ASSESSING CHANGE IN FISH HABITAT AND COMMUNITIES IN COASTAL WETLANDS

ASSESSING CHANGE IN FISH HABITAT AND COMMUNITIES IN COASTAL WETLANDS OF GEORGIAN BAY

By

JONATHAN DALE MIDWOOD, B.Sc.

A Thesis
Submitted to the School of Graduate Studies
in Partial Fulfillment of the Requirements
for the Degree
Doctor of Philosophy

McMaster University

Doctor of Philosophy (2012) McMaster University

(Biology) Hamilton, Ontario

TITLE: Assessing change in fish habitat and communities in coastal wetlands of Georgian Bay

AUTHOR: Jonathan Dale Midwood, B.Sc. (McMaster University)

SUPERVISOR: Professor Patricia Chow-Fraser

NUMBER OF PAGES: xvii, 280

PREFACE

The following Ph.D. thesis is comprised of five chapters that form separate manuscripts for publication in peer-reviewed journals. A General Introduction is provided to put this research into context. Complete references for all chapters that have been published or are in submission can be found below. *Chapter 1* has been published in *International Scholarly Research Network (ISRN) Ecology*, *Chapter 2* has been published in *Wetlands*, *Chapter 3* is in submission to *the Journal of Great Lakes Research*, *Chapter 4* has been published in *Global Change Biology*, and finally, *Chapter 5* is presented as a manuscript, but has not yet been submitted for publication. With the exception of *Chapter 1*, as first author on these manuscripts, I compiled and analyzed the data and wrote all the manuscripts, under the supervision of Pat Chow-Fraser. Part of the analysis and writing for *Chapter 1* was done by Daniel Rokitnicki-Wojcik, hence he was included as a co-author on this manuscript. For collection of data in the field, I am indebted to numerous graduate and undergraduate students who are more formally recognized in the acknowledgements section.

Midwood, J.D., Rokitnicki-Wojcik, D. & Chow-Fraser, P. 2012. Development of an inventory of coastal wetlands for eastern Georgian Bay, Lake Huron.

ISRN Ecology vol. 2012, Article ID 950173, 13 pages. doi:10.5402/2012/950173.

Midwood, J.D. & Chow-Fraser, P. 2010. Mapping floating and emergent aquatic vegetation in coastal wetlands of eastern Georgian Bay, Lake Huron, Canada. Wetlands 30(6):1141-1152.

Midwood, J.D. & Chow-Fraser, P. Predicting the response of submerged aquatic vegetation to low water levels in coastal wetlands of Georgian Bay, Lake Huron. (In submission, Journal of Great Lakes Research).

Midwood, J.D. & Chow-Fraser, P. 2012. Changes in aquatic vegetation and fish communities following five years of sustained low water levels in coastal marshes of eastern Georgian Bay, Lake Huron. Global Change Biology 18:93-105.

Midwood, J.D. & Chow-Fraser, P. Complexing coastal marshes of eastern Georgian Bay using movements of resident and migratory fishes.

GENERAL ABSTRACT

Aquatic vegetation in the pristine coastal marshes of eastern Georgian Bay (GB) provides critical spawning and foraging habitat for fish species, with complex habitat supporting the greatest diversity. These wetlands are threatened by a changing water level regime and forecasted lower water levels. To monitor and conserve these wetlands, we must understand how they function and respond to this stressor. The overall goals of this thesis are to determine the impact of declining water levels on both wetland fish habitat and the fish community as well as identify the spatial scale of habitat utilization by fishes.

We first delineate all coastal wetlands in eastern GB, identifying 3771 wetlands that provide habitat for Great Lakes fishes. Using satellite imagery, we develop an object-based classification method to classify four types of wetland vegetation. Since submerged aquatic vegetation (SAV) is not visible from satellite imagery in GB, we develop a model to predict potential area of this important habitat. The model suggests that the response of SAV to declining water levels depends on wetland geomorphology, but generally, the area of SAV decreases. To assess the response of fish habitat coverage and structure to sustained low-water levels, we classify vegetation in images collected in 2002 and 2008. The result is increasingly homogeneous habitat, a net loss of fish habitat and a decrease in fish species richness. Finally, mark-recapture and radio-tracking are used to evaluate fish movement among closely situated wetlands. Results suggest that the current distance used to group and protect small wetlands provincially (750 m), likely protects most resident fish species, but does not cover movement patterns of a top predator. This research will advance our scientific understanding of freshwater coastal ecosystems and aid in the creation of conservation strategies to mitigate future threats from declining water levels.

ACKNOWLEDGMENTS

First and foremost I must thank Dr. Pat Chow-Fraser who has mentored me these past five years. Her encouragement and enthusiasm have made all aspects of my graduate degree a true joy. Pat's relentless passion for conservation serves as a great source of inspiration for all of her students and I feel lucky to have been a part of the lab. One day I will acquire a taste for the good stuff, cheers!

I greatly appreciate the support I received from Dr. Jon Stone and Dr. Jonathan Dushoff. As members of my committee they have both helped greatly to shape and create the thesis before you. This thesis also would never have happened without the constant support of Mary Muter. She introduced me to the wonders of Georgian Bay and she continues to be its most passionate advocate.

Through the Chow-Fraser lab I have been fortunate to work with a large group of intelligent, kind, and passionate people. I've also been around so long that many have come and gone. Thank you to Mel Croft, Anhua Wei, and Titus Seilheimer who originally introduced me to wetland vegetation, remote sensing, and fish. Thank you to Dan Rokitnicki-Wojcik, with whom I struggled to solve the mysteries of satellite image interpretation. I also owe a great deal of thanks to Rachel deCatanzaro, April Stevens, Steph Yantsis, Sarah Thomasen, Catherine Dieleman, and Amanda Fracz for making field trips so enjoyable and teaching me about all things water quality. Thank you to Lyndsay for introducing me to the world of birding and also for being a constant source of support and guidance throughout my thesis. Thank you to Bob Christensen and JP Leblanc, who have made the end of my studies so enjoyable and who will both one-day produce amazing PhD theses!

There are many people I have not named here specifically but who have made very important contributions to my research in some way over the past years. There are far too many to thank individually, so to all of the undergraduate summer students, supportive members of GBA, and volunteers who assisted us in the field, I owe you all a giant thank you!

Thank you to the Phoenix for being ever so close and always having such delicious cold beer. Thank you to Henry Haiser, for being such a great friend through all of this, I look forward to one day figuring out how we can collaborate.

While they may not understand exactly what it is I do, my family has been so supportive through all of my graduate work, so hugs for everyone.

Finally, thank you Maja Cvetkovic, who is not only my best friend but also my favourite wetland elf. Without her love and support, none of this would have been possible.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

GENERAL INTRODUCTION

Wetlands

The term "wetland" refers to a diverse group of ecosystems that permanently, periodically, or seasonally contains standing water or saturated soil. Abiotic variables, such as hydrologic regime and water chemistry, that influence the development of unique types of vegetation or peat, further differentiate wetland types into: bogs, fens, marshes, swamps, and shallow open water (Finlayson and van der Valk 1995; Zoltai and Vitt 1995; NWWG 1997). While definitions of both wetlands in general and more specific types of wetlands are often contentious, in Canada, the National Wetlands Working Group has attempted to standardize some of the most common wetland types (NWWG 1997).

Fens are largely influenced by the chemistry of their surrounding soils because they are wetted by either groundwater or runoff. Their waters tend to be neutral or basic, with bryophytes (e.g., mosses and worts) the dominant form of vegetation. By comparison, bogs are typically acidic. This is due to minimal inflow and outflow of water except through precipitation and evaporation. This results in high levels of decomposition creating humic acid and water with a low pH. *Sphagnum* is the dominant vegetation in bogs where it often forms floating mats (Zoltai and Vitt 1995; NWWG 1997).

Swamps and marshes are similar because they are strongly influenced by water-level fluctuations, with the main distinction being the presence of trees in swamps. These trees can prevent water flow, allowing decomposition to occur in some swamps, resulting in acidification. Relatively continuous water flows in marshes allow vascular plants to thrive despite high rates of decomposition. Wetlands in the final category, shallow open water, occur when waters are consistently deep enough to support both aquatic and floating vegetation. These wetlands often occur along lake margins and consequently are highly influenced by lake properties and dynamics (Zoltai and Vitt 1995; NWWG 1997). For this study, we focused on coastal marshes, which incorporate the marsh and shallow open water categories from the NWWG (1997) as well as portions of the littoral zone of a lake (up to 6 m in depth). Li and Chen (2005) provide the most appropriate definition for coastal wetlands examined in this thesis: coastal wetlands are hydrologically connected to the Laurentian Great Lakes and their vegetation community is dominated by floating and submerged macrophytes.

