposed; however, none of them can be considered heavily impacted by human activities.

Regional Model Site Selection

According to the McMaster Coastal Wetland Inventory (MCWI; Midwood et al, in press) there are 6 500 ha of coastal wetlands in eastern Georgian Bay. Of these, only 519 are > 2 ha in size, which is the size criterion used by the Ontario Ministry of Natural Resources (OMNR) to determine if wetlands qualify for assessment in the Ontario Wetland Evaluation System (OMNR, 1993). For logistical reasons, we had to choose a subsample of these wetlands to conduct the regional study. We carried out a stratified random selection to ensure that we chose wetlands according to their availability in the 18 quaternary watersheds. This resulted in a subset of 103 wetlands for the regional study.

Determining the Depth of Wetland Opening and Elevation

For a wetland to be hydrologically connected to Georgian Bay, we assumed that the water level in the Great Lake had to be at least higher than the elevation of the deepest spot along the wetland opening. In many instances, the wetland opening is governed by a rock sill which cannot be eroded. Thus, when water levels in Georgian Bay drop below the lowest elevation of the sill, the coastal wetland is hydrologically disconnected or stranded. Sill depth was measured in different ways in both the site-specific and regional studies. In the site-specific study, detailed bathymetric information was collected to create a DEM of the entire wetland to model above and below the existing water level. For the regional method, coarse bathymetric information was collected along several transects to determine the depth at the outlet of the wetland. This rapid assessment method was meant to determine the lake elevation that would result in the entire wetland being inaccessible to fish, whereas the site-specific method would relate the extent of area lost within a wetland to declines in water level. Depth information for the site-specific method was collected from July to September 2009, while information for the regional method was collected from May to September 2010.

Bathymetric information was collected in three ways. First, we used a differential GPS (dGPS) (Magellan ProMark 3®) equipment with a base and roving unit to collect digital data from just below the shoreline to an elevation of about 3 m above the shoreline. We used a stadia rod and GPS unit (herein referred to as Mobile GPS) (Magellan ProMark 3® unit) to measure depth manually at depths up to 1.2 m. We also used a boat-mounted sonar depth sounder (Lowrance® sonar unit) to collect depths in water deeper than 1.2 m deep. For areas too deep for the dGPS, where the substrate was highly organic, difficult to walk in, or where there was very dense aquatic vegetation, we used the mobile GPS with a canoe. Accuracies varied among the three sources; the dGPS was the most accurate with sub-meter horizontal accuracy and centimeter vertical accuracy. The mobile GPS contained 1-m accuracy for location and 0.5 cm accuracy for depth. The boat sonar was the least accurate method in terms of horizontal accuracy (3-m) but had the capability of centimetre depth accuracy.

Parry Sound water level data (Canadian Hydrographic Service, 2012) was used to convert depth information to elevations above sea level. In all cases, we recorded the exact time when data was collected to relate it back to time-specific water level data recorded in Parry Sound.

Data Analysis

We used ArcMap 9.2 (Redlands, CA, USA) to develop DEMs for each of the seven wetlands sampled for the site-specific method. IKONOS imagery acquired in July 2002 and 2008 were used to help interpret the field data, especially along the shoreline where we had missing field information. We used the Topo to Raster function from the Spatial Analyst extension to create a hydrologically consistent DEM with an inverse distance weighting interpolation (deSmith et al, 2007). The DEM was modeled at 10 cm increments to determine the surface area lost for each water level elevation.

In the regional method, elevation data were visually inspected to determine the lowest elevation at wetland outlet. The lowest value indicates the elevation below which the wetland would become completely disconnected from Georgian Bay and all area within the wetland would be lost fish habitat. Wetlands were also classified as embayments or fringing based on the shape and degree of connectivity to Georgian Bay by visual inspection.

We used GenStats (version 14.2) software to derive the best-fit inverse logistical model for our data. Percent average surface area inaccessible within an individual wetland was calculated by determining the best-fit inverse logistic equation through data for the seven wetlands surveyed in 2009 (Equation a, Table

1.2). The percentage of total number of coastal wetlands inaccessible was calculated based on the number of wetlands that would cease to be inundated as a function of declining water levels (Equation b, Table 1.2). The percentage of total surface inaccessible was calculated based on the entire area of wetlands that would cease to be inundated as a function of declining water levels (Equation c, Table 1.2). From this percentage we can estimated the area associated with all wetlands that would no longer be accessible to fish. Matchedash Bay was excluded from this analysis because it is unusually large and is not representative of the other smaller wetlands (all <123 ha; Table 1.1).

Application of Models to Climate Change Predictions

We calculated projected water levels associated with different general circulation models and applied these to our models to determine how quantity of wetland fish habitat may be affected by global climate change. Studies by Mortsch and Quinn (1996), Lofgren et al (2002), and Angel and Kunkel (2010) were used to model water level declines; elevations were calculated based on the base level used in the study (Table 1.3). Mortsch and Quinn (1996) examined four GCMs in $2xCO^2$ emissions scenarios and predicted water level declines from 1 m to 2.48 m. Lofgren et al (2002) used two GCMs with transient CO^2 concentrations and projected increases of 0.35m to declines of 1.38 m by 2090. The range of predictions by Angel and Kunkel (2010) was quite large (3 m declines to increases of 1.5m) where they examined 23 GCMs with three transient emission scenarios. In this study, we only included projections that indicated a decline in water levels (i.e. most of the models) because we were only able to accurately depict how fish habitat would change at elevations above 176.0 m for seven sites.

Results

Description of Coastal Wetlands

The deepest point at the outlet for each of the wetlands ranged from 162.94 to 176.06 m (asl) and had a mean of 173.55 m (asl) (Table 1.1). Elevation data for the seven wetlands ranged from 170.7 to 181.8 m (asl). The size of the wetlands for both methods ranged from 0.12 to 122.82 ha. Seventy-nine percent of wetlands in the regional study were classified as embayments while the remaining 21 % was classified at fringing.

Average Percent Area Lost within an Individual Wetland

The relationship between percent loss and lake elevation for all seven wetlands in the site-specific study followed an inverse logistic curve (Figure 1.5). The greatest percent change occurred between 176.5 and 173.5 m (asl), indicating that water-level changes within this range would have the greatest impact on loss of wetland surface area and by implication loss of fish habitat. The average percentage of surface area that has already been lost within wetlands is approximately 24% (based on water level of 176.11 m asl; yearly mean 2010; Canadian Hydrographic Services). In all but one case, the amount of wetland area would level off once water levels approached the historic high of 177.5 m asl

(Canadian Hydrographic Services 2012); for North Bay 5, however, wetland area would continue to increase above 177.5 m asl since it is associated with a more gradual upland slope. Therefore, in general, aquatic area of wetland for the 7 sites is restricted both lake-ward and shoreward.

Percentage of the Total Number and Total Surface Area of Coastal Wetlands Lost and Inaccessible to Fish

Outlet elevations for the 103 wetlands in the regional study ranged from 162.94 to 176.06 m (asl) (Table 1.1). When we plotted outlet elevations against water levels (Figure 1.6), we found that the data was fitted well with an inverse logistic regression equation (Table 1.2). Similarly, total surface area that was deemed to be inaccessible to fish at the various water levels was also subjected to an inverse logistic regression analysis (Figure 1.7; Table 1.2). Both graphs showed that the greatest percentage of change occurred between 176 m to 173 m (asl).

Application of Models to Climate Change Scenarios

The General Circulation Models project a drop in water level that ranges from 0.55 m to 2.48 m below base case (Table 1.3). Associated changes in elevation occur from a water level decline of 174.92m (asl) to approximately 176.2 m (asl). The greatest declines are predicted by Mortcsh & Quinn (1996) with $2xCO^2$ models and for models that approximate conditions in the period 2080 and 2090; therefore, these models predict the greatest loss in fish habitat. In order to compare projections in a $2xCO^2$ environment to new models, Lofgren et al (2002) suggest that $2xCO^2$ best approximates a 2050 environment and thus predictions for water levels for 2050 range from 175.95 m (asl) to 174.92 m (asl). Water levels declines within this range show that as many as 50.8 % of total number of coastal wetlands will be inaccessible and that 48% of the total wetland surface area will be inaccessible to fish. This results in 3172 ha of area lost. Within an individual wetland, on average, 30 to 95 % of area will be inadequate for fish habitat based on projections for 2050. Projections near the end of the century show declines of approximately 1 meter or more with up to 687 hectares of wetland lost as well as large losses within wetlands.

Discussion

This is one of the first regional studies to examine the impact of water-level declines on amount of fish habitat in coastal wetlands of Lake Huron. We have used two different approaches to examine this and found similar results. In both cases, a drop in water level between 176.5 and 173 m (asl) will have a profound effect on the amount of fish habitat in coastal wetlands of eastern Georgian Bay. In 2009, at the time of the site-specific study with a mean annual water level of approximately 176 m (asl) for Lake Huron, there had already been an average loss of 24 % of fish habitat within individual wetlands. Predictions from General Circulation Models indicate that low water levels will continue to be a problem and if this occurs, there will be substantial loss of wetland fish habitat due to disconnection with Georgian Bay.

Data collection in the site-specific method was labour intensive; on average, one week was required per wetland with a dedicated crew of two people and a boat. The end result was a DEM with very good coverage of the wetland's elevation that allowed for detailed modeling above and below the water level at the time of sampling. The regional method, on the other hand, only yielded coarse bathymetric information but was able to cover a large region and did not require as much effort per wetland. Transects could be completed in less than an hour in most cases, since the objective was to find the lowest elevation at the wetland outlet. Due to the rapid assessment, we were able to sample more than a 100 wetlands in a statistically valid fashion. The trade-off was that the data could only be analyzed in a limited number of ways because no DEM could be created from the data. The regional method also gave a conservative estimate of the amount of surface area lost, as the area is only considered gone when the entire marsh is disconnected from Georgian Bay, while the site-specific method showed how fish habitat within wetlands would shrink as water levels declined.

The Levels Reference Study (1993) stated that initial action should be taken to change incremental flow from Lake Superior to Lake Huron when water levels reach 176.20 m (asl). We are currently at the tipping point of the state of coastal wetlands in eastern Georgian Bay and have been for thirteen years. Lake Michigan-Huron has had a small range of fluctuations of 0.38 m between 1999 and 2010 around a sustained low (Canadian Hydrographic Services, 2012). Compared with the other Great Lakes, Lake Superior`s range was 0.44 m, while that of Lakes Erie and Ontario are 0.34 and 0.32 m, respectively during the same time period (Canadian Hydrographic Services, 2012). The lower lakes (Ontario and Erie), however, have had these ranges deviate around their all-time annual mean (from 1908 to 2011). It is also important to note that Lake Ontario is regulated which has caused this reduction and overall range stabilization around the long term mean (Hudson et al, 2006). Current climate change projections predict further declines most severely in Lake Michigan-Huron (Angel & Kunkel, 2010; Table 1.3) creating urgency for management plans to protect the coastal wetlands and the habitat that it provides without deviating from the systems natural fluctuations.

Future Water Levels Based on Climate Change Predictions

Regardless of the methods used to forecast the effects of increased carbon dioxide in the atmosphere, all General Circulation Models predict lower water levels in Lake Michigan-Huron. The result of these lower water levels is a net loss in fish habitat. In some scenarios, we were unable to measure the predicted loss because the equations were modeled after a sampling period completed during a period of low water levels (176.11 m (asl), Canadian Hydrographic Services). In these cases, the site-specific model demonstrated how decreases in water level will continue to lead to habitat loss and alteration within the seven wetlands.

The 5th percentile results of emission scenarios were used from Angel and Kunkel (2010) as they showed the most extreme declines (Table 1.3). Projections occurring in low percentiles should be viewed as equally likely to occur since there is no objective method for us to evaluate the relative accuracy of a GCM; therefore all should be viewed as possible future outcomes (Angel & Kunkel, 2010). Caution should be taken when comparing $2xCO^2$ projections to newer models as these models are based on an equilibrium response and negative forcings are not included (Lofgren et al, 2002; Mortsch et al, 2000). Significant advancements have been made in models since 2000, (Mortsch et al, 2000) but the longer the predicted timespan, the greater the variability (see Angel & Kunkel, 2010).

Water Levels and the Implication on Fish Habitat

This study models fish habitat in terms of surface area in hydrologically connected wetlands of a large lake. This could be modeled with a DEM to show how area within a particular wetland would change when flooded, or it could be done with an entire shoreline within a region based on the point of connection with the lake. In reality, fish habitat and utilization depends on many variables that we were unable to include in this study. We know that factors such as the type of macrophytes found in the wetland, the type of landscape features, water quality and fish community structure are all important determinants of whether or not fish would use the wetland as habitat (Randall et al; 1996; Wei et al, 2004;

Trebitz et al, 2007; Midwood & Chow-Fraser, 2012). Therefore, our estimates should be considered as the maximum potential fish habitat in coastal wetlands.

Very few studies have attempted to document the relationship between fish use of wetlands and habitat variation associated with short- or long-term changes in water level change (Keough et al, 1999). There exists, however, an association between plant structure and fish communities (Jude & Pappas, 1992; Randall et al, 1996; Trebitz et al, 2007; Cvetokovic et al, 2010) and there have been various studies that examine water- level fluctuations on the plant community (Quinlain & Mulamoottil, 1987; Keddy & Rezicek, 1986; Wilcox & Nichols, 2008). During low water level periods, emergent and wetland vegetation colonize on new substrate, while open water and submerged aquatic area is reduced or shifted lakeward (Keddy & Rezicek, 1986; Wilcox, 2004; Midwood, 2012). Sustained lows result in structural wetland plant changes as there is a loss of species that regenerate in high periods and dominance of emergent and meadow vegetation (Keddy & Rezicek, 1986; Midwood, 2012).

Sustained water levels lows in Georgian Bay have been found to reduce habitat complexity in coastal wetlands, resulting in an overall loss of fish habitat (Trebitz et al, 2007; Midwood & Chow-Fraser, 2012). Ideal fish habitat occurs in heterogeneous plant structure of submerged and floating vegetation or along edges of aquatic vegetation and open water, where resources are high and there is protection from predators (Mortsch, 1998; Trebitz et al, 2007; Midwood & Chow-Fraser, 2012). Midwood & Chow-Fraser (2012) found a significant increase in the amount of meadow and high density floating vegetation while low density floating vegetation decreased significantly. As a result of habitat becoming more homogenous with increased densities, fish species richness decreased significantly and tadpole madtoms (*Noturus gyrinus*), blackchin shiners (*Notropis heterodon*), black crappie (*Pomoxis nigromaculatus*), and the Cyprinidae family all decreased significantly in the proportion of catch (Midwood & Chow-Fraser, 2012). The response of SAV to declines in water levels can be variable, but there is often a loss with potential declines as great at 76.7 % (Midwood, 2012). This suggests that the current estimate of fish habitat loss based on our models is under-representing potential fish habitat given the shift from low density floating vegetation to high density floating; a less desirable habitat for some and loss of SAV.