While there is great diversity among the different types of wetlands, as a group, they are some of the most biologically diverse ecosystems on the planet (Jones et al. 2008). The duration of saturation, coupled with their transitional position between terrestrial and aquatic habitats encourages high levels of biodiversity (Finlayson and van der Valk 1995; Zoltai and Vitt 1995). Wetlands also provide a disproportionally large number of ecosystem services including: water purification, nutrient sequestration, and shoreline buffering (Costanza et al. 1997; Jones et al. 2008; Keddy et al. 2009). This thesis will focus on the importance of wetlands in providing critical habitat (spawning, staging, mating, nursery, hibernation, prey) for a wide variety of organisms including birds, invertebrates, mammals, reptiles, and most importantly, fish (Chow-Fraser and Albert 1999; Wei et al. 2004; Smith-Cartwright and Chow-Fraser 2011).

Regardless of their importance in providing ecosystem services and habitat for wildlife, coastal wetlands have been highly impacted by human development. Coastal wetlands form naturally in shallow, protected embayments and these areas are also ideal for building houses, cottages and marinas. In southern Ontario, 75% of wetlands have been lost since colonial settlement (Snell 1987; Findlay and Houlahan 1997). The majority of the remaining coastal wetlands have been severely altered or impacted by human development (Chow-Fraser 2006). In their seminal paper, Costanza et al. (1997) suggested that the cost to replace the natural services provided by wetlands could be as much as $14,785 per hectare. This highlights the need for conservation and protection of coastal wetlands that remain in our natural landscape.

In Ontario, the best source of protection for wetlands is through the Provincial Policy Statement (PPS) under Section 3 of the Planning Act (PPS 2005). Under this act, wetlands that are deemed “provincially significant” through the Ontario Wetland Evaluation System (OWES) are afforded the highest level of protection (OMNR 1993). This system ranks wetlands by their hydrologic, social, and biological contributions. Wetlands that score highly in all categories, contain high biodiversity, or contain unique species and habitat are designated a "provincially significant wetland". In order for a wetland to be evaluated, it must be at least 2 hectares in size. Smaller wetlands may be evaluated if they are found to be part of a larger, regional complex of wetlands. Inclusion of these smaller wetlands is left to the discretion of the biologist conducting the evaluation, provided that the wetlands fall within 750 m of each other (OMNR 1993). The OWES and PPS are currently under review and we hope the findings of this research will facilitate the creation of new standards for evaluation and designation.

The Great Lakes and Georgian Bay Marshes

Formed during the last ice age, the Laurentian Great Lakes represent the largest freshwater resource in the world (Figure 1; Herdendorf 2004). Within these lakes it is estimated that there are well over 1500 large coastal wetlands that are >25 ha in size with a total surface area of approximately 1700 km^2 (Herdendorf 2004). The drainage basin of the Great Lakes is home to over 33 million people (23.8 in the U.S. and 9.2 in Canada; Mayer et al. 2004). As a result of this large population, over 45% of the land has been altered for urban, agricultural, or commercial purposes (Mayer et al. 2004).

Several studies have documented the response of coastal wetland water quality to human development and land-use alterations throughout the Great

Lakes basin (Chow-Fraser 2006; Danz et al. 2007; Trebitz et al. 2007). Increasing development consistently leads to a loss of natural land cover in a wetland's drainage basin and subsequent declines in both wetland water quality and habitat. The coastline of eastern and northern Georgian Bay is one of a few remaining regions with minimal human disturbance, and consequently it contains some of the healthiest wetlands in the entire Great Lakes basin (Chow-Fraser 2006; Cvetkovic and Chow-Fraser 2011).

Georgian Bay is located in the eastern basin of Lake Huron and has a total area of 16,300 km^2. The extensive and complex shoreline of Georgian Bay makes it an ideal location for the formation of coastal wetlands (Wei et al. 2004). Wetlands here have formed on the weather-resistant granite of the Canadian Precambrian Shield, and consequently their waters are typically dystrophic with low nutrient levels (DeCatanzaro and Chow-Fraser 2011). Surveys of these wetlands have found high levels of diversity for both macrophytes and fishes (Croft and Chow-Fraser 2007; Seilheimer and Chow-Fraser 2007). Within the Great Lakes basin, the coastal wetlands of Georgian Bay can be considered reference conditions for the Great Lakes due to limited human disturbance and high biological diversity that is driven primarily by variation in the natural environment (EPA 2008). As a contracting party of the Ramsar Convention, Canada has an obligation to catalogue and monitor all of its wetlands (Jones et al. 2008). An important component of effective wetland conservation and management is to develop a comprehensive inventory of the wetlands and develop an understanding of the environmental processes that control their formation, distribution, and persistence (Finlayson and van der Valk 1995). Currently, a complete and comprehensive inventory of coastal wetlands is lacking for Georgian Bay (Ingram et al. 2004).

Georgian Bay is the World's largest freshwater archipelago, and in 2004 was named a UNESCO World Biosphere Reserve. While this affords Georgian Bay some level of recognition, it is still under threat due to climate change, invasive species, consistently low water levels, development pressures, and boating impacts (GBBR 2012). For this thesis, I focused on the potential impacts of climate change and resulting impacts of changing water levels on coastal wetlands of Georgian Bay.

Effects of Climate Change – Water Levels

Changing water levels are essential to the health and functioning of coastal marshes because they prevent one species of vegetation from becoming dominant (Wilcox and Meeker 1991). The natural cycle of high and low water levels has been disturbed in the Great Lakes through regulation (Quinn 2002). Not only does regulation allow monocultures to persist (Wilcox and Meeker 1991), but when it results in low water level, this allows terrestrial plants to colonize previously aquatic habitat (Hudon 2004). Climate-change scenarios predict increasing water demand for municipal, industrial, and agricultural use, as well as an increase in evaporation, all of which will act to lower water levels (Meyer et al. 1999). Climate change models predict that Great Lakes water levels will drop by 0.2 to 2.5 m over the next 50 years (Mortsch and Quinn 1996; Magnuson et al. 1997; Sellinger et al. 2008; Angel and Kunkel 2010). Lower water levels, compression or loss of natural water level cycles and increasing temperatures may cause a loss of wetland habitat and changes in vegetation species (Meyer et al. 1999).

Coastal wetlands are naturally dynamic systems, where a diversity of terrestrial and aquatic biota have alternated their dominance according to the natural 7-10 yr cycles of water-level fluctuations in the Great Lakes; in high-water years terrestrial vegetation dies, and in low-water years aquatic vegetation disappears (Keddy and Reznicek 1986). Without interannual water-level variation, either the aquatic or the terrestrial vegetation dominates at the expense of the other. Since 1999, water levels in Lake Huron have been relatively stable and close to their historic low (Figure 2). Jude et al. (2005) noted that despite the clear threat from lower water levels, little research had been done on the potential impact on Great Lake coastal wetlands. It is therefore important to determine how sustained-low water levels will impact aquatic habitat in coastal wetlands.

Aquatic Vegetation

Herbaceous vegetation is an integral component of marshes and this broad classification can be further subdivided into emergent, floating, and submerged plants. Emergent species have leaves and flowering parts emerging from the water, floating species have leaves and flowering parts lying on top of the water, and submerged plants have leaves and stems entirely under the water (Newmaster et al. 1997; Chadde 2011). Aquatic macrophytes, along with algae, are the primary producers in aquatic ecosystems, trapping the sun's energy and making it available for other species. These macrophytes have developed special adaptations to survive in dynamic wetland environments, which are prone to both drawdown and flooding. Unlike most terrestrial vegetation, which primarily relies on sexual reproduction, aquatic macrophytes typically reproduce asexually from plant fragments or parts of their rhizomes (Sawada et al. 2003). They also form overwintering buds called turions, which sink to the bottom when the water freezes and are capable of surviving droughts and low temperatures before rising again in spring. Some remain in the wetland until favourable conditions return, and others colonize distant habitats by floating in currents or hitching a ride on boats, birds, and mammals.

Aquatic plants are the component of wetlands that facilitate or produce the numerous wetland services afforded by wetlands. They help to purify water by decreasing turbidity through the stabilization of sediments, thus reducing resuspension of fine particles. In a similar fashion, they limit erosion along the margins of lakes and rivers while also reducing the inflow of nutrients from the watershed (Madsen et al. 2001; Lacoul and Freedman 2006). Finally, aquatic vegetation plays a central role in supporting high levels of biodiversity by oxygenating the water column and providing food and shelter for a wide variety of invertebrates, shellfish, birds, and fishes (Jude and Pappas 1992; Costanza et al. 1997; Wei et al. 2004). Fish are known to preferentially utilize vegetated areas over non-vegetated areas (Jude and Pappas 1992; Randall et al. 1997) and it is this provision and maintenance of fish habitat by aquatic macrophytes that drives research focused on understanding how macrophytes respond to changing water levels.