An estimated 79% of wetlands in Georgian Bay occur within an embayment, which is more likely to become disconnected to the bay as water levels drop. Aquatic connectivity plays an important role in structuring wetland fish assemblages (Johnson et al, 1997; Baber et al, 2002; Bouvier et al, 2009) and if water levels continue to decline, connection to these embayment wetlands will be limited and species richness is likely to decline. This underscores the inability of our regional model to accurately estimate loss of fish habitat since fish species richness will decline before there is complete disconnection; therefore the effect of habitat loss may be greater than what is predicted in this study.

Fringing wetlands (21% of those in this study) have a high degree of connectivity with the Bay and are less likely to become stranded, but are more highly exposed to wave and wind action. Exposure is a determinant of macrophyte growth as it limits the establishment and dispersal of seedlings (Keddy, 1982, 1984). SAV can be prevented from establishing in these areas, and this may influence the quality of fish habitat (Keddy, 1982, 1984; Wei & Chow-Fraser, 2006). These more exposed sites are likely to favour species that are tolerant of this type of disturbance such as rock bass (*Ambloplites ruperstris*), while intolerant species such as the pumpkinseed (*Lepomis gibbosus*) will be selected against (Cvetokovic, 2008). In this way, even if fringing wetlands remain connected to the Bay, lower water levels may bring an overall change in the community structure and species richness in wetlands.

Water levels lows, invaders and effect on fish habitat

Declines and sustained lows create a disturbance that makes the environment more susceptible to invaders (Davis et al, 2000). Low water levels provide ideal habitat for expansion of emergent species as it exposes shoreline sediments, allowing the species to regenerate from buried seeds (Keddy & Rezicek, 1986; Wei & Chow-Fraser, 2006; Wilcox & Nichols, 2008). Water level declines have been proposed as a cause for rapid expansion of *Phragmites australis* (hereafter *Phragmites)* in Green Bay, Lake Michigan (Tulbure et al, 2007; Tulbure & Johnston, 2010), Lake Erie (Wilcox et al, 2003), Southern Lake Huron and Saginaw Bay (Uzarski et al, 2009). Once *Phragmites* invades, it spreads rapidly through rhizomes and seeds (Bart & Hartman, 2003) and forms mono-dominant cultures in wetlands that displace native plants (Chambers et al, 1999). Tulbure et al (2007) found that an open lagoon in Green Bay became dominated by invasives over 3 years (from survey completed in 2001 and 2004) when water levels declined and reduced connectivity with the bay. *Phragmites* was already present in this study at the initial time of sampling, but was only present at one plot (out of 11) with 3% cover (Tulbure et al, 2007). While the distribution of *Phragmites* in eastern Georgian Bay is currently limited, it has been spotted in parts of Severn Sound and Tadenac Bay, Georgian Bay (A.Fracz, Pers. Obs.), and may expand with the further predicted decrease in water levels. If this happens, fish habitat will be further degraded since they create a dense homogenous emergent community that is impenetrable to fish.

Creation of New Wetlands with Lower Water Levels

As water levels decline, it may be possible for wetland vegetation to shift lake-ward as new habitat is available; however, this would be highly dependent on wetland geomorphology (Albert et al, 2005; Midwood, 2012). Due to the irregular topography of the Canadian Shield, coastal wetlands are less likely to have suitable sites for downslope colonization as compared to other regions (Mortsh, 1998). In order to determine to what degree and extent new wetlands would form, bathymetry data from Light Detection and Ranging (LiDAR) technology would be needed to create regional DEMS for the entire Georgian Bay coastal line to be able to quantify wetland movement lake-ward (Quinlan & Mulamoottil, 1987;

Wozencraft & Millar, 2005). If migrations are possible, wetlands would likely become more exposed to wind and wave action as shown in the conceptual response of SAV habitat by Midwood (Figure 6, 2012). As previously stated, exposure can limit growth of macrophytes (Keddy and Reznicek 1986) and resulting habitat may become less suitable for certain fish species (Trebitz et al, 2007; Cvetokovic, 2008). Randall et al (1996) have documented percent cover of macrophytes and fish species to be significantly negatively correlated with maximum exposure. Changes to a more exposed environment will undoubtedly cause changes to a more homogenous structure in the macrophyte and fish community.

We have shown by using a regional method that water level declines below 176 m will drastically decrease the number of wetlands and the total area that fish can utilize as spawning and nursery habitat (Jude and Pappas, 1992). This can be applied to all of eastern Georgian Bay given that the wetlands were selected at random in proportion to their availability in 18 quaternary watersheds. Furthermore, the site-specific method tells us that an average of 24 % of surface area within the coastal wetlands has already been lost due to a drop in water level of 176.11 m (asl) (year mean 2010; Canadian Hydrographic Services). A decline of 2.48 m from base case predicted by Mortsch & Quinn (1996) would be accompanied by a loss of 3 172 ha of wetland which is 50% of eastern Georgian Bay's coastal wetlands.

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Tables

Table 1.1: General descriptions of study sites

Wetland Name	Latitude	Longitude	Outlet Depth (m asl)	Area (ha)	Shape	Quaternary Watershed	Watershed Code
2009 Wetlands							
Alexander Bay	45.05279	-80.00201	-	4.46	Embayment	Moon-Musquash Islands	2EB-01
Coffin Rock	45.04792	-79.98725	-	11.70	Fringing	Moon-Musquash Islands	2EB-01
Miners Creek	45.06246	-79.94721	-	3.46	Embayment	Moon-Musquash River	2EB-02
North Bay 1	44.89863	-79.79398	-	5.46	Embayment	Severn River	2EC-17
North Bay 5	44.88238	-79.80240	-	1.54	Embayment	Severn River	2EC-17
Oak Bay	44.79867	-79.73715	-	43.03	Fringing	Severn River	2EC-17
Treasure Bay	44.87158	-79.85857	-	12.72	Embayment	Beausoleil Severn Islands	2EC-18
2010 Wetlands							
Clifton Bay	44.95673	-79.86122	176.06	2.36	Embayment	Severn River	2EC-17
Black Rock (D)	45.04500	-79.96900	175.85	2.75	Embayment	Moon-Musquash River	2EB-02
Jergens Island (B)	45.55200	-80.43800	175.78	0.67	Embayment	Eastern Coast Islands	2EA-24
Franklin Island at Cunningham Bay	45.40713	-80.35288	175.77	2.54	Embayment	Eastern Coast Islands	2EA-24
Shawanaga Inlet 5	45.49566	-80.38669	175.69	1.84	Embayment	Shawanaga River	2EA-13
Moore Point South (B)	44.81200	-79.79200	175.62	0.71	Embayment	Beausoleil Severn Islands	2EC-18
Spider Bay at Anthony Is	45.22717	-80.10286	175.62	4.90	Embayment	Eastern Coast Islands	2EA-24
Roberts is 3 (C)	44.85200	-79.83500	175.55	10.76	Embayment	Beausoleil Severn Islands	2EC-18
Gooseneck Bay	45.20973	-80.11189	175.52	2.93	Embayment	Eastern Coast Islands	2EA-24
Boyne River	45.30449	-80.04280	175.47	4.97	Embayment	Sequin	2EA-15
North Bay 4 West (A)	44.89359	-79.81909	175.46	6.40	Embayment	Severn River	2EC-17
Sturgeon Bay at Bigwood Island (B)	45.60010	-80.40301	175.46	1.32	Embayment	Eastern Coast Islands	2EA-24
Outer Cognashene Lake	44.95575	-79.93118	175.38	2.47	Embayment	Moon-Musquash Islands	2EB-01

Howl Island (A)	45.19661	-80.09192	175.16	0.99	Embayment	Spider Lake	2EA-08
Outer Prisque Bay North	45.69100	-80.59500	175.13	2.21	Embayment	Giroux River	2EA-04
Cognashene Lake	44.95953	-79.91176	175.11	3.60	Embayment	Moon-Musquash River	2EB-02
Peak Island	45.23341	-80.17728	175.08	1.59	Embayment	Parry Island	2EA-07
Grapps Marsh (B)	45.17305	-80.01821	174.97	7.72	Embayment	Moon-Musquash River	2EB-02
Barrys Channel at Elizebeth Island (B)	45.39300	-80.07900	174.96	0.66	Fringing	Eastern Coast Islands	2EA-24
Juanita Is South	44.77494	-79.71485	174.96	3.05	Embayment	Severn River	2EC-17
Mud Channel (B)	45.67800	-80.60400	174.91	0.97	Embayment	Eastern Coast Islands	2EA-24
Norgate 1	45.72517	-80.62400	174.88	18.47	Embayment	Eastern Coast Islands	2EA-24
Bone Island North Channel	44.94994	-79.84144	174.88	6.01	Embayment	Severn River	2EC-17
Huckleberry Island	45.37905	-80.12304	174.84	2.16	Fringing	Eastern Coast Islands	2EA-24
Big Ship Island	44.92624	-79.87310	174.82	2.62	Embayment	Beausoleil Severn Islands	2EC-18
Hole in the Wall	45.52373	-80.43729	174.82	6.21	Embayment	Eastern Coast Islands	2EA-24
Outer Prisque South	45.68641	-80.59697	174.81	5.83	Embayment	Eastern Coast Islands	2EA-24
Charles Inlet at Naiscoot Middle Channel (B)	45.64400	-80.54200	174.80	64.95	Embayment	Eastern Coast Islands	2EA-24
Ni Bay (Mud Channel)	45.51233	-80.45194	174.79	9.18	Embayment	Eastern Coast Islands	2EA-24
Gunn Island (B)	45.02371	-79.98903	174.77	1.59	Embayment	Moon-Musquash River	2EB-02
Moore Point South (A)	44.81300	-79.79219	174.76	1.37	Embayment	Beausoleil Severn Islands	2EC-18
Moon Bay 2	45.12572	-80.00993	174.73	2.07	Embayment	Moon-Musquash River	2EB-02
Charles Inlet at Naiscoot Middle Channel (A)	45.64957	-80.55464	174.72	61.09	Embayment	Eastern Coast Islands	2EA-24
Shawanaga Island	45.50920	-80.42791	174.68	11.61	Embayment	Eastern Coast Islands	2EA-24
Key River (A)	45.88600	-80.72100	174.67	2.90	Fringing	Henvey Inlet	2EA-01
Key River (B)	45.88560	-80.71742	174.67	4.52	Fringing	Henvey Inlet	2EA-01
Rose Island	45.32219	-80.22954	174.67	11.43	Embayment	Eastern Coast Islands	2EA-24
North Bay 4 West (B)	44.89200	-79.82100	174.67	0.44	Embayment	Severn River	2EC-17
Vennings Bay	44.84123	-79.77966	174.62	11.12	Embayment	Severn River & Beausoleil Severn River	2EC-17;2EC-18

Whitefish Channel at Olga Island	45.74178	-80.63729	174.57	2.14	Embayment	Giroux River	2EA-04
Moore Point North	44.81539	-79.78983	174.53	2.48	Embayment	Beausoleil Severn Islands	2EC-18
Bernadette Is	45.01912	-79.98264	174.53	6.27	Embayment	Moon-Musquash River	2EB-02
Deep Bay 3 (A)	45.40484	-80.21672	174.47	4.56	Embayment	Parry Sound	2EA-14
Roberts Is 2	44.86292	-79.83790	174.41	9.63	Embayment	Beausoleil Severn Islands	2EC-18
Beausoleil Is at Turtle Bay	44.88357	-79.87043	174.40	7.92	Embayment	Beausoleil Severn Islands	2EC-18
Roberts Is 3 (B)	44.85900	-79.82900	174.38	1.70	Embayment	Beausoleil Severn Islands	2EC-18
Franklin Island at Burritt Point	45.39015	-80.32180	174.37	16.01	Embayment	Eastern Coast Islands	2EA-24
Foster Is North	45.69253	-80.61163	174.36	2.38	Embayment	Eastern Coast Islands	2EA-24
Barrys Channel at Elizabeth Island (C)	45.39200	-80.07800	174.25	1.88	Fringing	Eastern Coast Islands	2EA-24
Barrys Channel at Elizabeth Island (A)	45.39142	-80.07952	174.22	3.02	Embayment	Eastern Coast Islands	2EA-24
Tonches Island	45.57034	-80.43019	174.21	4.86	Embayment	Eastern Coast Islands	2EA-24
Hermans Bay	45.08675	-79.99711	174.19	3.47	Embayment	Moon-Musquash River	2EB-02
Black Rock (A)	45.04254	-79.97425	174.13	6.14	Embayment	Moon-Musquash River	2EB-02
Grapps Marsh (A)	45.17200	-80.01600	174.10	3.69	Embayment	Moon-Musquash River	2EB-02
Shawanaga Inlet 6	45.50379	-80.38874	174.10	5.93	Fringing	Shawanaga River	2EA-13
Blackstone Bay 2	45.15982	-79.99166	174.06	5.34	Fringing	Moon-Musquash River	2EB-02
Twelve Mile Bay	45.08933	-80.00853	173.95	4.22	Embayment	Moon-Musquash River	2EB-02
Mud Channel (A)	45.67734	-80.60104	173.91	1.52	Embayment	Eastern Coast Islands	2EA-24
Port Rawson Bay	45.19685	-80.02186	173.88	5.93	Embayment	Moon-Musquash River	2EB-02
Sturegon South	44.73720	-79.73831	173.82	122.82	Fringing	Sturgeon River	2ED-04
Vanderdasson Is	45.18530	-80.06541	173.82	7.61	Embayment	Moon-Musquash Islands	2EB-01
Moose Bay (A)	45.07000	-80.05100	173.79	8.41	Embayment	Moon-Musquash Islands	2EB-01
Roberts Is 3 (A)	44.85540	-79.83264	173.69	13.41	Embayment	Beausoleil Severn Islands	2EC-18
Mackenzie Island	45.55543	-80.44129	173.59	1.38	Fringing	Eastern Coast Islands	2EA-24