Importance of Fishes in the Great Lakes

Fishes are very important from a cultural perspective since they are often the main source of protein and/or financial income (Arlinghaus et al. 2002). Globally, the fishery industry provides a livelihood for over 44 million people and when processing and distribution are included, these numbers increase to hundreds of millions (FAO 2010). On the Ontario side of the Great Lakes, commercial fisheries bring in approximately 14,808 metric tonnes annually at a value of between $180 and $215 million dollars (OMNR 2010). Historically in the Great Lakes, commercial fishing was the most common and profitable link between people and fish communities; it has since been eclipsed by the recreational fishery industry. The Great Lakes are considered to be among one of the best places in the world for recreational freshwater fishing. In Canada, over 25% of all recreational fishing occurs in the Great Lakes, contributing over $350 million to the economy each year (OMNR 2010).

While both recreational and commercial fisheries provide important sources of income and sustenance, it has long been thought that commercial fisheries have a far greater negative impact on fish populations. Based on this assumption, the transition from a primarily commercial fishery to a recreational one would seem to benefit the natural fish stocks. Unfortunately, there is increasing evidence that both forms have a negative impact on fish stocks, and therefore proper regulation of both commercial and recreational fisheries is crucial if natural populations are to be maintained (Post et al. 2000; Cooke and Cowx 2006). A strong understanding of factors that maintain fish communities and stocks is critical for establishing appropriate management strategies. Since habitat is of critical importance for maintaining healthy fish communities, this thesis focused on providing a better understanding of the dynamics of fish habitat as well as the scale of habitat selection by different fishes.

Fish can be found globally in a wide variety of habitats. This wide distribution coupled with diverse and potentially extreme environments has facilitated evolutionary development and novel adaptations such that there are more fish species than all other vertebrates combined (Powers 1989). As a result of this prevalence and diversity, they are some of the most heavily studied species in the world. From a research perspective, fish are ideal model organisms because of their unique adaptations, high fecundity, visible eggs, and successful propagation in laboratories. This has led to their use in fields as varied as developmental biology, endocrinology, neurobiology, embryology, toxicology, and environmental biology (Powers 1989). From a more ecological perspective, piscivorous fishes are typically the top aquatic predators in freshwater systems, providing important top-down control in these systems (Jackson et al. 2001; Craig 2008). Their dependence on water prevents extensive range expansion and can drive local variations in community structure. Finally, since range expansion is limited, they respond readily to local environmental changes caused both naturally and by human disturbances (Brazner and Beals 1997; Chow-Fraser et al. 1998; Seilheimer and Chow-Fraser 2006; Trebitz et al. 2009).

Within the Great Lakes, all fish diversity (with the exception of introduced species) represents those species that were able to find suitable refuge during the last glacial period (Bailey and Smith 1981). This has resulted in relatively low diversity given the size of the aquatic system (Scott and Crossman 1998). Across Ontario, 132 different species have been identified. From this group, 92 are known to occur in Lake Huron (Scott and Crossman 1998; GLFC 1995 [Lake

Huron Fish Community Objectives]) and, of these 58 have been recorded in the coastal wetlands of Georgian Bay (Chow-Fraser, unpublished data).

Complex aquatic habitats support the highest levels of fish diversity and in coastal marshes aquatic macrophytes provide this structure (reviewed in Smokorowski and Pratt 2007). Fish utilize aquatic vegetation for a variety of purposes including spawning and nursery habitat, refuge from predators, shade and cooler temperatures, and as a substrate to support food sources (Jude and Pappas 1992; Weaver et al. 1997; Höök et al. 2001; Smokorowski and Pratt 2007). While strong linkages have been demonstrated for some common wetland fishes and specific aquatic plants or plant groups (Killgore et al. 1989; Mundahl et al. 1998; Jacobus and Ivan 2005; Cvetkovic 2008), it is unclear how individual species and the community as a whole will respond to changes in the provision of habitat by aquatic macrophytes.

Ficke et al. (2007) identified changes to hydrologic regimes as a major mechanism through which climate change will impact global fisheries. Since current low-water levels are a newly identified stressor in the Great Lakes, few studies have attempted to establish the response of the fish community to these changes. One notable exception is the work of Webb (2008) who found no change in the coastal fish community in response to 9 years of lowering water. A major issue of contention for this paper is that Webb did not report how or whether habitat had changed during this time period. Cvetkovic et al. (2010) demonstrated that fish respond more readily to changes in the plant community than they do to changes in water quality. While water quality is not necessarily analogous to low water levels, conceptually the findings of Cvetkovic et al. (2010) would suggest that to properly elucidate the response of the fish community, it is necessary to link fish community changes not to changes in water level, but instead to changes in aquatic vegetation. Therefore, given the strong relationship between vegetation and the fish community, we will expand on the work done by Webb (2008) to determine the effects of low water levels on the fish community by measuring vegetation changes.

Remote Sensing and Geographic Information Systems (GIS)

Remote sensing and GIS are excellent tools that allow researchers to analyze ecological data at a landscape level. In general terms, remote sensing is the process of collecting information by using a sensor that is not in direct contact with the object of interest (Lillesand et al. 2004). In the context of this thesis, remote sensing refers to the acquisition of pictures of the Earth's surface from satellite sensors. These sensors collect the amount of sunlight in different wavelengths that is reflected by objects on the ground. Using remotely sensed imagery, researchers can collect data on a plethora of environmental variables such as vegetative cover and biomass, water chemistry, and the amount of human development (Lehmann and Lachavanne 1997; Jones et al. 2008). By collecting a time-series of images covering years or different seasons, changes in land cover can also be assessed (see Leahy et al. 2005).

Geographic Information Systems provide a framework where spatial environmental data, such as that collected using remote sensing, can be combined with species data. With this tool, data can be overlaid and analyzed to link species to their habitat, delineate movement patterns, identify potential sources of disturbance, and model future changes (Lehmann and Lachavanne 1997). Both methods have been used extensively for wetland habitat delineation, monitoring, and evaluation (OWES 1993; Jones et al. 2008).

The wide spatial distribution of Georgian Bay coastal wetlands, along with the difficulty in accessing these wetlands, makes remote sensing the only feasible method to accurately map and monitor these wetlands. In this study we utilized the IKONOS satellite, which collects data in four distinct wavelengths or "spectral bands". The red (632-698 nm), green (505-595 nm) and blue (445-516 nm) bands form our visible spectrum and the fourth band covers the near-infrared range (757-853 nm; Lillesand et al. 2004). When the satellite captures each image or "scene", all four bands are broken up into square pixels 4 m by 4 m in size. The pixel size or "resolution" can be increased to 1 m^2 following the acquisition of the image. Platforms that are capable of acquiring imagery in the meter to sub-meter range are generally considered to be "high-resolution" (Lillesand et al. 2004). Each of the four spectral bands is broken into pixels and the satellite assigns each pixel a digital number (DN) for each band. The DN represents the amount of light reflected by that 1-m^2 pixel. By using variations in DN among bands different types of land cover can be identified including water, rocks, and different types of vegetation.

The high resolution of IKONOS imagery coupled with its four distinct spectral bands make it a useful tool for wetland vegetation delineation. Several groups have utilized IKONOS satellite imagery to map wetlands (Olmanson et al. 2002; Fuller et al. 2006; Wei and Chow-Fraser 2007; Roktinicki-Wojcik et al. 2011). Similarly, Wei and Chow-Fraser (2007) used IKONOS imagery and a supervised classification in Georgian Bay and Lake Huron coastal wetlands to separate vegetation into floating, emergent, and in some cases submerged vegetation. In a supervised classification, a technician selects representative pixels for the desired classes and the computer then uses this information to classify the entire image. A drawback to the supervised classification method is that it requires initial ground control points in order to classify each image. A regionally applicable approach would minimize the need for extensive fieldwork, thereby reducing cost and time.

Most studies discussed thus far have classified images at the pixel level. Based upon their unique spectral properties, similar pixels are grouped together according to rules laid out by the operator or by a predefined algorithm. A new method (object-based classification) has emerged that clusters similar pixels together and allows the operator to work with the remaining object. This object represents a combination of all the properties associated with each pixel that is found within it (Chuber et al. 2006). This new method accounts for the spatial heterogeneity of most wetlands since it minimizes extremely high or low DN, which may affect a regional classification scheme (Fournier et al. 2007; Grenier et al. 2007). It also provides more spatial information than pixel-based classification such as object area, length, shape, nearest neighbours, and texture (Navular 2007). This novel approach has yet to be applied to habitat mapping in coastal environments.

Thesis Objectives

The primary objectives of this thesis were to provide a better understanding of the dynamics of fish habitat in response to changing water levels and to determine the scale of habitat selection by fishes. Towards these goals, in Chapter 1 we developed a comprehensive inventory of coastal wetlands in eastern Georgian Bay using IKONOS satellite imagery and manual delineation. This represents the first complete inventory and should allow future studies to properly subsample Georgian Bay wetlands.

In order to facilitate future wetland habitat-mapping initiatives, in Chapter 2 we developed a regionally applicable object-based method to classify dominant coastal wetland vegetation types using high-resolution satellite imagery. Habitat classifications based on satellite imagery could not map submerged aquatic vegetation in Georgian Bay coastal wetlands. To map this important component of fish habitat, in Chapter 3 we modeled the response of submerged aquatic vegetation to lower water levels using depth and exposure.

Using a slight modification of the classification method developed in Chapter 2, in Chapter 4 we acquired more recent IKONOS imagery in order to perform a change detection analysis and determine the impact of sustained low-water levels on coastal wetland emergent and floating vegetation. We also examined fish community data that were collected concurrently to determine if there were also changes to the fish community.

In Chapter 5 we addressed the movement of fish among coastal wetlands to determine if the numerous, small wetlands identified in our inventory are in fact operating as a collection or complex of proximate coastal wetlands. As a whole, the research outlined in this thesis presents novel methods for delineating and mapping coastal wetland habitat, an assessment of the impact of low-water levels on fish habitat and communities, and new insight into habitat utilization by the coastal fish community. This work will not only advance our knowledge of coastal wetlands and their dependent fish community, it will also help to inform future management strategies and policy.

References

Angel JR, Kunkel KE (2010) The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. Journal of Great Lakes Research 36: 51–58.

Arlinghaus R, Mehner T, Cowx IG (2002) Reconciling traditional inland fisheries management and sustainability in industrialized countries, with emphasis on Europe. Fish and Fisheries 3: 261–316.

Bailey RM, Smith GR (1981) Origin and geography of the fish fauna of the Laurentian Great Lakes basin. Canadian Journal of Fisheries and Aquatic Sciences 38: 1539–1561.

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

Chadde S (2011) A Great Lakes wetland flora: a complete guide to the aquatic and wetland plants of the upper Midwest. Self Published. ISBN: 978-1456484965.

Chow-Fraser P, Albert D (1999) Identification of ecoreaches of Great Lakes coastal wetlands that have high biodiversity value. State of the Lakes Conference: 1–107.

Chow-Fraser P (2006) Development of the Water Quality Index (WQI) to Assess Effects of Basin-wide Land-use Alteration on Coastal Marshes of the Laurentian Great Lakes. In: Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Authorhouse, Bloomington, pgs 137–166.

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

Chuber MS, Franklin SE, Wulder MA (2006) Object-based analysis of Ikonos-2 imagery for extraction of forest inventory parameters. Photogrammetric Engineering & Remote Sensing 72: 383-394.

Cooke S, Cowx I (2006) Contrasting recreational and commercial fishing: searching for common issues to promote unified conservation of fisheries resources and aquatic environments. Biological Conservation 128: 93–108.

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill R, Paruelo J (1997) The value of the world's ecosystem services and natural capital. Ecological Economics 25: 3–15.

Croft M, Chow-Fraser P (2007) Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of Great Lakes coastal marshes. Journal Of Great Lakes Research 33: 172-197.

Cvetkovic M (2008) Factors affecting macrophyte and fish distribution in coastal wetlands of Georgian Bay. Master's Thesis, McMaster University pgs - 117.

Cvetkovic M, Wei A, Chow-Fraser P (2010) Relative importance of macrophyte community versus water quality variables for predicting fish assemblages in coastal wetlands of the Laurentian Great Lakes. Journal of Great Lakes Research 36: 64-73.

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

Danz NP, Niemi GJ, Regal RR, Hollenhorst T, Johnson LB, Hanowski JM, Axler R, Ciborowski J, Hrabik T, Brady V, Kelly JR, Morrice JA, Brazner JC, Howe RW, Johnston CA, Host GE (2007) Integrated measures of anthropogenic stress in the U.S. Great Lakes basin. Environmental Management 39: 631-647.

DeCatanzaro R, Chow-Fraser P (2011) Effects of landscape variables and season on reference water chemistry of coastal marshes of eastern Georgian Bay. Canadian Journal of Fisheries and Aquatic Sciences. 68: 1–45.

EPA (Environmental Protection Agency) (2007) Biocriteria – Aquatic Life Use Support. Available at: [http://www.epa.gov/waterscience/biocriteria/alus/ref2.html].

FAO (Food and Agriculture Organization of the United Nations) (2010) The state of the World fisheries and aquaculture. Office of Knowledge and Exchange, Research and Extension, FAO. Rome, Italy.

Ficke A, Myrick C (2007) Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries 17: 581-613.

Findlay C, Houlahan J (1997) Anthropogenic correlates of species richness in southeastern Ontario wetlands. Conservation Biology 11: 1000–1009.

Finlayson C, van der Valk A (1995) Wetland classification and inventory: a summary. Plant Ecology 118: 185–192.

Fournier R, Grenier M, Lavoie A, Helie R (2007) Towards a strategy to implement the Canadian Wetland Inventory using satellite remote sensing. Canadian Journal of Remote Sensing 33: 1–16.

Fuller L, Morgan T, Aichele S, Geological Survey R (2006) Wetland Delineation with IKONOS High-Resolution Satellite Imagery, Fort Custer Training Center, Battle Creek, Michigan, 2005.

GBBR (Georgian Bay Biosphere Reserve) (2012) Conservation Available at: [http://www.gbbr.ca/].

Grenier M, Demers AM, Labrecque S, Benoit M, Fournier RA, Drolet B (2007) An object-based method to map wetland using RADARSAT-1 and Landsat ETM images: test case on two sites in Quebec, Canada. Canadian Journal of Remote Sensing 33: 1–18.

Herdendorf C (2004) Morphometric factors in the formation of Great Lakes coastal wetlands. Aquatic Ecosystem Health & Management 7: 179–197.

Höök T, Eagan N, Webb P (2001) Habitat and human influences on larval fish assemblages in northern Lake Huron coastal marsh bays. Wetlands 21: 281–291.

Hudon C (2004) Shift in wetland plant composition and biomass following low-level episodes in the St. Lawrence River: looking into the future. Canadian Journal of Fisheries and Aquatic Sciences 61: 603–617.

Ingram J, Holmes K, Grabas G, Watton P, Potter B, Gomer T, Snow N (2004) Development of a coastal wetland database for the Great Lakes Canadian Shoreline. Final Report to: The Great Lakes Commission. Ann Arbor, MI.

Jacobus J, Ivan LN (2005) Evaluating the Effects of Habitat Patchiness on Small Fish Assemblages in a Great Lakes Coastal Marsh. Journal Of Great Lakes Research 31: 466–481.

Jones K, Lanthier Y, van der Voet P, van Valkengoed E, Taylor D, Fernández-Prieto D (2008) Monitoring and assessment of wetlands using Earth

Observation: The GlobWetland project. Journal of Environmental Management 90: 2154–2169.

Jude D, Pappas J (1992) Fish utilization of Great Lakes coastal wetlands. Journal of Great Lakes Research 18: 651–672.

Jude D, Albert D, Uzarski DG, Brazner J (2005) Lake Michigan's coastal wetlands: distribution, biological components with emphasis on fish, and threats. In: State of Lake Michigan: Ecology, Health and Management (eds Edsall T, Munawar M), pp. 439–477. Ecovision World Monograph Series. Backhuys Publisher, Kerkwerve, The Netherlands.

Keddy PA, Fraser LH, Solomeshch AI, Junk WJ, Campbell DR, Arroyo MTK, Alho CJR (2009) Wet and Wonderful: The World's Largest Wetlands Are Conservation Priorities. BioScience 59: 39–51.

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

Killgore K, Morgan R, Rybicki N (1989) Distribution and abundance of fishes associated with submersed aquatic plants in the Potomac River. North American Journal of Fisheries Management 9: 101–111.

Lacoul P, Freedman B (2006) Environmental influences on aquatic plants in freshwater ecosystems. Environmental Reviews 14: 89–136.

Leahy M, Jollineau M, Howarth P (2005) The use of Landsat data for investigating the long-term trends in wetland change at Long Point, Ontario. Canadian Journal of Remote Sensing 31: 240-254.

Lehmann A, Lachavanne J (1997) Geographic information systems and remote sensing in aquatic botany. Aquatic Botany 58: 195–207.

Li J, Chen W (2005) A rule-based method for mapping Canada's wetlands using optical radar and DEM data. International Journal of Remote Sensing 26:5051-5069.

Lillesand TM, Kiefer RW, Chipman JW (2004) Remote Sensing and Image Interpretation. 5th Edition. John Wiley & Sons Publishing. Hoboken, New Jersey.

Madsen J, Chambers P, James W, Koch E, Westlake D (2001) The interaction between water movement, sediment dynamics and submersed macrophytes. Hydrobiologia 444: 71–84.

Magnuson J, Webster K, Assel R, Bowser C, Dillon P, Eaton J, Evans H, Fee E, Hall R, Mortsch L (1997) Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian shield region. Hydrological processes 11: 825–871.

Mayer T, Edsal T, Munawar M (2004) Factors affecting the evolution of coastal wetlands of the Laurentian Great Lakes: an overview. Aquatic Ecosystem Health and Management 7: 171-178.