Woods Bay	45.14957	-80.00781	173.54	4.35	Embayment	Moon-Musquash River	2EB-02
Pittsburg Channel at Moreau Bay	45.00787	-79.94159	173.47	4.13	Embayment	Moon-Musquash Islands	2EB-01
Captain Allan Channel	45.12947	-80.00299	173.45	2.01	Fringing	Moon-Musquash Islands	2EB-01
Spider Lake Portage	45.21654	-80.07522	173.45	3.11	Fringing	Spider Lake	2EA-08
Roberts Is 3 (D)	44.85600	-79.84300	173.45	23.30	Embayment	Beausoleil Severn Islands	2EC-18
Rathlyn Island	45.55327	-80.41143	173.40	6.10	Embayment	Eastern Coast Islands	2EA-24
Nightingale Bay	45.15933	-80.06770	173.30	3.13	Embayment	Moon-Musquash Islands	2EB-01
Minominee Channel	45.29784	-80.08355	173.25	4.92	Embayment	Parry Island	2EA-07
Sturgeon Bay 1	45.61456	-80.41566	173.22	6.75	Embayment	Point au Baril	2EA-05
Henvey Inlet 1	45.83427	-80.67851	173.07	16.64	Embayment	Sandy Bay	2EA-03
Magnetawan River South side	45.76626	-80.61248	172.93	2.46	Fringing	Magnetawan	2EA-10
Blackstone Bay 1	45.18119	-80.00291	172.81	31.48	Embayment	Moon-Musquash River	2EB-02
Black Rock (B)	45.04500	-79.97100	172.81	3.69	Embayment	Moon-Musquash River	2EB-02
Hay Bay	45.32263	-80.07668	172.64	3.03	Embayment	Parry Island	2EA-07
Deep Bay 3 (B)	45.40400	-80.21300	172.50	1.14	Fringing	Parry Sound	2EA-14
Big Bass Bay North (A)	45.03559	-79.99035	172.42	9.49	Embayment	Moon-Musquash River	2EB-02
Big Bass Bay North (B)	45.03700	-79.99200	172.42	1.47	Fringing	Moon-Musquash River	2EB-02
Big Bass Bay South	45.03123	-79.98676	172.42	24.93	Embayment	Moon-Musquash River	2EB-02
Pointe au Baril Channel 1	45.57455	-80.45712	172.26	3.86	Embayment	Eastern Coast Islands	2EA-24
Matchedash Bay	44.74778	-79.67816	172.25	1027.35	Embayment	Cold Water River	2ED-05
Roberts Is 1 (A)	44.86900	-79.83000	172.24	0.88	Fringing	Beausoleil Severn Islands	2EC-18
Roberts Is 1 (B)	44.86823	-79.83174	172.24	1.55	Embayment	Beausoleil Severn Islands	2EC-18
Moon River 2	45.10891	-79.95346	172.21	27.07	Embayment	Moon-Musquash River	2EB-02
Henvey Inlet 2	45.85811	-80.64597	172.19	2.73	Embayment	Henvey Inlet	2EA-01
Howl Island (B)	45.19500	-80.09200	172.18	1.06	Embayment	Spider Lake	2EA-08
Surprise Channel	45.26200	-80.19989	172.10	10.57	Embayment	Eastern Coast Islands	2EA-24

North Bay	44.89258	-79.78852	172.03	2.68	Fringing	Severn River	2EC-17
Pointe au Baril Channel 2	45.57827	-80.43990	171.44	3.40	Embayment	Eastern Coast Islands	2EA-24
Coffin Rock	45.04734	-79.98757	171.25	7.35	Fringing	Moon-Musquash Islands	2EB-01
Deep Bay 1	45.39713	-80.23590	171.18	6.97	Embayment	Parry Sound	2EA-14
Moon River 1	45.11115	-79.96693	171.00	12.43	Fringing	Moon-Musquash Islands	2EB-01
Jergens Island (A)	45.55186	-80.43957	170.79	2.36	Embayment	Eastern Coast Islands	2EA-24
Sturgeon Bay 2	45.60546	-80.43275	170.74	3.22	Embayment	Point au Baril	2EA-05
Sturgeon Bay at Bigwood Island (A)	45.59874	-80.40345	170.08	5.55	Embayment	Eastern Coast Islands	2EA-24
Shawanaga Inlet 4	45.47568	-80.39524	169.87	5.93	Fringing	Eastern Coast Islands	2EA-24
Collins Bay at Deep Bay	45.39348	-80.22062	167.30	1.82	Fringing	Parry Sound	2EA-14
Ireland Point	45.39186	-80.31088	167.07	0.61	Fringing	Shebeskekong	2EA-06
Beausoleil Is 2	44.85034	-79.86227	164.78	2.52	Fringing	Beausoleil Severn Islands	2EC-18
Giroux River Outlet	45.70437	-80.60192	162.94	17.87	Embayment	Giroux River	2EA-04

Note: Some wetland are smaller than 2 ha as upon field evaluation the wetland was smaller or a complex of multiple wetlands

Table 1.2: Inverse logistical equations found to estimate the fish habitat

	Y Variable	Equation
a	Percent Average Surface Area Inaccessible Within an Individual Wetland	=2.653+100.832/(1+EXP(1.8241*(Water Level-175.4000)))
b	Percent Total Number of Coastal Wetlands Inaccessible	= -37.03+133.38/(1+EXP(0.9066*(Water Level-174.785)))
c	Percent Total Surface Area Inaccessible	=-14.43 +109.31/(1+EXP(1.3854*(Water Level-174.2798)))

Table 1.3: Declines in water levels with different climate change scenarios applied to models.

Model/Emission	Condition of Model	Δ Water Level (m)	Elevation (m asl)	% Total Number of Coastal Wetlands Inaccessible	% Total Surface Area Inaccessible	Total Wetland Area Inaccessible (ha)	% Average Surface Area Inaccessible Within Individual Wetland
GISS[a]	2xCO2	-1.31	175.23	16.39	8.68	567.82	60.82
GFDL[a]	2xCO2	-2.48	174.06	50.82	48.48	3171.60	95.43
OSU[a]	2xCO2	-0.99	175.55	7.42	1.62	105.93	46.21
CCC1[a]	2xCO2	-1.62	174.92	25.58	17.46	1142.21	73.83
CGCM1[b]	2030	-0.72	175.82	0.48	N/A	N/A	34.65
CGCM1[b]	2090	-1.38	175.16	18.43	10.50	686.66	63.93
B1- 5th[c]	2020-2034	-0.6	176.144	N/A	N/A	N/A	23.29
B1- 5th[c]	2050-2064	-0.79	175.954	N/A	N/A	N/A	29.56
B1- 5th[c]	2080-2094	-0.87	175.874	N/A	N/A	N/A	32.54
A1B- 5th[c]	2020-2034	-0.55	176.194	N/A	N/A	N/A	21.84
A1B- 5th[c]	2050-2064	-0.91	175.834	0.14	N/A	N/A	34.09
A1B- 5th[c]	2080-2094	-1.41	175.334	13.40	6.16	403.18	56.10
A2- 5th[c]	2020-2034	-0.63	176.114	N/A	N/A	N/A	24.21
A2- 5th[c]	2050-2064	-0.94	175.804	0.87	N/A	N/A	35.29
A2- 5th[c]	2080-2094	-1.18	175.564	7.04	1.36	88.67	45.58

a-Mortsch & Quinn, 1996; b-Lofgren et al, 2002; c-5th percentile of simulated 23 GCM/GLERL model by Angel & Kunkel, 2010

Figures

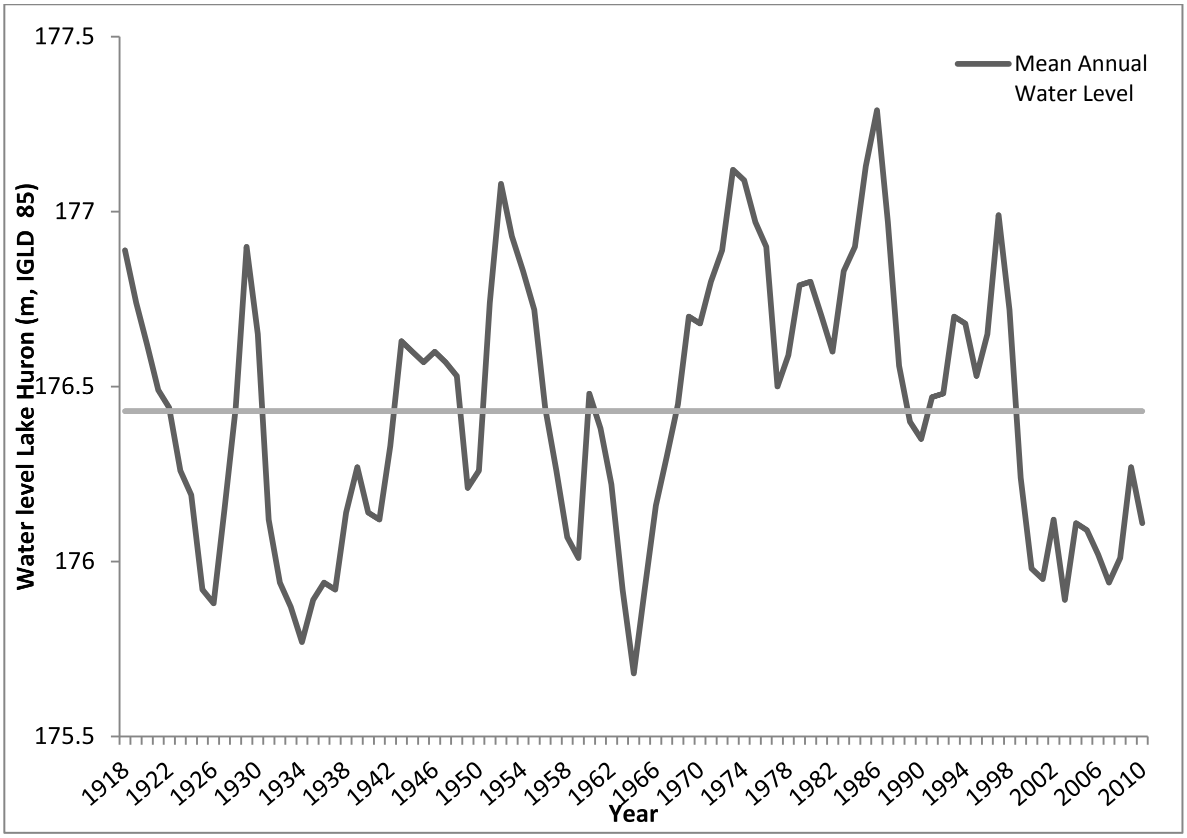

Figure 1.1: Mean annual water levels for Lake Huron from 1918 to 2010. The straight line is the overall average of Lake Huron throughout this time period. Data obtained from Canadian Hydrographic Services (Canadian Hydrographic Services, 2012).

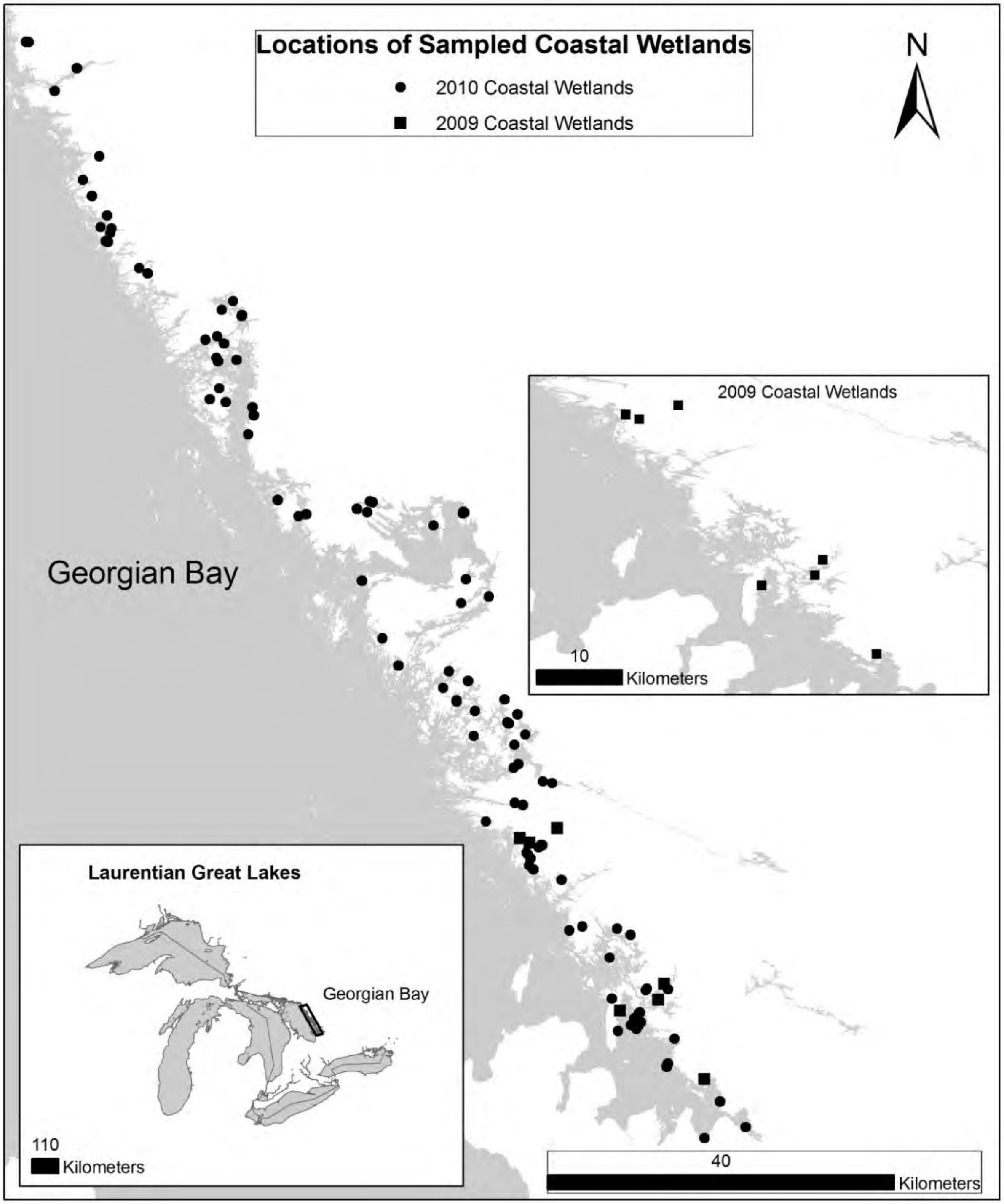

Figure 1.2: Location of sites sampled in this study during 2009 (open circles) and in 2010 (solid circles).