Meyer J, Sale M, Mulholland P, Poff N (1999) Impacts of climate change on aquatic ecosystem functioning and health. Journal of the American Water Resources Association 35: 1373–1386.

Mortsch L, Quinn F (1996) Climate change scenarios for Great Lakes Basin ecosystem studies. Limnology and Oceanography 41: 903–911.

Mundahl N, Melnytschuk C, Spielman D, Harkins J, Funk K, Bilicki A (1998) Effectiveness of bowfin as a predator on bluegill in a vegetated lake. North American Journal Of Fisheries Management 18: 286–294.

National Wetlands Working Group (NWWG) (1997) The Canadian Wetland Classification System, 2nd Edition. Warner, B.G. and C.D.A. Rubec (eds.), Wetlands Research Centre, University of Waterloo, Waterloo, ON, Canada. 68 p.

Navulur, K. 2007. Multispectral Image Analysis Using the Object-Oriented Paradigm. CRC Press, Taylor & Francis Group. Boca Raton, FL, USA.

Newmaster SG, Harris AG, Kershaw LJ (1997) Wetland Plants of Ontario. Lone Pine Publishing. Edmonton, Alberta.

Olmanson L, Bauer M, Brezonik P (2002) Aquatic vegetation surveys using high-resolution Ikonos imagery. Pecora 15/Land Satellite Information IV/ISPRS Commission I/FIEOS Conference Proceedings pgs 10–15.

OMNR (1993) Ontario Wetland Evaluation System. Northern Manual. Ontario Ministry of Natural Resources (OMNR), No.50254.

OMNR (Ontario Ministry of Natural Resources) (2010) Great Lakes Fisheries. Available at:[http://www.mnr.gov.on.ca/en/Business/GreatLakes/2ColumnSubPage/STEL02_173913.html].

Post D, Pace M, Hairston N (2000) Ecosystem size determines food-chain length in lakes. Nature 405: 1047–1049.

Powers DA (1989) Fish as model systems. Science 246: 352.

Provincial Policy Statement. 2005. Section 3 of the Planning Act, Ministry of Municipal Affairs and Housing. Queen's Printer for Toronto, Ontario.

Quinn F (2002) Secular changes in Great Lakes water level seasonal cycles. Journal of Great Lakes Research 28: 451–465.

Randall RG, Minns CK, Cairns VW, Moore JE (1996) The relationship between an index of fish production and submerged macrophytes and other habitat features at three littoral areas in the Great Lakes. Canadian Journal of Fisheries and Aquatic Science 53: 35-44.

Rokitnicki-Wojcik D, Wei A, Chow-Fraser P (2011) Transferability of object-based rule sets for mapping coastal high marsh habitat among different regions in Georgian Bay, Canada. Wetlands Ecology and Management: 1–14.

Sawada M, Viau A, Gajewski K (2003) The biogeography of aquatic macrophytes in North America since the last glacial maximum. Journal of biogeography 30: 999–1017.

Scott WB, Crossman EJ (1998) Freshwater Fishes of Canada. Galt House Publications, Oakville, ON

Seilheimer T (2006) Development and use of the Wetland Fish Index to assess the quality of coastal wetlands in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 63: 354–366.

Seilheimer T, Chow-Fraser P (2007) Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. Journal of Great Lakes Research 33: 154–171.

Sellinger C, Stow C, Lamon E, Qian S (2008) Recent Water Level Declines in the Lake Michigan− Huron System. Environ. Sci. Technol 42: 367–373.

Smith-Cartwright LA, Chow-Fraser P (2011) Application of the index of marsh bird community integrity to coastal wetlands of Georgian Bay and Lake Ontario, Canada. Ecological Indicators 11: 1482–1486.

Smokorowski K, Pratt T (2007) Effect of a change in physical structure and cover on fish and fish habitat in freshwater ecosystems-a review and meta-analysis. Environmental Reviews 15: 15–41.

Snell, E.A., 1987. Wetland Distribution and Conservation in Southern Ontario. Inland Waters and Land Directorate, Environment Canada, Burlington, Ontario.

Trebitz A, Brazner J, Cotter A, Knuth M, Morrice J, Peterson G, Sierszen M, Thompson J, Kelly J (2007) Water quality in Great Lakes coastal wetlands: Basin-wide patterns and responses to an anthropogenic disturbance gradient. Journal Of Great Lakes Research 33: 67–85.

Trebitz A, Brazner J, Danz N, Pearson M, Peterson G, Tanner D, Taylor D, West C, Hollenhorst T (2009) Geographic, anthropogenic, and habitat influences on Great Lakes coastal wetland fish assemblages. Canadian Journal of Fisheries And Aquatic Sciences 66: 1328–1342.

Webb P (2008) The impact of changes in water level and human development on forage fish assemblages in Great Lakes coastal marshes. Journal of Great Lakes Research 34: 615-630.

Wei A, Chow-Fraser P (2007) Use of IKONOS Imagery to Map Coastal Wetlands of Georgian Bay. Fisheries 32: 167–173.

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

Wilcox DA, Meeker JE (1991) Disturbance effects on aquatic vegetation in regulated and unregulated lakes in northern Minnesota. Canadian Journal of Botany 69: 1542–1551.

Zoltai S, Vitt D (1995) Canadian wetlands: environmental gradients and classification. Plant Ecology 118: 131–137.

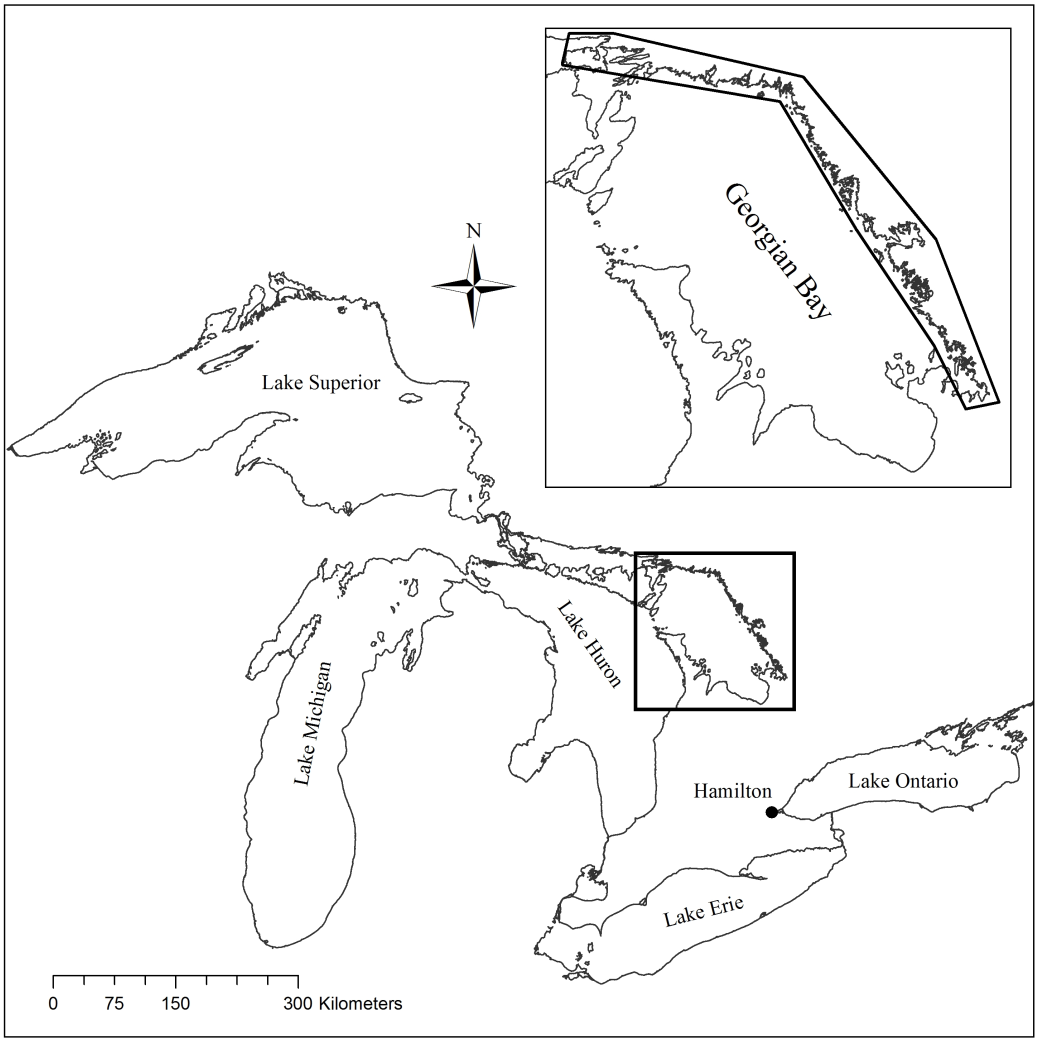


Figure i.1: The Laurentian Great Lakes with Georgian Bay highlighted in the inset. Research for this thesis was conducted along the eastern and northern shorelines of Georgian Bay (outlined in black in the inset).

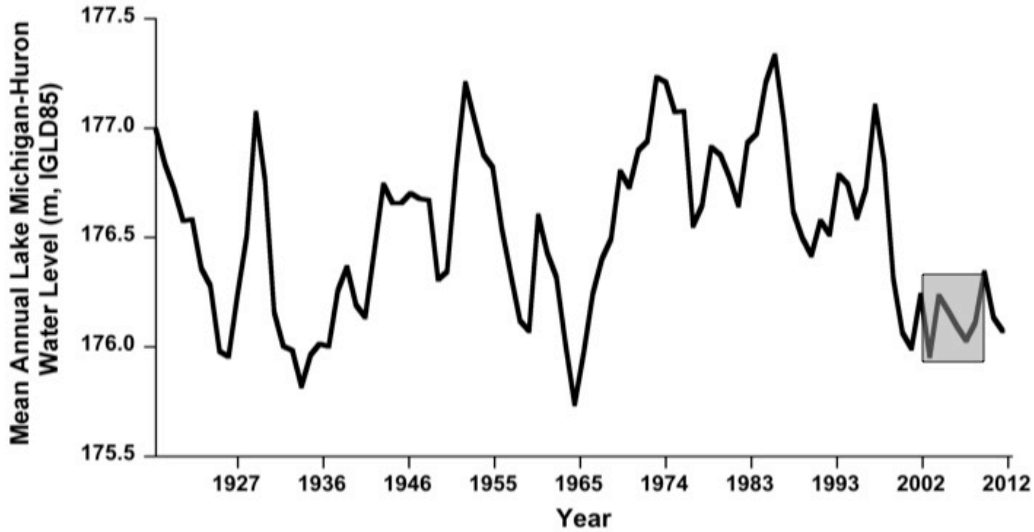


Figure i.2: Change in water levels of Lake Huron from 1918 to 2012 (Data from Canadian Hydrographic Services, Department of Fisheries and Oceans). The shaded area highlights sustained low-water levels from 2000 until 2012.

Chapter 1:

Development of an inventory of coastal wetlands for eastern Georgian Bay, Lake Huron

Reprinted with permission from:

Midwood, J.D., Rokitnicki-Wojcik, D. & Chow-Fraser, P. 2012. ISRN Ecology vol. 2012, Article ID 950173, 13 pages. doi:10.5402/2012/950173.

Abstract

Coastal wetlands of eastern Georgian Bay provide critical habitat for a variety of wildlife, especially spawning and nursery habitat for Great Lakes fishes. This unique insular landscape within a rock and water matrix has potentially the largest remaining concentration of coastal wetland habitat in the Great Lakes. Although the eastern shoreline has been designated a World Biosphere Reserve by UNESCO, a complete inventory is lacking, impeding conservation and management efforts. Prior effort by the Great Lakes Coastal Wetland Consortium (GLCWC) was unable to fully identify coastal wetland habitat in eastern Georgian Bay due to limited data coverage. Here we outline the methodology, analyses, and applications of the McMaster Coastal Wetland Inventory (MCWI) created from a comprehensive collection of satellite imagery from 2002-2008. The coastal zone was operationally defined as all habitat within 2 km upstream of the 1:100 year floodline of the lake, adopted from the Ontario Wetland Evaluation System (OWES). Wetlands were manually delineated in a GIS as two broad habitat types: coastal marsh and upstream wetland. Coastal marsh was further subdivided into low marsh (LM; permanently inundated) and high marsh (HM; seasonally inundated) habitat. Due to time constraints and the large areal extent of upstream habitat, the wetland polygon layer from the Ontario Base Map surveys was incorporated for upstream habitat north of Parry Sound. Within the coastal zone of eastern and northern Georgian Bay there are 12629 distinct wetland units comprised of 5376 ha of LM, 3298 ha of HM and 8676 ha of upstream habitat. The MCWI identifies greater total wetland area within the coastal zone than does the GLCWC inventory (17350 ha vs. 3659 ha respectively). The MCWI provides the most current and comprehensive inventory of coastal wetlands in eastern Georgian Bay.

Keywords: Georgian Bay, Inventory, Coastal Wetlands, Habitat

Introduction

Wetlands represent some of the most biologically diverse ecosystems on the planet yet globally, estimates of wetland loss due to human development range from 50% to 90 % [1]. Canada has approximately 25% of the world's wetlands [2, 3]. As a signatory in 1981 of the Ramsar convention, Canada has an obligation to identify and protect ecologically important wetlands (http://www.ramsar.org/). To date, Canada has protected more wetland area than any other country, but the prevalence of wetlands in the Canadian landscape means that there are many wetlands that have not yet been delineated. In many regions of Canada there is still an urgent need to catalogue and monitor wetlands [3].

The Laurentian Great Lakes, shared by Canada and the United States, represent the largest freshwater resource in the world. A review of wetland research conducted in this region by Herendorf [4] identified over 1500 large coastal wetlands with a total surface area of 1700 km^2. These marshes provide many important ecosystem services including water purification, nutrient sequestration, and shoreline buffering [1, 3], as well as important feeding and nursery habitat for a wide variety of organisms including fish, birds, invertebrates, mammals and reptiles [5, 6, 7]. In Ontario, majority of these wetlands have been lost or degraded as a result of human disturbance [8], except in the eastern and northern shore of Georgian Bay, where they are mostly in relatively pristine condition [9].

During the early 2000s, the Great Lakes Coastal Wetland Consortium (GLCWC) [10], consisting of both U.S. and Canadian scientists and policy makers, worked together to create a Geographic Information Systems (GIS) inventory for all five Great Lakes. The GLCWC aimed to delineate all coastal wetlands along the Great Lakes shoreline that were >2 ha in size. The Consortium included all wetlands in the Ontario Great Lakes Coastal Wetland Atlas (OGLCWA)[11], which is a GIS atlas of all wetlands that had been evaluated by the Ontario Ministry of Natural Resources using the Ontario Wetland Evaluation System (OWES)[12]; information used in this atlas dated back to 1983 and were updated with information current to 1999. The OGLCWA was complete for Lakes Superior, Erie, Ontario, as well as all connecting channels, but was incomplete for Lake Huron because it was missing some large wetlands occurring in eastern and northern Georgian Bay. Therefore, the

Consortium employed experts to manually identify wetlands using available aerial photographs for this region. For eastern Georgian Bay, aerial photographs (taken in the summers between 1984 to 2000) were available at a scale of 1:10 000 from Severn Sound to Parry Sound, and this allowed for a pixel resolution of 0.4 m. From Parry Sound to Key River, however, photos were only available at a scale of 1:20 000 (taken in the summers between 1986 to 1996), and allowed for a more coarse pixel resolution of 0.8 m. There was only limited coverage of aerial photos for northern Georgian Bay from the Key River to McGregor Bay.

Lack of information along the shore and in the surrounding islands of eastern Georgian Bay is a serious concern because this region holds some of the most pristine coastal marshes in the entire Great Lakes basin [9, 13]. This area is unique due to the low levels of agriculture and urban development that have allowed these wetlands to maintain the highest proportions of disturbance-intolerant fish and plant taxa within the Great Lakes coastal system [9, 13, 14, 15]. Although much of the shoreline was logged prior to the 1900s, easier access to inland logging sites and slow successional growth has prevented continuous logging along the shoreline in the past century. Hence, most of the wetlands have been able to persist in relatively natural condition, free of human disturbance. Furthermore, watersheds of eastern and northern Georgian Bay have thin, nutrient-poor soils on top of Precambrian Shield, which has created naturally oligotrophic coastal wetlands with very soft water mixed with more alkaline water of Georgian Bay [16]; this creates unique geochemical characteristics that supports regionally high biodiversity of aquatic plants [17]. Lastly, the complex shoreline of eastern Georgian Bay is composed of both large riverine wetlands as well as thousands of small (<2 ha) shallow rocky embayments that are protected from the strong wind and wave action that characterize the region. Therefore, Georgian Bay has an assembly of coastal wetlands that are unique in the Great Lakes basin in terms of geochemistry, biodiversity, areal cover and abundance and can be considered reference conditions for the Great Lakes [18].

Besides the ubiquitous potential for indiscriminate human development [16], coastal wetlands of Georgian Bay have been strongly influenced by the sustained low water levels that have prevailed over the last decade (Figure 1). This trend is expected to continue, with climate change scenarios predicting a further decline in water level of Lakes Huron-Michigan by >1m during the next 25 years [19], accompanied by reduced interannual variation [20]. Coastal wetlands are dynamic systems, where diversity of habitat and biota are maintained by a natural disturbance in the form of fluctuating water levels; in years of high water, terrestrial vegetation dies back, and in years of low water levels, aquatic vegetation dies back [21]. Without interannual water-level variation, either the aquatic or the terrestrial vegetation would dominate at the expense of the other. The current episode of sustained low water levels would favour terrestrial vegetation at the expense of aquatic vegetation [22], which is an important component that provides critical spawning and nursery habitat for the Lake Huron fish community [23, 24].