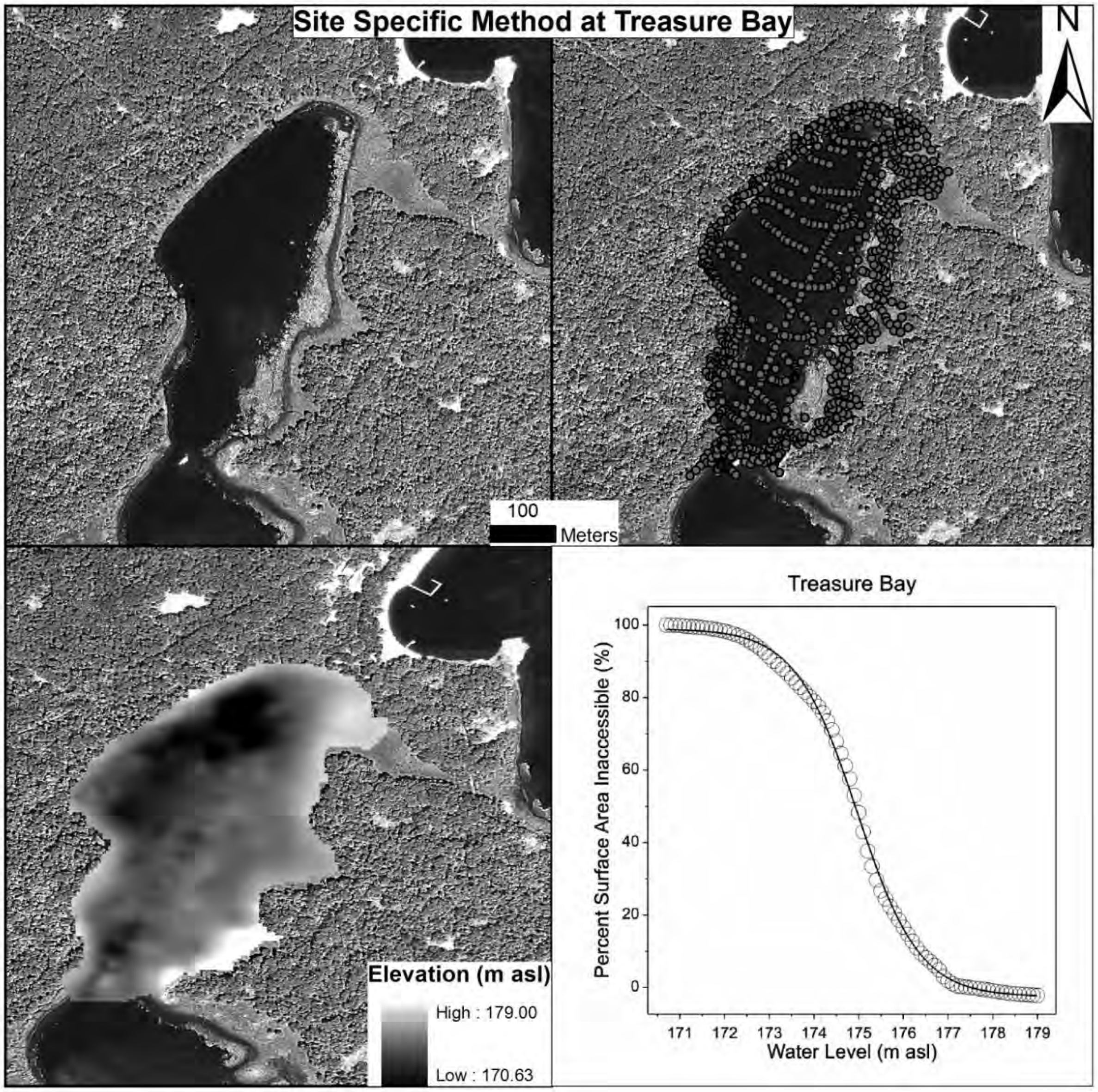

Figure 1.3: Stages in the creation of DEM for use in the site-specific method. a) Obtaining an image of Treasure Bay on IKONOS imagery, ensuring that the entire wetland is covered b) Overlaying data points collected in the field on the IKONOS imagery c) Deriving a DEM for Treasure Bay by using topo to raster method of interpolation and d) Plotting the relationship between percentage of surface area lost as a function of water level for Treasure Bay.

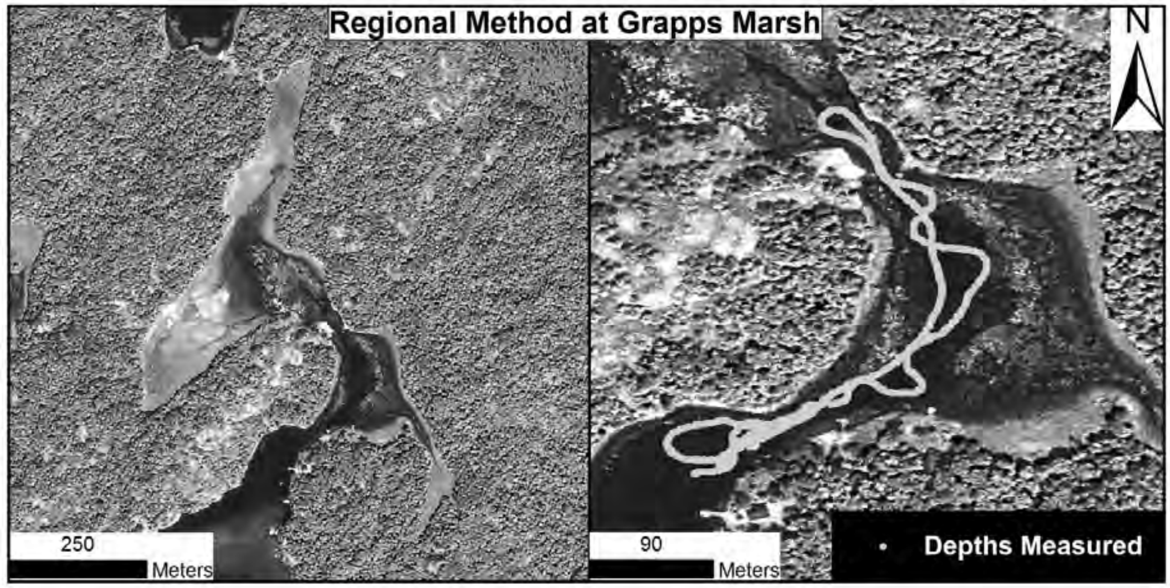

Figure 1.4: Stages in determining the deepest spot in wetland outlet for the regional method a) Obtaining an image of Grapps Marsh on IKONOS imagery, ensuring that the entire wetland is covered b) Overlaying data points of transects collected in the field to determine the lowest elevation.

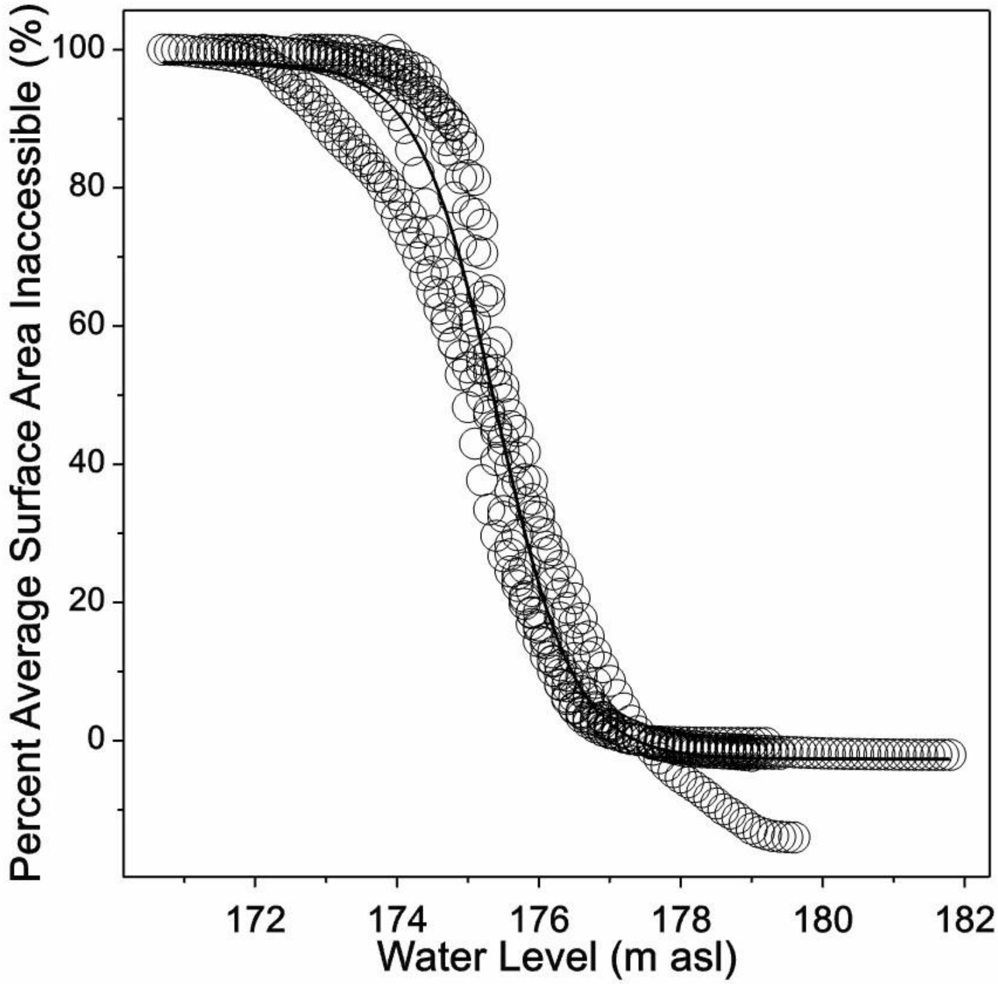

Figure 1.5: Percentage of total surface area of wetlands that would become inaccessible to fish as a function of water levels for seven wetlands in eastern Georgian Bay. Line is the inverse logistic regression line through all data points.

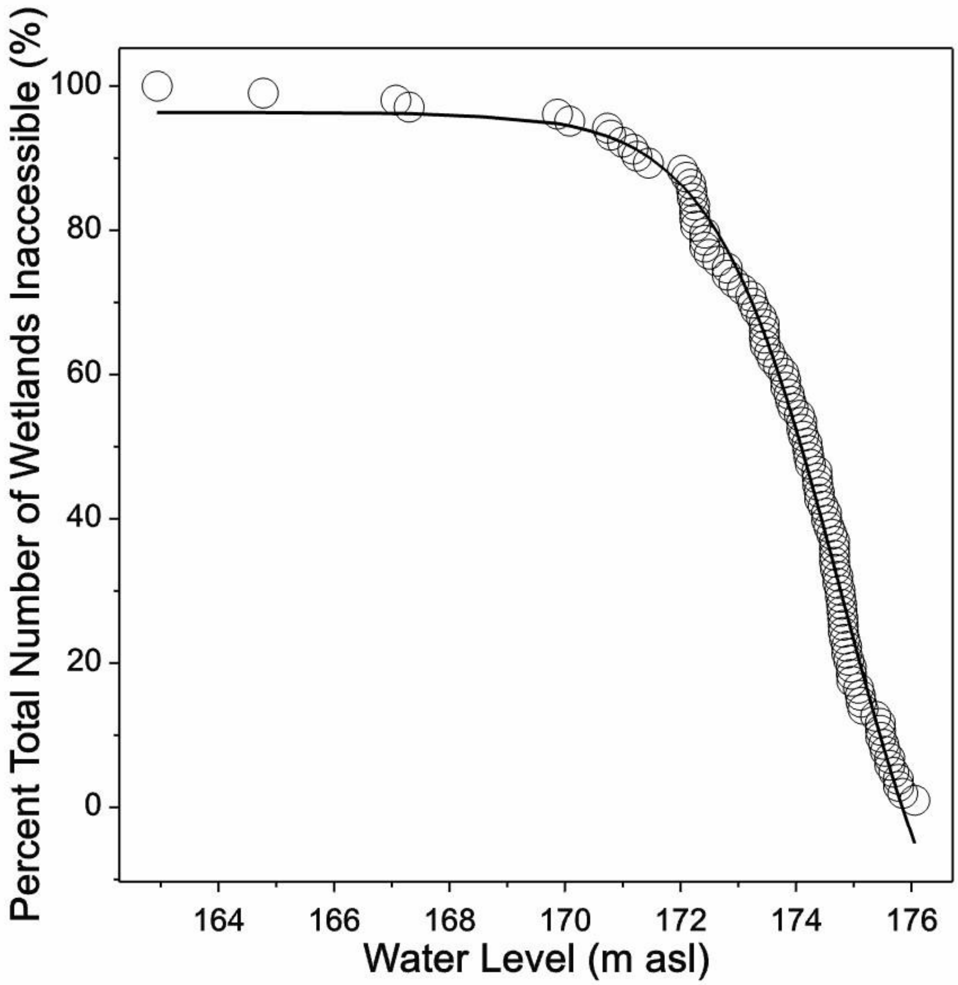

Figure 1.6: Percentage of total number of wetlands that would become inaccessible to fish as a function of water levels for 103 wetlands in eastern Georgian Bay. Line is the inverse logistic regression line through all data points.

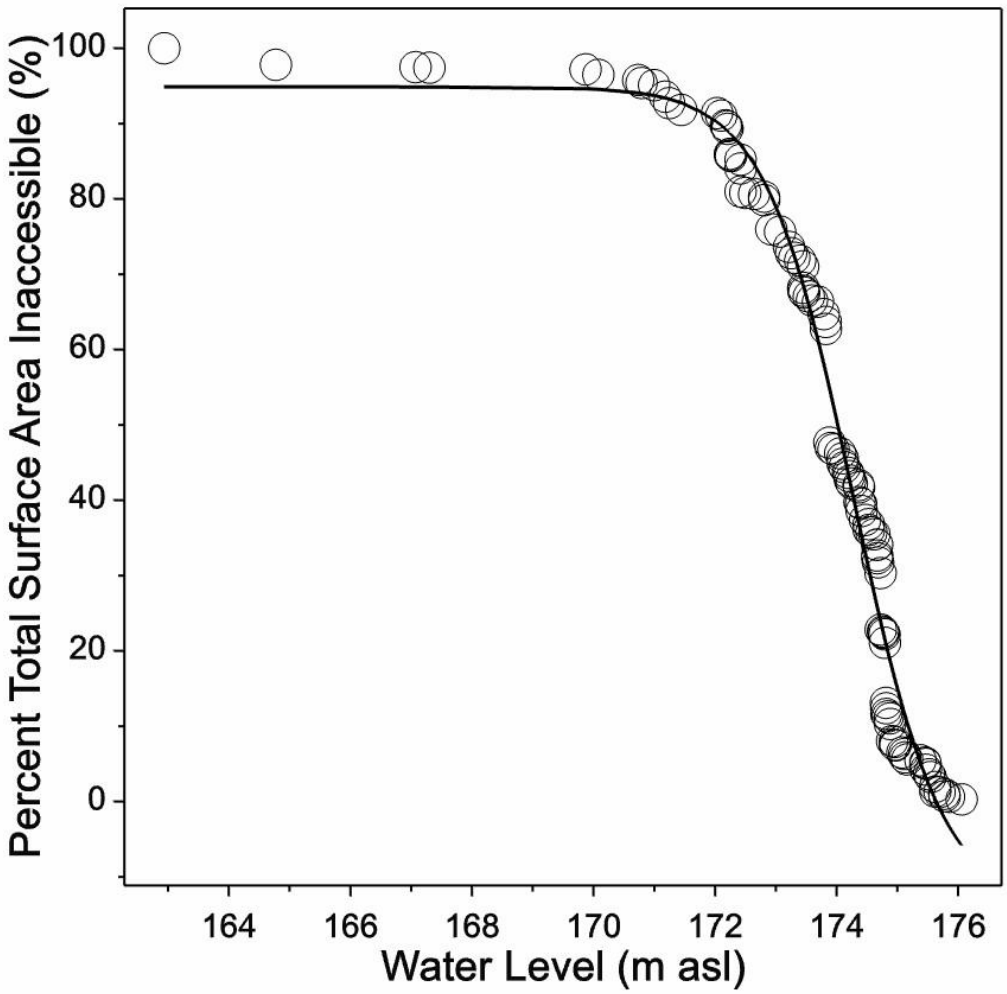

Figure 1.7: Percentage of total surface area of wetlands that would become inaccessible to fish as a function of water level for 103 wetlands in eastern Georgian Bay. Line is the inverse logistic regression line through all data points.

Chapter 2:

Hydrologic connectivity of coastal marshes and their associated ecological and chemical alterations in wetlands of eastern Georgian Bay

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Abstract

Coastal Marshes of eastern Georgian Bay are naturally dystrophic systems with water chemistry that reflects mixing between the relatively ion-rich waters of Georgian Bay and the ion-poor water draining the Shield landscape. Since many of these marshes have formed along the convoluted shoreline of granite bedrock, they have narrow outlets that can be easily breached by beaver activity especially when lake levels drop. Once an impoundment forms, the marsh becomes hydrologically disconnected from Georgian Bay and this altered hydrology may dramatically affect the water chemistry of the impounded wetland. During the summers of 2010 and 2011, we tested this hypothesis by sampling 35 coastal marshes in eastern Georgian Bay, 17 of which had beaver impoundments built at the outlet of the coastal wetland. Impounded marshes had significantly higher soluble reactive phosphorus, (13.33 vs. 12.96 µg/L, $p \leq 0.0001$) but significantly lower pH (6.06 vs. 6.89, $p=0.0024$), conductivity (47.15 vs. 81.76 µs/cm, $p=0.0007$), dissolved oxygen (3.1 vs. 4.91 mg/L, $p=0.011$) and sulphate concentrations (0.54 vs. 2.02 mg/L, $p=0.0438$), conditions indicative of reduced mixing with bay water. Larval amphibian surveys conducted in the beaver impoundments confirmed that these environments are favourable for 7 species of frogs, with the most common being green frogs and the least common being American toads and chorus frogs. We conclude that hydrologic connectivity with the bay is a critical process that governs the water chemistry of coastal marshes in eastern Georgian Bay, and the presence of beaver impoundments must be taken

into consideration when defining criteria for reference water-quality conditions of coastal wetlands in the Georgian Bay Biosphere Reserve.

Introduction

Coastal marshes are some of the most threatened habitats in the Laurentian Great Lakes basin because they form in rivermouths, deltas and in protected embayments, where human development has also been concentrated (Maynard and Wilcox, 1996). They are located at the interface between the land and water and have water chemistry that reflects both offshore processes and runoff from the watershed (deCatanzaro & Chow-Fraser, 2011, Morrice et al, 2011). The extent to which each can influence the water chemistry in a particular wetland will depend on its geomorphology. Wetlands that have a permanent strong hydrologic connection with the Great Lake will be more heavily influenced by the chemical properties of the lakewater, whereas those with a weak and intermittent connection will be heavily influenced by runoff in the watershed and its associated land cover. The Precambrian Shield of eastern Georgian Bay naturally exports large amounts of sediments, dissolved organic carbon, phosphorus and colour to coastal marshes (Dillon and Molot, 2005; Eimers et al, 2008; deCatanzaro & Chow-Fraser, 2011), and its thin layer of acidic soils and granitic rocks contribute very few dissolved ions to its runoff (Weiler, 1988). By contrast, the open water of Georgian Bay represents a source of conductivity, alkalinity, nitrates and sulphates due to sedimentary bedrock at the southern and western portions (deCatanzaro & Chow-Fraser, 2011). Therefore, based on characteristics of the water chemistry in these coastal wetlands, it is relatively easy to determine if the water has a disproportionate contribution from Georgian Bay or from the watershed (see deCatanzaro & Chow-Fraser, 2011).

There are over 6,500 ha of coastal marshes in eastern Georgian Bay (Midwood et al, in press), which contain some of the best water quality in the entire Great Lakes basin (Chow-Fraser, 2006; Cvetkovic and Chow-Fraser 2011). The excellent water-quality conditions in many of these wetlands can be attributed to the absence of human disturbance in the adjacent landscape (Chow-Fraser, 2006; deCatanzaro & Chow-Fraser, 2009). Based on the research of deCatanzaro and Chow-Fraser (2011), we know that water chemistry in these wetlands is heavilly influenced by the degree of connectivity between the coastal marsh and Georgian Bay. Fracz and Chow-Fraser (2012; Chapter 1) has already shown that the hydrologic connection of coastal wetlands to Georgian Bay is a function of water level in Lake Huron (Fracz and Chow-Fraser 2012; Chapter 1). Unlike open embayments where wetland vegetation can migrate lakeward and shoreward with changes in water level, the marshes of eastern Georgian Bay have formed in fringes along the shorelines and in protected embayments that have a narrow marsh outlet (Midwood et al, in press). When water levels drop below the maximum depth of the outlet, many of these marshes can become hydrologically disconnected from Georgian Bay. Even if water levels do not drop below the outlet, a wetland may become cut off if a beaver impoundment is built across the outlet.

The water levels of Lake Huron, and therefore Georgian Bay, have been kept at extremely low levels since 1999 (Assel et al, 2004; Sellinger et al, 2008). This decline has been attributed to several factors, including a change in conveyance of the St. Clair River (see IUGLS, 2009) and hydroclimatic factors such as increased evaporation (see IUGLS, 2009; Hanrahan et al, 2010). The International Joint Commission has acknowledged that anthropogenic activities have influenced natural fluctuations: human-induced climate change (Karl & Trenberth, 2003) and dredging in the St. Clair River in 1962 which led to erosion at the outflow of Lake Huron (IUGLS, 2009). Lower water levels have led to shallower depths at marsh outlets and have created very favourable conditions for beavers to build impoundments across them. Ideal wetlands to dam would have a small cross-sectional area at outlet (Barnes & Mallik, 1997; Jensen et al, 2001), low gradient (Howard & Larson, 1985; Beier & Barrett; 1987; Jenson et al, 2001) and intermediately sized watershed (Howard & Larson,1985; Barnes &Mallik, 1997).

Alterations of stream environments by beaver dams have resulted in retention of sediment and organic matter, alteration in nutrient cycling and decomposition, as well as, changes in plant and animal community composition and diversity behind the dam (Naiman et al, 1986, 1988). Since fish movement across beaver impoundments is restricted, piscivorous predators are lacking (Keast and Fox, 1990) benefiting amphibians, and thus explaining why frogs and toads often propagate profusely in beaver ponds (Rosell et al, 2005). Young-of-year amphibian captures have been significantly higher in beaver ponds compared with those in unobstructed stream reaches in the boreal foothills for *Bufo boreas* (western toad), *Pseudacris maculata* (chorus frog) and *Rana sylvatica* (wood frogs) (Stevens et al, 2007). The age of the beaver impoundment has also been shown to influence populations of the wood frog in west-central Alberta (Stevens et al, 2006).

Few studies have documented the relationship between mature amphibians and beaver impoundments, while even fewer have examined the impact of beaver impoundments on the distribution and species richness of larval amphibians (Cunningham et al, 2007). France (1997) found that *Rana clamitans* (green frog) tadpoles and *Notophthalum viridescens* (red spotted newts) were significantly more abundant near beaver lodges. Cunningham et al (2007) found that beavers created habitat to support species-rich amphibian assemblages at a local scale while also increasing connectivity of wetlands and diversity at a landscape scale. Understanding how beaver impoundments support amphibian breeding is important as global declines of amphibian populations are well documented (Semlitch, 2000; Lannoo, 2005).

In this paper, we examine the effects of a beaver impoundment on the water chemistry of the impounded marsh and compare it to coastal marshes in eastern Georgian Bay. To avoid any confounding effects of human disturbance, we selected beaver-impounded sites with reference conditions (as determined by deCatanzaro and Chow-Fraser 2011) and with similar-sized watersheds. We predict that water above a beaver impoundment will have water chemistry that more closely reflects processes in the watershed of the Precambrian Shield with increased amounts of nutrients and sediments, while water below the impoundment will have water chemistry more reflective of the open water. In addition, we predict that water chemistry in coastal marshes will be distinct from that of beaver impounded marshes as they are hydrologically connected to Georgian Bay and will be influenced by open water. An additional objective of this study is to determine the biotic community that utilizes this modified habitat. We focused on the amphibian community, given the current lack of knowledge between their association with beaver dams and their declining global status. Results from this study will fill an important gap in the literature concerning the role of beaver impoundments on the water chemistry of coastal wetlands and help managers anticipate ecosystem changes that may accompany the current period of sustained low water levels in Lake Huron.

Methods

Study Area and Site Selection

The coastal wetlands in eastern Georgian Bay are unique compared to other regions within the Laurentian Great Lakes drainage basin. There are two geologic regions; sedimentary limestone surrounds the southern and western portion and Precambrian Shield granitic rock encompasses the remaining portion.

Water chemistry is naturally oligotrophic due to low nutrient inputs and the connection to Georgian Bay (deCatanzaro & Chow-Fraser, 2009). The ecology of these ecosystems depend on mixing with lake water to maintain its unique character and biodiversity of plants (Croft & Chow-Fraser, 2009), fish (Wei et al, 2004; Cvetkovic et al, 2010) and species at risk (deCatanzaro & Chow-Fraser, 2010). The soils are thin, sandy, acidic, and patchy (Weiler, 1988). Vegetation on land is a mixture of second growth deciduous and coniferous forest consisting of white pine (*Pinus strolobus*), red pine (*Pinus resinosa*), hemlock (*Tsuga canadensis*), white spruce (*Picea glauca*), sugar maple (*Acer saccharum*), red oak (*Quercus rubra*) and beech (*Fagus grandifolia*) (deCatanzaro &Chow-Fraser, 2011).

Although there is no recent population census of the beaver populations in Ontario, the Ontario Ministry of Natural Resources (OMNR) has kept a record of the number of beavers harvested along trap lines from 1993 to 2010 in eastern Georgian Bay along an approximate 40 km stretch from just south of Parry Sound to north of Honey Harbour (Figure 2.1). The annual harvest over the past 20 years have ranged from 15 in 1993 to 106 in 1998 and has been trending downwards since 2004. Without knowing the number of permits issued, however, it is inappropriate to draw any conclusions about the current status of the population. Beavers can reproduce three to four kits each year, and kits will generally disperse during their second spring (Muller- Schwarze & Sun, 2003), therefore, a cessation in harvest could lead to quick expansion of the population, especially when conditions become favourable for dam creation. The average density of dams in other regions has been estimate to be 5 to 19 dams/km of stream for inland populations in Ekwan Point, Northern Ontario (Woo & Waddington, 1990) and 8.6 to 16 dams/km of stream in southeastern Quebec (Naiman et al, 1988).

In a previous study, we sampled 103 wetlands that had been randomly selected from 18 quaternary watersheds in eastern Georgian Bay (see Fracz and Chow-Fraser, Chapter 1); a subset of 18 of these coastal marshes was used for the present study (Figure 2.2). These marshes were over 2 ha in size, which is the size criterion used by the Ontario Ministry of Natural Resources (OMNR) to determine if wetlands qualify for assessment under the Ontario Wetland Evaluation System (OMNR, 1993). Since these marshes are a subset of randomly selected coastal wetlands, we believe that they are representative of marshes typically found in eastern Georgian Bay.

An inventory of beaver-impounded wetlands in eastern Georgian Bay was assembled by visual examination of IKONOS satellite imagery in a Geographic Information System (GIS) for a stretch of shoreline between from Parry Sound to Severn Sound. From this inventory, we chose 17 impounded wetlands that appeared to have been at one time connected to Georgian Bay, but which has become blocked by a beaver dam at the wetland outlet (Figure 2.2).

Field Sampling

We sampled the 18 coastal marshes (not associated with any impoundments) between May and September of 2010 and the 17 impounded wetlands between the end of May and mid-July of 2011. Water samples in the latter were collected at mid-depth within 25 meters above the beaver dam, and within 25 meters below the dam to determine the effect of the dam on water chemistry. Water from coastal marshes that were connected to the bay was collected at mid-depth in an area that appeared representative of the marsh. Effort was made not to disturb submergent vegetation or underlying sediment while collecting water samples. We avoided sampling within 24 hours of a noticeable (>5mm) rain event. Timing of water collection varied between 9:00 and 16:30 h, but all dissolved oxygen measurements were taken between 11:00 and 13:00 h to minimize bias caused by diurnal events and create standardization. In situ measurements of dissolved oxygen (DO), pH, and conductivity (COND) were measured with an YSI 6600 multiprobe (YSI, Yellow Springs, Ohio, USA). Turbidity (TURB) was measured in situ with a Turbidimeter LaMotte 2020e (LaMotte, Chestertown, Maryland, USA).