Although water levels encountered presently are not the lowest in recorded history for Lakes Huron-Michigan, the sustained low water levels that began in 2001 and that have persisted through to 2011 have not occurred in the last 100 years, and it is difficult to predict how the wetland community will adapt to these extremes, and how the fish community, in particular, will be able to adjust to losses and gains in aquatic habitat. Hence, there is an urgent need to conduct research to determine how this trend towards lower water levels will affect the quantity and quality of wetland habitat so that they can be monitored and protected from further human activities. To aid environmental agencies and municipal planners and to enable valid extrapolation, it is important that the research be conducted on a set of randomly chosen wetlands. To date, however, there is no such inventory, because the one created by the GLCWC is incomplete for this region. There are other limitations of the GLCWC inventory (herein GLCWCI) that make it unsuitable for research on fish habitat. First, since wetlands have been delineated from photos taken in different years (1983 to 1999) and at different water levels, size of habitat zones in wetlands within the inventory cannot be directly compared because they are not standardized to one water level. Secondly, the GLCWCI excludes most of the wetlands that occur in the rocky coastal region and island archipelagos of northeastern Georgian Bay, where there could be extensive fish habitat. The last and perhaps the greatest limitation for fish ecologists is that the inventory does not distinguish between terrestrial and aquatic habitat types and it is impossible to determine the distribution of fish habitat across the region.

In this paper, we show how a comprehensive coastal wetland inventory can be created for eastern and northern Georgian Bay that is both cost-effective and suitable for use in studies of fish habitat at the scale of the entire Georgian Bay (over 4500 km of shoreline). We propose to use high-resolution IKONOS satellite imagery acquired during a 5-year period with similar water levels to ensure that wetland habitat can be directly comparable across the region, and show the inconsistencies that can result when imagery under different water-level scenarios are used to delineate wetlands boundaries. We will also apply a simple rule to delineate coastal wetland habitat into low marsh zone (fish habitat) and high marsh zone (meadow habitat). The approach we develop here can be used in coastal projects of other large lakes where there is a need to monitor changes in fish habitat at a scale of an entire lake basin.

Methods

Satellite imagery

IKONOS satellite images covering all of eastern Georgian Bay and parts of the North Channel were acquired by Georgian Bay Forever (GBF;

http://www.georgianbayforever.org/) and licensed to McMaster University. Images covering the regions between Severn Sound and Parry Sound were acquired in July 2002, images covering up to Key River were acquired in July 2003, images covering the McGregor Bay and Bay of Islands regions were collected in July 2005, an image covering Matchedash Bay was collected in September 2005, and an image covering Beaverstone Bay was acquired in August 2008 (Figure 2). The IKONOS satellite images used for this inventory have a pan-sharpened resolution of 1 m and provide spectral information in the red, green, blue and near-infrared wavelengths. IKONOS satellite images were not available for two regions of northern Georgian Bay, but through the Ontario Provincial QuickBird Project (2007), we were able to obtain QuickBird satellite coverage for all of the remaining gaps except for only a 10 km stretch between the French River and Beaverstone Bay (Figure 2). Like IKONOS imagery, QuickBird images provide high-resolution (60 cm) multispectral data (visible and near-infrared spectrum). In total, 8 QuickBird images were used, all of which were acquired in September 2006. In a comparison of IKONOS and QuickBird images for the purpose mapping mangroves, Wang et al. [25] found little difference in their ability to map this habitat type. We therefore concluded the use of different image types would have a negligible impact on the quality of the final inventory.

Manual Delineation

The McMaster Coastal Wetland Inventory (MCWI) was created by manually delineating wetlands from satellite images in a GIS. The IKONOS images were initially stacked for easier use. The three visible bands (red, green, blue) of the IKONOS images were used to create a true colour image. A second image was then created through the substitution of the near-infrared (NIR) band in place of the red band (i.e. NIR, green, blue). QuickBird images were already combined into both a true colour and a near-infrared form. The NIR wavelength is a good indicator of vegetation, especially in aquatic systems [26] and by switching between the true colour and the near-infrared images, technicians are better able to discriminate wetland vegetation from surrounding land cover. Stacked images were then imported into ArcMap 9.2 (ESRI Inc., Redlands, California, U.S.A., 2006) in the working projection Universal Transverse Mercator (UTM), zone 17. For each wetland, the technicians traced the boundary of wetland habitats (i.e. Low Marsh, High Marsh, Upstream Wetland) following specific rules regarding the upper and lower limits of each category (outlined below). The result was a single polygon for each applicable wetland category for each wetland.

Rules for delineations/Accuracy Assessment

The Ontario Ministry of Natural Resources (OMNR) in its Ontario Wetland Evaluation Systems (OWES)[12] defines coastal wetlands as wetlands that are influenced by large water bodies and generally found within 2-km of the high water mark. The coastal zone is therefore operationally defined as land within 2-km of the shoreline and within this zone, only wetlands that are hydrologically connected via surface water to Georgian Bay are considered coastal wetlands. This 2-km coastal zone equates to roughly 177000 ha along the shores of eastern and northern Georgian Bay. Since the major focus of this inventory is to quantify fish habitat, surface hydrologic connectivity is an essential criterion. Despite the existence of many hydrologically isolated wetlands upstream of the shoreline, disconnected wetlands were not included. These wetlands occur above the high-water mark and therefore they do not serve as current or potential Great Lakes coastal fisheries habitat. They also can be difficult to identify visually because they exist along a continuum of succession from open water to areas that are fully forested and their delineation would have greatly prolonged the time to completion of the MCWI inventory. They were thus omitted from this inventory.

Each wetland found to be within 2-km and hydrologically connected to Georgian Bay was delineated into three habitat categories (i.e. High Marsh, Low Marsh, Upstream Wetland) and the area of each polygon was calculated in

ArcMap 9.2 (ESRI Inc., Redlands, California, U.S.A., 2006). Rules for delineating boundaries of each habitat category are as follows:

- The **High Marsh** (**HM**) category represented wetland habitat that were inundated on a seasonal basis; this area is often referred to as 'wet meadow' habitat. Wet meadows provide important habitat for a variety of species including birds, reptiles and amphibians [27]. The lower limit of the HM habitat was defined by the shoreline, and was the upper limit of LM. The upper limit of the HM was the forest boundary and/or when there was change from HM to upstream habitat (swamp, bog or fen).
- The **Low Marsh** (**LM**) category represented portions of the wetland that were permanently inundated and essential areas for fish spawning and foraging. Upper limits of LM habitat were defined by the water's edge and exclude meadow vegetation. The lower limit of the wetland includes submerged aquatic vegetation (SAV), which is a critical component of fish habitat. Unfortunately, the dystrophic conditions of the water in most regions of eastern Georgian Bay did not allow us to map SAV using satellite images. Therefore, the lower limit was approximated by calculating a distance that is 2.5 times the width of the emergent and/or floating vegetation zone (visible in the image) and applying this from the water's edge along the longest axis of the wetland. This distance was reduced if the lower limit extended beyond the opening of an embayment.

This lake-ward boundary is considered a conservative estimate of the maximum depth of colonization by SAV, based on dozens of underwater surveys in wetlands in this region (Midwood pers. obs.).

- **Upstream Wetland** (**UP**) habitat corresponded to all remaining wetlands that were hydrologically connected to the bay via surface water and that occur within a 2 km buffer of the shoreline. In vast majority of cases, beaver activity created conditions that separated the upper limit of the HM from the lower limit of the UP Wetland habitat. Since these beaver ponds can be seasonally connected with downstream habitat, they can act as potential fish habitat for fish communities in the affected coastal wetlands and were therefore included in the inventory. Due to the large number of UP along the eastern and northern shore of Georgian Bay, there was not enough time to delineate UP habitat in all 81 images. UP was delineated for 21 IKONOS images covering the region from Parry Sound south to Severn Sound (Figure 3). These delineations were compared to existing wetland delineations in the OMNR's [28] Ontario Base Map (OBM) that corresponds to our UP habitat. We found the OBM wetland delineations provided comparable coverage (data not shown) and therefore the OBM wetland layer was directly incorporated into the MCWI for the remaining areas (60 images). We should note, however, that some of the wetlands incorporated into the MCWI from the OBM survey may not be directly connected by surface water and the extent of this error has not yet been determined.

To standardize variations among technicians working on this project, all technicians were first trained on the same five images. Only when they achieved an acceptable level of precision (greater than 85% similarity) were they allowed to contribute to the project. To further reduce technician-bias associated with discerning the lower extent of LM, only one technician was assigned to digitize this habitat category for majority of the satellite images. There was less subjectivity associated with delineations of HM and UP, and hence, more than one technician was assigned to these habitats. Once all polygons were digitized, a single technician went through the entire data set to ensure that edges between habitat categories did not overlap, and that no wetlands had been missed.