We collected samples for determination of total nitrate nitrogen (TNN), total ammonia nitrogen (TAN), colour (COL) and sulphate (SULPH) and processed them the same day with Hach reagents using a Hach DR/890 colourimeter and prescribed protocols. Pre-weighed 0.45µm GF/C filters were used to process water samples for total suspended solids (TSS) while unweighed filters were used for determination of chlorophyll a concentration (Chla). The filtrate was frozen and stored for soluble reactive phosphorus (SRP) and DOC analysis. Raw water samples for total phosphorus (TP) were also frozen and stored. TP samples were digested by persulfate digestion in an autoclave. SRP samples (undigested) and TP samples were then analyzed with the molybdenum blue method (Murphy and Riley 1962) and absorbance readings were measured with the Genesys 10 series spectrophotometer (GENEQ inc., Montreal, Quebec, Canada). TSS filters were dried in a drying oven at 100^{o}C, placed in a dessicator for at least an hour and then weighed. Dissolved organic carbon (DOC) samples were analyzed with the NPOC method in a Shimdzu TOC-VCHP analyzer. Chla samples were extracted in 10 mL of acetone in a freezer for a minimum of two hours of extraction. The extract was then measured in a flourometer and acidified with hydrochloric acid and re-measured to account for phaeophytin. COL, SULPH, DOC, and standardized DO were only measured during 2011.

Larval amphibians were surveyed only in impounded marshes in 2011. The survey was completed by rapidly sweeping a D Amphibian net (Wildlife Supply Company, Yulee, Florida, USA) along the perimeter of the wetland during a standardized 60-min period. Two impoundments, Robert Islands 4 and North Bay 4 West, were sampled only for 30 minutes because of time constraints. Captured larval amphibians were held in buckets filled with water from the impoundment. Care was taken to keep the water temperatures in the buckets close to the ambient water temperature. In addition, water in the bucket was exchanged with freshwater at a regular interval to avoid hypoxic conditions in the bucket. Once the larval amphibian survey was completed, larval amphibians were identified on site and released. To maximize sampling efficiency, we sampled only at the time when amphibian eggs were expected to hatch (late May to early July).

Data Analysis and Statistics

Statistical analysis was performed in JMP version 7.0.1 and R version 2.13.1, with $\alpha = 0.05$. SULPH, TAN and TNN values that fell below the detection limit (1 mg•L^{-1}, 10 µg•L^{-1} and 10 µg•L^{-1}, respectively) were assigned half the detection limit value, a technique commonly used to treat data with concentrations below detection (see Trebitz et al, 2007; deCatanzaro & Chow-Fraser, 2011). The data were normalized and parameters were either $\log_{10}$ or square-root transformed to achieve normality. A pairwise t-test was used to determine statistically significant differences between water chemistry parameters measured above and below the beaver impoundment. Due to processing and sampling errors, there were missing values for some of the variables (4 for SRP, 2 for DO and 1 each for SULPH and TURB). A Principal Components Analysis (PCA) was run with a correlation matrix that included Chla, TSS, TP, SRP TNN, TAN, pH, and COND for 15 beaver-impounded sites and 18 coastal marshes. We determined the association between water chemistry variables and the first three PC axes using Spearman correlations.

Results

Comparison of Water Chemistry Above and Below the Beaver Dam

Five variables differed significantly above and below the beaver dam (pairwise t-tests in all cases; Table 2.1). SRP was significantly higher within the impoundment compared with water below the dam (13.33 vs 12.96; $p \leq 0.0001$) whereas pH (6.06 vs 6.86; p=0.0024), COND (47.15 vs 81.76; p=0.0007), DO (3.1 vs 4.91; p= 0.011), and SULPH (0.54 vs 2.02; p= 0.0438) were all significantly higher below the dam than in the impoundment. While there were no significant differences for the other water chemistry parameters, concentrations of all variables that were lake-driven were lower above the beaver dam than below it, and all variables that were watershed-driven (except for TAN) had concentrations higher above the dam than below the dam.

Comparison of Coastal Marsh Water Chemistry to Beaver Impounded Coastal Marsh

Since all beaver impounded marshes were sampled in 2011, we wanted to confirm that there was no year-to-year variation. Three of the same beaver impoundments had been opportunistically sampled in 2010 and a non-parametric t-test was conducted to determine if there was significant variation between years. No significant differences (all p-values > 0.05) in water chemistry variables were found and therefore, it is appropriate for us to compare coastal-marsh data collected in 2010 with beaver-impoundment data collected in 2011. This is consistent with other studies (see Cvetokovic & Chow- Fraser, 2011) that used data from multiple years to make comparisons in the same geographic locations.

Since coastal marshes that were not associated with beaver dams were distributed between Key River and Severn Sound, while impounded marshes only occurred south of Parry Sound, we conducted a non-parametric t-test to determine if wetlands north of Parry Sound were significantly different from that South of Parry Sound. We found no significant differences, and therefore conclude that no such bias exists. Hence, the PCA was performed with the combined dataset (18 from 2010 and 17 from 2011).

The PCA yielded three axes, with the first explaining 53.3 % of the variation in the dataset, the second explaining an additional 13% and the third explaining an additional 11.8 % (Figure 2.3). PC1 was highly positively correlated with Chla, TSS, TP and SRP, and negatively correlated with pH, COND and TNN (Table 2.2), while PC2 was negatively correlated with TAN. Sites with high positive PC1 scores corresponded with beaver impoundments that had high concentrations of phosphorus, suspended sediments and algae in the water (solid circles in Figure 2.3), whereas those with high negative PC1 scores were the coastal marshes without any association with beaver dams, that had more alkaline water (higher pH and COND), higher buffering capacity, and higher concentrations of TNN (open circles in Figure 2.3). Sites with high negative PC2 scores had high concentrations of TAN and also corresponded with beaver impoundments.

Impacts on Biota: Amphibians and Beaver Impoundments

Seven species of larval amphibians were found in beaver-impounded wetlands in Georgian Bay (Figure 2.4; Figure 2.5). These species included *Rana clamitans* (green frog), *Rana catesbeiana* (bullfrog), *Rana pipiens* (leopard frog), *Pseudacris triseriata* (chorus frog), *Bufo americanus* (American Toad), *Psuedacris crucifer* (spring peeper), and *Hyla versicolor* (gray treefrog). The greatest numbers of larval amphibians found were bullfrog and green frog species as they made up approximately 82 percent of the individuals caught cumulatively in our wetlands (Figure 2.5). There was a skewed distribution of bullfrog tadpoles, however, with their occurrence in only 5 of 17 wetlands. Even though numerically bullfrogs were as abundant as green frog tadpoles, the latter had a more widespread distribution and were found in all but three wetlands (Figure 2.4; Table 2.3). Spring peepers were the second most common species in the impoundments while treefrog, leopard frog, American toad and the chorus frog were found in much lower numbers and at fewer sites (Table 2.3).

Discussion

Few published studies include data on impounded coastal marshes of large lakes. Most focus on lotic environments where the investigators examine changes caused by beaver impoundments before the water enters the impoundment and directly below the dam where it is flowing into a stream (see Cirmo & Driscoll, 1993; Magolis et al, 2001). In our system, however, the outlet of the impoundment flows into a lake rather than a stream. Therefore, caution must be exercised when comparing water chemistry in “below dam” situations across studies, as in our system, this is more reflective of lake conditions whereas in riverine systems this tends to reflect the effect of the beaver dam. Despite this difference, when comparing the effects of the dam, there are many similarities between studies with respect to landscape context. Similar to our study, there were higher concentrations of TP (Klotz, 1998), Chla (Bledzki et al, 2011) and TSS (Maret et al, 1987) due to the formation of a beaver impoundment. This reinforces the generally accepted assumption that impoundments are sinks for phosphorus and suspended particulates and potentially sources depending upon hydrology and structural integrity of the dam.

All of the lake-driven variables sampled above the impoundment, with the exception of TNN, were significantly different from those sampled below (Table 2.1). This is because the beaver dam severed the hydrologic connection between the wetland and Georgian Bay. Although we found a significantly lower pH in the beaver impoundment compared with water below the dam, others have reported increased pH and high acid-neutralizing capacity in beaver ponds (Cirmo & Driscoll, 1993; Margolis et al, 2001; Bledzki et al, 2011). This discrepancy is likely due to the location of our study; the Canadian Shield landscape is characteristic of ion-poor runoff and a thin layer of acidic soil (Weiler, 1988), whereas other studies have taken place in different geologic environments such as the Adirondack Mountains, south-western Pennsylvania, and Massachusetts. The lower DO and SULPH in the beaver impoundments are supported by Cirmo & Driscoll (1993) and Margolis et al (2001) and is reflective of altered connectivity. We speculate that the lack of significant differences for TNN is a reflection of the dynamic cycling of nitrates and the existence of multiple sources of nitrate that can enter the system both above and below the beaver dam. Surprisingly, we did not find significant differences in many watershed-influenced parameters for samples collected above and below the dam (e.g. DOC, TAN, COL, TSS, Chla, TP and TURB; Table 2.1). There may have been confounding landscape factors (e.g. drainage basin size, slope) or ones related to differences in time of sampling and our small sample size. Further studies should be conducted to examine these relationships under more standardized conditions.

Water chemistry in beaver impounded coastal marshes is highly dependent upon the degree of connectivity to both its catchment and the lake (Morrice et al, 2011). Although, we did not have any data for open waters of Georgian Bay, we were aware that the Ontario Ministry of the Environment (OMOE) had conducted a survey of water-quality conditions in many open-water sites in the nearshore zone of Georgian Bay in 2005 (N. Diep, unpub. data). Because of the large overlap in geographic coverage between OMOE sites and those in this study, we felt justified to compare the data. We also used data from deCatanzaro & Chow-Fraser (2011) to supplement information for coastal marshes.

This comparison clearly illustrates the differential influence of the catchment and Georgian Bay with respect to several water-chemistry variables, as one moves from beaver impoundment to the coastal marsh to open water of Georgian Bay (see Table 2.4). Watershed-influenced variables such as TP, SRP, TAN, COL, and TSS all have concentrations that are highest in beaver-impounded marshes, intermediate in coastal marshes, and relatively low in open water, whereas lake-influenced variables such as TNN, SULPH, COND and pH are highest in open water, intermediate in the coastal marsh and lowest in the beaver impoundments. This was expected since TP, SRP, TAN, COL, and TSS are highly affected by the landscape elements of the catchment, such as the size and slope of the catchment as well as the amount of wetlands within it (Curtis & Schindler, 1997; Dillon & Molot, 2005; Eimer et al, 2008; deCatanzaro & Chow-Fraser, 2011; Table 2.4). By contrast, water of Georgian Bay has more alkaline pH, and higher levels of COND, SULPH and TNN because the southern and western portion of the Bay is composed of sedimentary rock that is easily weathered (Weiler, 1988; deCatanzaro & Chow-Fraser, 2011; Table 2.4). The significantly higher concentration of catchment-influenced chemicals in beaver impoundment is therefore a direct result of cessation of mixing between Georgian Bay and the marsh, and this highlights the importance of hydrologic connection between these water bodies for maintenance of the unique water-quality conditions in coastal marshes of eastern Georgian Bay (deCatanzaro and Chow-Fraser 2011).

Contrary to expectation, DOC was higher in coastal marshes than in beaver-impounded marshes, and lowest in open water. Mean values for DOC were obtained from a study by deCatanzaro & Chow-Fraser (2011) who chose only protected embayments with very narrow outlets. We speculate that the decreased connectivity to the lake may have prevented dilution from bay water. The amount of peatlands in the watersheds is a confounding factor that may have resulted in higher DOC values (Dillon & Molot, 1997) in marshes sampled by deCatanzaro & Chow-Fraser. Nevertheless, values in this study are consistent with those in the literature (Cirmo & Driscoll, 1993; Bledszki et al, 2011), and confirm that DOC originates from the watershed.

Fish and Beaver Dams

There are over 80 species of fish that utilize coastal wetlands during their life cycle (Jude & Pappas, 1992). Beaver dams created at the outlets of coastal marshes form a barrier to fish that prevent migratory movement (Schlosser & Kallemeyn, 2000; Collen & Gibson, 2001) and by implication, the use of these sites as spawning or nursery habitat. For example, dams may prevent northern pike (*Esox lucius*) from migrating into the wetland to spawn, and this is supported by evidence from a Wisconsin study conducted by Knuden (1962) who observed large numbers of pike lying below a beaver dam during spring. Important factors that influence the passage of fish through such barriers include flow conditions, dam characteristics and size and species of fish (Rossell et al, 2005). It may be possible for fish to move through dams by leaping over them or penetrating through interstices (Keast & Fox, 1990; Collen & Gibson, 2001), but the degree to which this may occur in Georgian Bay is not known and has not been studied. Given that our system generally has a weak hydrologic connection upstream, the source of the fish populations is highly dependent upon connectivity to open water. Any fish that manage to cross the barrier may ultimately succumb to summer or winter hypoxia if the impoundment is not sufficiently deep (Keast & Fox, 1990).

The dissolved oxygen content measured in the impoundments were generally low, with 80% of DO readings below 5.5 mg/L, which is considered a minimum level for protection of freshwater aquatic life by Canadian Water Quality Guidelines (CWQG) (McKibbin et al, 2008). Despite these low levels, species such as the *Lepomis gibossus* (pumpkinseed), *Culaea inconstans* (brook stickleback), *Ameiurus nebulosus* (brown bullhead) and *Umbra limi* (central mudminnow) were captured as by-catch during a larval amphibian survey. Similarly, Keast & Fox (1990) found 12 fish species in a small beaver pond in Ontario, including species that we found in this study. They determined that a low proportion of pumpkinseed adults were due to high annual mortality in the pond and that overall fish species richness was low as the population was mainly small-bodied fish relative to nearby lake environments. Future studies should examine the role of beaver dams on connectivity for fish habitat in Georgian Bay to determine if this is viable habitat.

Amphibians and Beaver Impoundments

In some jurisdictions, beaver-impounded wetlands are not considered permanent (see OMNR, 1993) because the dam may potentially fail, however, many beaver dams have persisted for decades and function like permanent habitats (see Babbitt, 2005; Stevens et al, 2007). In this study, we found tadpoles of green frogs consistently throughout beaver-impounded wetlands, while bullfrog tadpoles occurred in only a few sites but in high abundance. Both amphibians have a long developmental time; bullfrogs need two to three years, while green frogs need one to two years until metamorphosis (Tarr & Babbitt, 2008). Therefore, it is not surprising that these two species were found in the highest abundances as they require a breeding area that can support their longer development, but which also may expose them and other species of tadpoles to more predators. Previous research supports this and suggests that seasonal hydrologic permanence influences amphibian breeding assemblages (Babbitt et al, 2003). The longer developmental time of the bullfrog may make it more susceptible to winter anoxia as there needs to be two to three consecutive seasons of highly oxygenated water to ensure its presence. Green frog tadpoles may also be more adept at overwintering than bullfrog tadpoles and points to a gap in knowledge relating to the overwintering status of tadpoles.

A wide diversity of species was found breeding in beaver-impounded coastal marshes, indicating the importance of such habitat for amphibians, especially in the face of global declines. Others studies have pointed to the lack of habitat for amphibians without beaver modification (Cunningham et al, 2007). This suggests that future studies should determine the importance of beaver-impounded coastal marshes relative to connected coastal marshes as amphibian habitat in Georgian Bay.

Climate Change and Beaver Impoundments

Global climate change is predicted to lead to further declines in water levels of Georgian Bay and one estimate is as low as a 3-m decline by 2080 (Angel & Kunkel, 2010). This could create more favourable conditions for beaver-impoundments as the depths of marsh outlet continue to decrease. Additional climate change predictions suggest that the frequency and intensity of storms will increase between periods of drought (IPCC, 2008). Increased intensity of storms may result in discharge above the critical threshold value for a dam and thus cause it to collapse (Butler, 1989; Hillman, 1998; Butler & Malason, 2005). Butler (1989) concluded that three conditions, that can be generalized and applied to Georgian Bay, can lead to a dam failure. These include 1) a high amount of rain in a short period of time, 2) the presence of granite or other impervious surfaces, and 3) a relatively steep course of water flow (Butler, 1989). The slope of catchments will vary for each marsh, but the geology and future climate change predictions of intense storms, may create more favourable conditions for dam failure in Georgian Bay.

Little is known about the consequences of failure of beaver dams (Butler & Malanson, 2005). Bledzki et al (2011) found that after collapse of a dam in a stream environment, temperature and turbidity were increased downstream indicating potential stress to fauna and possible mortality. The consequences of a dam outburst in Georgian Bay are unknown and the severity will likely depend on the morphology and position of the impoundment. For example, broken impoundments with outlets that empty onto another coastal marsh would cause a sediment and nutrient pulse that could have negative effects on biota below, while the effects of impoundments with outlets that are highly exposed to Georgian Bay might be quickly dissipated. Regardless of morphology, organisms present behind the dam would be washed out from their environment and exposed to new stressors (Stanley & Doyle, 2003).

Conclusion

Hydrologic connections can drastically influence the water chemistry observed in coastal marshes. We have found that alterations of connectivity by a beaver can create significant differences in water chemistry by lowering the pH and amount of dissolved ions such as TNN and SULPH in the marsh. DO levels are also lower than if the marsh remained connected and this will have ecological consequences for species that utilize this habitat. Continued declines in water levels could increase habitat availability for beavers to dam additional coastal marshes. Scenarios with climate change predictions of increased storm frequency and intensity could impact beaver dams as they may be more susceptible to failure. Given that water quality in coastal marshes is an important determinant of fish and plant assemblages in wetlands of eastern Georgian Bay (Seilheimer & Chow-Fraser, 2006; Croft & Chow-Fraser, 2007) altered water chemistry in impoundments may affect their suitability as habitat for aquatic biota. Beaver-impoundments have been determined to be breeding habitat for seven species of amphibians, but little is known in regards to the importance of the impoundment in the creation and maintenance of amphibian habitat. Further studies should therefore examine the alterations in species assemblage as a result of impoundment as well as the consequence of potential dam failures. This study has been the first to examine how water chemistry in coastal marshes is altered by hydrologic disconnection due to the biotic influence of the beaver. If the current period of low water levels persists, we will likely see even greater dam-building activity throughout eastern Georgian Bay, and it is important that appropriate research be conducted to fully understand how the biotic assemblages are affected by these impoundments. Therefore, it is critical that we not only study the ecological effects of dam formation, but also the consequences of their failure because the coastal zone of eastern Georgian Bay is one of the most ecologically sensitive habitats in Lake Huron.

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Tables

Table 2.1: Mean water chemistry parameters measured above and below the dam at the beaver impounded coastal marshes. Bolded values indicate there was a significant difference as indicated by a paired t-test (n=17). Some parameters were log_{10}-transformed (indicated with *) or square-root transformed (indicated with **) before statistical tests were performed. The untransformed values are shown in the table.

Associated with	Parameter	Above Dam	Below Dam
Watershed	Total Phosphorus (µg/L)	30.18	25.53
	Turbidity* (NTU)	5.44	5.86
	Dissolved Organic Carbon*(mg/L)	13.48	12.33
	Soluble Reactive Phosphorus **(µg/L)	**13.33**	**12.96**
	Total Ammonia Nitrogen (µg/L)*	0.034	0.042
	Colour (mg/L Pt)	201.13	175.56
	Total Suspended Solids* (mg/L)	15.55	18.37
	Chlorophyll a* (µg/L)	6.23	5.39
Georgian Bay	**Sulphate* (mg/L)**	**0.54**	**2.02**
	Total Nitrate Nitrogen (µg/L)	0.026	0.032
	pH	**6.06**	**6.89**
	Dissolved Oxygen (mg/L)	**3.1**	**4.91**
	Conductivity(µS/cm)	**47.15**	**81.76**

Table 2.2: Spearman correlations between PCA axes and water chemistry variables.

Axis	% Explained Variance	Variable	Correlation coefficient
PC1	53.28	Soluble Reactive Phosphorus	0.78
		Chlorophyll a	0.82
		Total Suspended Solid	0.84
		Total Phosphorus	0.81
		Conductivity	-0.88
		Total Nitrate Nitrogen	-0.57
		pH	-0.69
PC2	12.96	Total Ammonia Nitrogen	-0.91

Table 2.3: The total number of each larval amphibian species found and the number of beaver impounded wetlands that each species was found at (n=17).

	Total Number of Sites Present	Total Number Found
Green frog	14	106
Spring Peeper	6	26
Bullfrog	5	141
Treefrog	4	14
Leopard Frog	3	3
American Toad	1	1
Chorus Frog	1	5

Table 2.4: Comparison of mean water chemistry values associated with open water in Georgian Bay, coastal marshes, and beaver-impounded coastal marshes.

Associated with	Parameter (mean values)	Open Water (n=11)[a]	Coastal Marsh (n=18)(n=33)[b]	Beaver Impound ed Marsh (n=15)
Watershed	Total Phosphorus (µg/L)	5.5	15.3	31.3
	Soluble Reactive Phosphorus (µg/L)	0.6	3.7	13.3
	Total Ammonia Nitrogen (µg/L)	8	14.7	36.5
	Dissolved Organic Carbon (mg/L)	2.5	16.6[b]	12.5
	Colour (mg/L Pt)	5	116[b]	205
	Total Suspended Solids (mg/L)	0.8	2.1	16.9
Lake	Total Nitrate Nitrite Nitrogen (µg/L)	225	38.3	25.7
	Sulphate (mg/L)	11.1	1.4[b]	0.5
	Specific Conductivity (µS/cm)	180	133	36.5
	pH	8.1	7.3	6

a- Offshore data taken from Table 1 in deCatanzaro & Chow-Fraser (2011)
b- taken from Table 1 in deCatanzaro & Chow-Fraser (20 11)

Figures

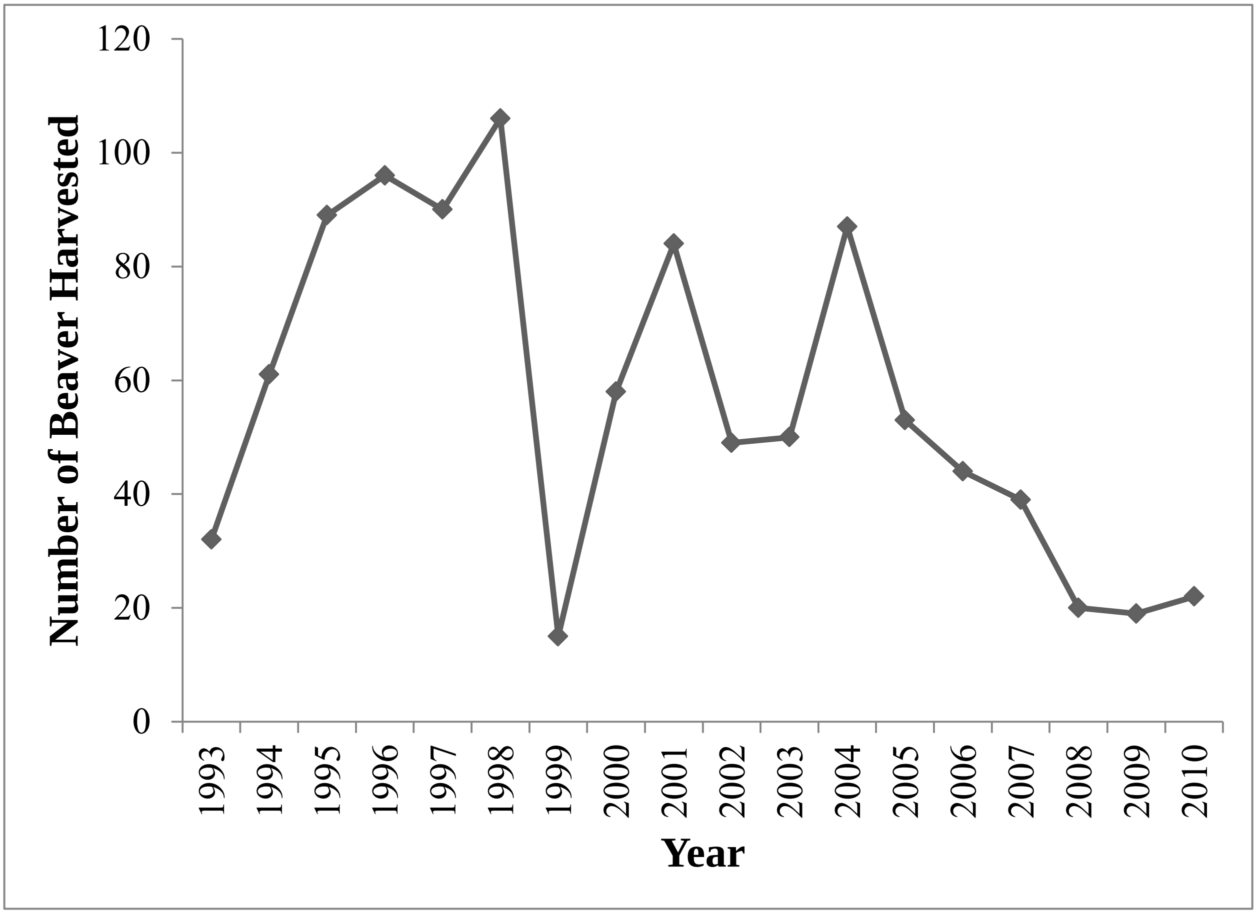

Figure 2.1: The number of beavers harvested in impoundments within the study area (Below Parry Sound to Severn River) from 1993 to 2010. Data provided by the Ontario Ministry of Natural Resources (OMNR, unpublished).

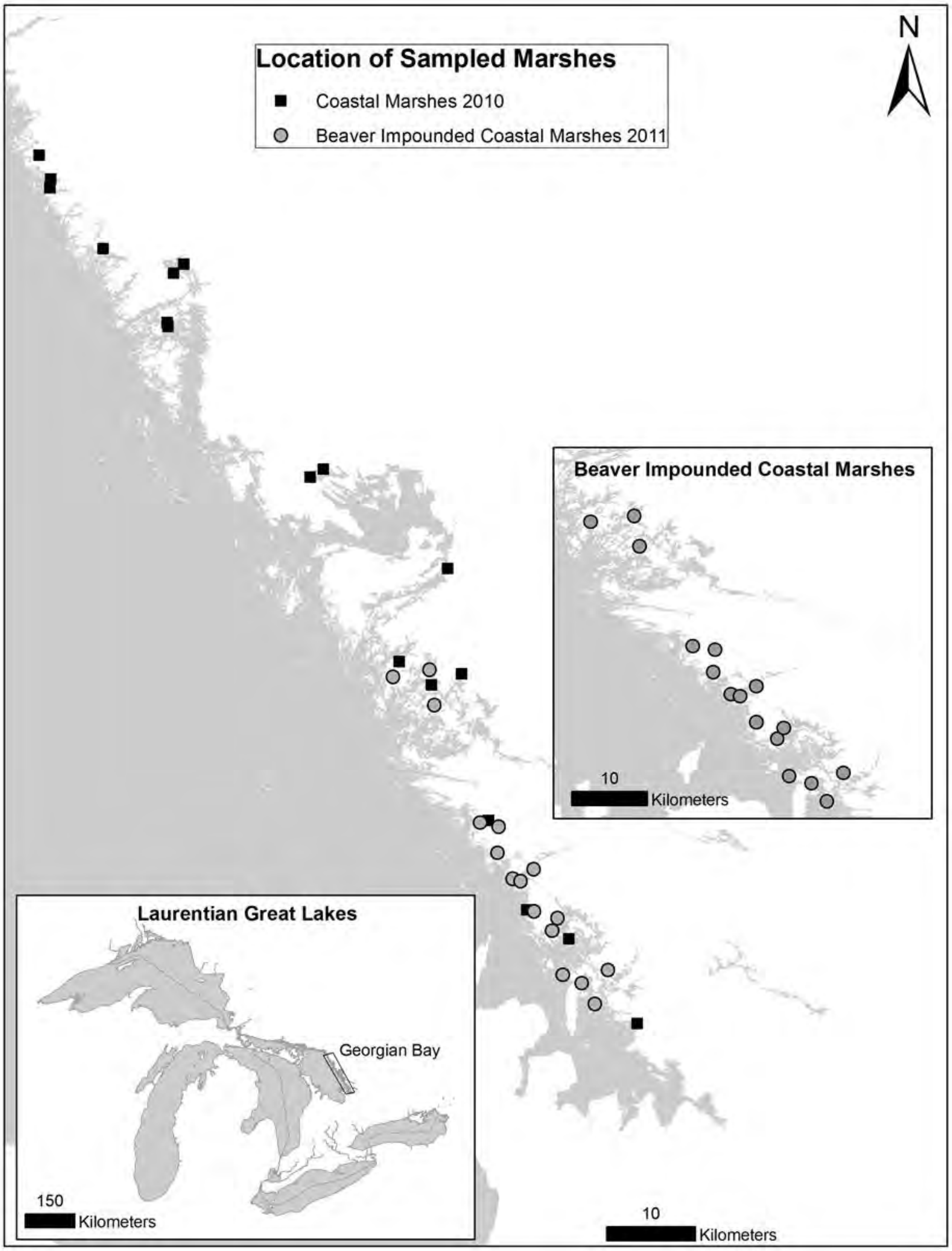

Figure 2.2: Sampling locations of coastal marshes sampled in 2010 (square symbols, n=18), beaver impounded coastal marshes (circle symbols, n=17).

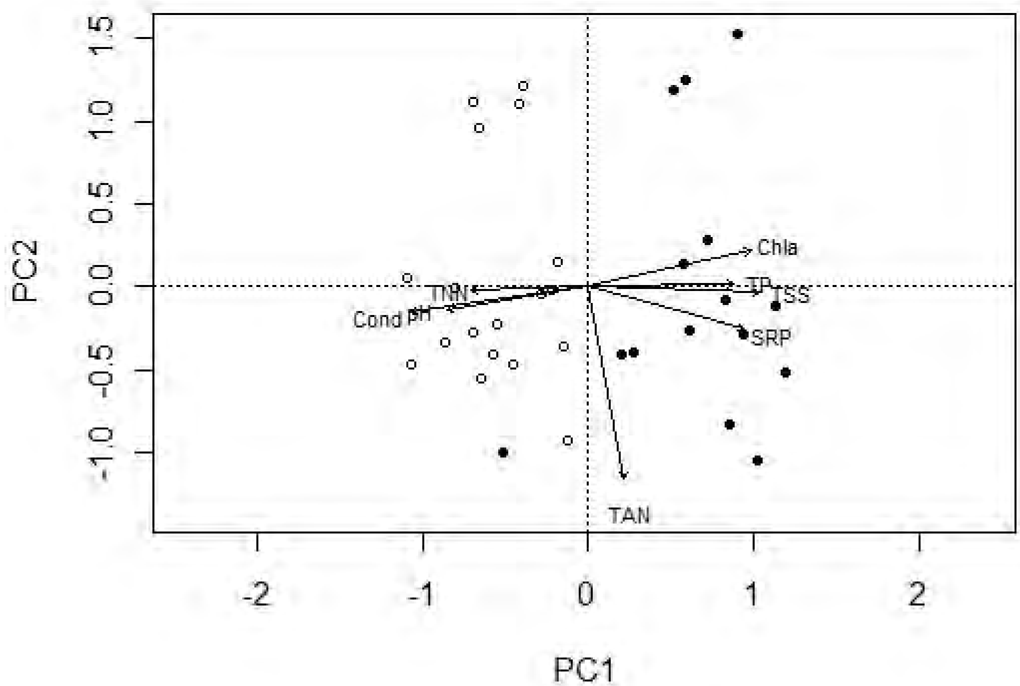

Figure 2.3: Principal Component Analysis (PCA) with impoundments (closed circles, n=15) and coastal marshes (open circles, n= 18). See Methods for explanation of abbreviations.

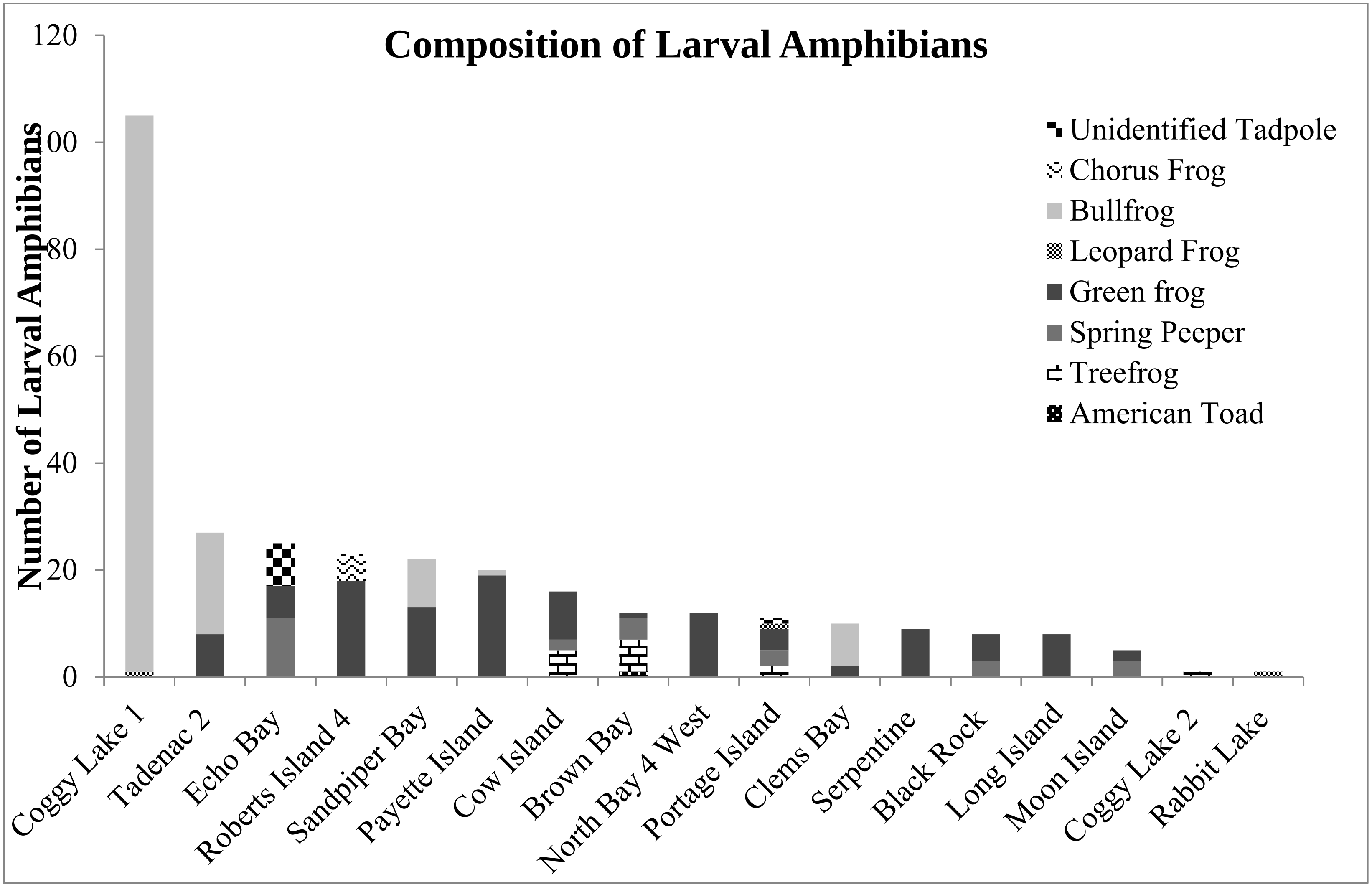

Figure 2.4: Composition of larval amphibian catch in beaver impounded coastal marshes (n=17).

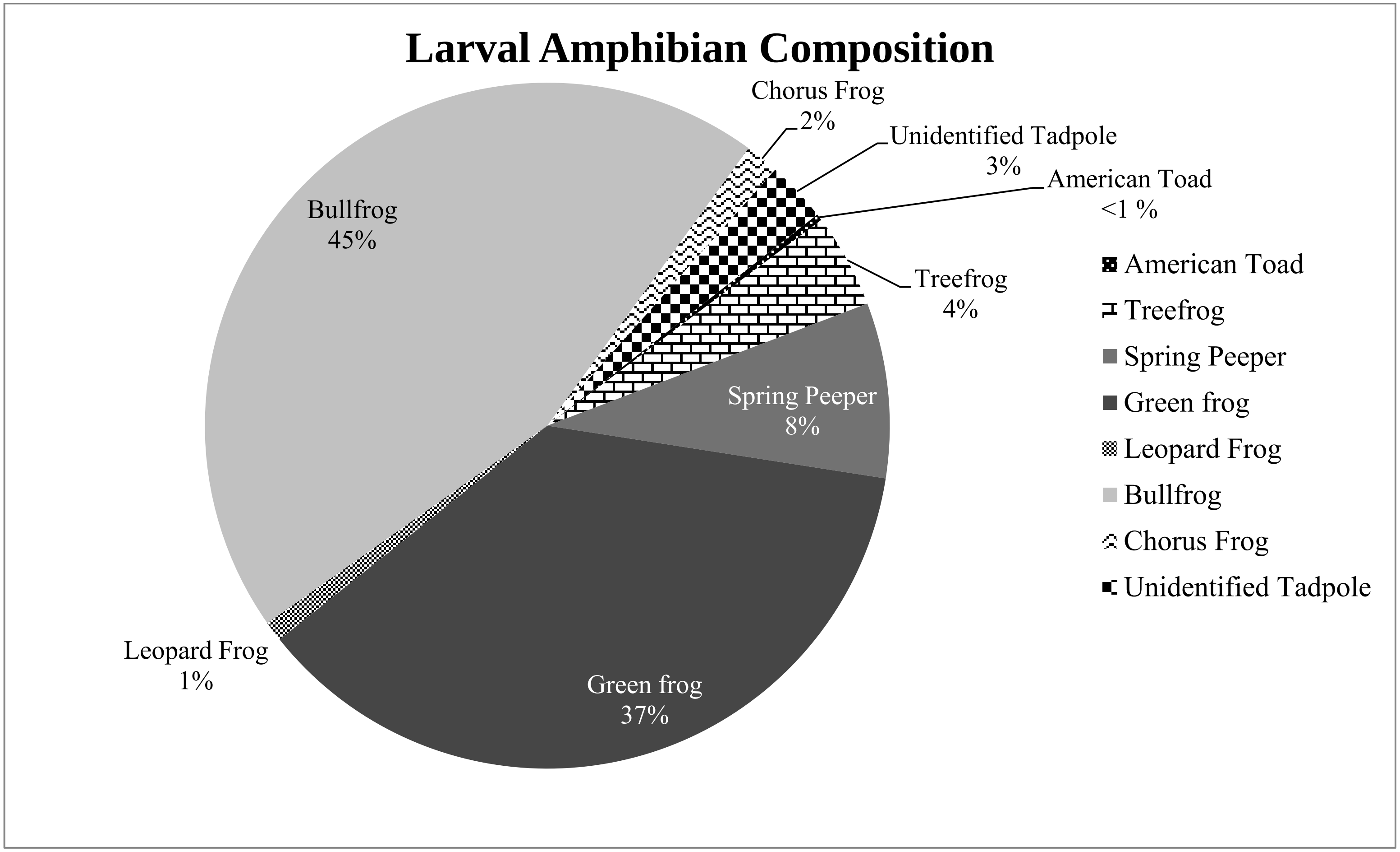

Figure 2.5: The composition of larval amphibians found in 17 beaver impounded coastal marshes. The percentage represents the total number of larval amphibians caught throughout the sampling season.

GENERAL CONCLUSION

The studies completed in this thesis examine the important role that connectivity plays within a coastal wetland ecosystem. Due to the complexity of studying hydrologic connectivity and large unknowns regarding human influence, current knowledge on hydrologic connectivity and its interactions with biodiversity are lacking (Pringle, 2003). The overall purpose and aim of this thesis was to fill some of those gaps of knowledge for coastal wetlands specifically within the region of eastern Georgian Bay.

With the decline in water levels in 1999 and the predicted declines that are forecasted by climate change scenarios (Angel & Kunkel, 2010), coastal wetlands are at an increased risk of being disconnected from Georgian Bay. Complete or partial disconnection, as measured by the regional and site-specific method (see Chapter 1), will reduce the amount of fish habitat that approximately 73 percent of fish species of Great Lakes utilize as nursery and spawning habitat (Jude & Pappas, 1992). Lower water levels and decreased connectivity can promote a more homogenous plant community within wetlands, and thus negatively influence the quantity and quality of fish habitat (Midwood & Chow-Fraser, 2012). In addition, reduced connectivity has been shown to decrease fish species richness (Bouvier et al, 2009) and can create conditions that are suitable to the expansion of invasive species such as *Phragmities* (see Tulbure et al, 2007). Due to the unique geology of the area, it is unlikely that wetland migration will occur with lower water levels and thus this habitat will essentially be lost. Detailed bathymetry such as that provided by LiDAR to create regional DEMs would be needed to be able to quantify the extent of wetland migration towards the lake.

A decrease in hydrologic connection can also alter the water chemistry observed in coastal wetlands. By examining disconnection created by beaver dams, significant differences in water chemistry were observed such as lowering of the amount of sulphate, conductivity, pH and dissolved oxygen (Chapter 2; Table 2.1). These beaver-impounded coastal wetlands with severed connected to Georgian Bay had higher levels of phosphorus and suspended solids than wetlands that were connected with Georgian Bay, which had water chemistry that was more representative of Bay water (Figure 2.3). Beaver impoundments were also found to be breeding habitat for 7 species of amphibians. It is unknown how this would compare to connected coastal wetlands and further research should examine changes in biota between impounded and open ecosystems. Declines in water levels will likely increase the amount of available habitat for beavers to dam. In addition, they may build new dams to maintain connectivity to their current habitat. Conversely, changes in climate with increased storm frequency and intensity may also make beaver dams more susceptible to failure. Therefore, it is important that we understand the ecological changes in dam formation as well as the result of their failure on coastal wetlands.

Overall, the results of modelling potential fish habitat with lower water levels (Chapter 1) is the first study of its kind to cover and be representative of such as large geographic area. It quantitatively shows the amount of habitat available and connected with varying water levels and describes potential ecological effects. Chapter 2 furthers scientific knowledge regarding natural water chemistry alterations in an environment that has been minimally impacted by human influences. It has shown that hydrologic disconnection created by a biotic factor such as a beaver will alter wetland water chemistry and raises questions on how this may affect other biota and the stability of the habitat in the future.

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