We assigned wetlands to the quaternary watersheds (acquired as OBM from [28]; Figure 4) that surrounded them using a spatial join tool in ArcMap 9.2 (ESRI Inc., Redlands, California, U.S.A., 2006). When a wetland occurred on the boundaries of two or more watersheds, it was assigned to the watershed that held the majority of the wetland. If it was unclear to which particular watershed a wetland should be assigned, the wetland was assigned to both. This is the reason why there is a slight discrepancy (occurred in <5% of the wetlands) between total area when all wetlands are summed without regard to watershed origin, and when they are pooled after they have been sorted by quaternary watershed.

Comparison of differences in inventories

The shapefile for the GLCWCI was available online from the Great Lakes Commission's website [29], and was used to conduct a comparison of the MCWI and GLCWCI. The wetland layers created in the MCWI were used to clip out portions of the GLCWCI corresponding to the Coastal and Upstream regions. Our "Coastal" zone is the same as the "Lacustrine" class defined by the GLCWCI and can be used interchangeably. We note, however, that there is no category that matches our "Upstream" portion, but in the GLCWCI shapefile, there are regions that would be defined as "Upstream", even though they were not actually classified as such. To assess differences between inventories, we used a GIS to calculate total area of wetlands for the respective inventories. The comparison included wetlands in the MCWI and the GLCWCI along the northern and eastern shore of Georgian Bay. We excluded all wetlands <2 ha in the MCWI to make this criterion consistent with that of the GLCWCI.

Results

McMaster Coastal Wetland Inventory

In total, 73 IKONOS and 8 QuickBird images were digitized and each image was delineated separately for both the High Marsh and Low Marsh habitats. For Upstream Wetland habitat, however, only 21 IKONOS images were delineated (south of Parry Sound) with the remaining data being filled in from the OBM wetland layer (Figure 3).

Despite our best efforts, we were unable to acquire appropriate imagery to fill one small gap in northern Georgian Bay (Figure 2). We know that wetlands exist in these gaps because we have conducted field sampling there (Chow-Fraser, unpub. data). Therefore, the estimate in this document should be considered a slight underestimate of the actual amount of coastal wetland habitat in northern Georgian Bay, and future efforts should be made to fill this gap with some other satellite media of the same vintage. Relative to the remainder of the shoreline in eastern and northern Georgian Bay, this gap in imagery amounts to only a small fraction of the shoreline and should be relatively easy to update as soon as appropriate imagery has been acquired.

In total, 3771 units (414 units > 2 ha) of Low Marsh, 6355 units (289 units > 2 ha) of High Marsh and 2603 units (883 units > 2 ha) of Upland Wetland are included in the MCWI. Size of wetlands in LM habitat varied a great deal, with a mean of 1.4 (± 12.0) ha and a median of 0.3 ha. By comparison, those in HM habitat were more uniform in size, with a mean of 0.5 (± 2.2) ha and median of 0.1 ha, and those in the UP habitat were larger, with a mean of 3.3 (± 9.0) ha and a median of 1.1 ha. UP habitat covered the largest area (8676 ha), followed by LM (5376 ha) and HM (3298 ha) (Table 2; Figure 4).

Along the eastern and northern shores of Georgian Bay, there are a total of 37 quaternary watersheds ranging in size from 564 ha (Giants Tomb) to 126103 ha (French River) (Figure 5; Table 1). The largest amount of wetland habitat (2394 ha) was found in the Moon-Musquash watershed (Table 1). When sorted by different type of habitat, however, we found that the Coldwater watershed was associated with the greatest amount of LM habitat (49 units with a total area of 797 ha) (Table 3). It was surprising that this LM habitat only accounted for 3.7% of the total Coldwater watershed area, when the LM habitat in Beausoleil-Severn Island accounted for 25.6 % of the total watershed area; Islands in Beausoleil-Severn were also associated with the highest percentage of HM habitat (8.7%). The Eastern Coast Islands watershed contained 1035 units of HM, for a total area of 404 ha (Table 3). The Moon-Musquash River watershed contained the greatest amount of UP habitat, with 159 units and a total area of 1708 ha (Table 3), while the MacGregor-Sampson Islands watershed had the highest percentage of UP habitat (8.6%) of all 37 quaternary watersheds.

MCWI/GLCWCI Comparison

We compared the total amount of wetlands in both the MCWI and the GLCWCI with respect to wetlands > 2ha (Table 2). There were 1586 wetland units in the MCWI, covering an area of 13267 ha; by comparison, the GLCWCI only included 696 wetland units, covering a total area of 3660. Within the Coastal zone, the MCWI included more than twice as many LM (414 vs 170) and UP (883 vs 379) units, and a greater number of HM units (289 vs 234). In terms of area, however, there was almost 14 times the area of LM (4044 vs 298 ha), greater than three times the amount of HM (1842 vs 587), and more than six times the amount of UP (7381 vs 1762 ha) habitat. The greater number of wetland units and area included in the MCWI is despite the inclusion of 1014 ha of wetland area that was unique to the GLCWCI (see Figure 6).

Discussion

The McMaster Coastal Wetland Inventory is currently the most comprehensive inventory of coastal wetlands for eastern and northern Georgian Bay. With the completion of this inventory we can now update the total coastal wetland habitat area for the Canadian side of the Great Lakes. With the inclusion of the non-overlapping areas of the MCWI, the Great Lakes contain 78405 ha of coastal wetland habitat in Canada, increasing the total for Lake Huron by 47.6% to 30882 ha. We have identified all coastal marshes in a region where complete data has been lacking and have filled an important void in the distribution of coastal wetland habitat in the Great Lakes basin. This is an important advancement in the tools available for wetland managers where basin-wide decision-making is essential for the future persistence of these habitats. By accessing a large collection of satellite imagery, we have been able to fill in major gaps in the Great Lakes Coastal Wetland Consortium Inventory along eastern and northern Georgian Bay that had been noted by the authors [10]. Coastal wetlands are dynamic systems that vary in wetland size and dominant vegetation type when water levels fluctuate seasonally and annually [21, 30, 31]. In order to create an inventory that provides consistent wetland coverage, it is essential that all delineations utilize imagery acquired during similar water-level conditions. If there is any temporal discrepancy in image acquisition, wetlands in images that were acquired during low water levels may have more HM habitat [31] than the same wetlands delineated using imagery acquired during higher water levels. All wetlands in the MCWI were digitized from high-resolution IKONOS and Quickbird satellite imagery acquired during a period of low water levels between 2002 and 2008 (mean of 176.12 m ± 0.13.). This means that all wetland areas in our inventory are standardized and are directly comparable. Conversely, wetlands in the GLCWCI were digitized from aerial photos or satellite images acquired at different years (from 1983 to 1999; mean of 176.78 ± 0.29), and were not standardized to a consistent water level. Although wetlands digitized in the

GLCWCI may provide information on wetland location and some UP boundaries, regional comparisons of LM and HM wetland area may not be possible due to the potential influence of interannual water level variation and resulting vegetation changes.

The MCWI and the GLCWCI also differ with respect to the level of detail provided by each inventory for eastern and northern Georgian Bay. The GLCWCI only identified 21% of wetlands (by area) available in the MCWI. The MCWI raises some important issues concerning the minimum mapping unit of inventories. The GLCWCI used a minimum mapping unit of 2ha to identify coastal wetland habitat. In our objectives for the MCWI we decided to identify all coastal marsh habitat possible with the resolution of our imagery. This proved to identify a unique characteristic of this region in that a majority of the coastal wetlands are <2ha. A minimum mapping unit of <2ha would exclude a large portion of the data in this project which was meant to identify critical fish habitat. Accordingly, we suggest that the GLCWCI should not be used to estimate the amount of coastal fish habitat in Georgian Bay because the amount of LM habitat in the MCWI was 13 times higher for wetlands > 2ha and 18 times higher for all wetlands regardless of size. As outlined previously, the major differences between inventories reflect how they were created. Authors of the GLCWCI indicated that they relied heavily on OWES-identified wetlands (minimum size of 2 ha) and availability of aerial photography to fill the considerable gaps in eastern and northern Georgian Bay. Unfortunately, there were more data available for upstream wetlands than for those in the coastal zone because the initial acquisitions of aerial photographs were for forest survey purposes. As a result, the authors also recognized that there were major gaps in coverage for the GLCWCI, and that the missed areas likely contained a considerable number of coastal wetlands [10]. In this respect, the MCWI should have the most complete coverage given that the satellite images we acquired provided a seamless coverage of the entire 2-km coastline of eastern and most of northern Georgian Bay.

Herdendorf [4] identified approximately 1500 wetlands in the Great Lakes (total area of 1730 km^2) that were sufficiently large to have local ecological importance. Among these, only one, Matchedash Bay, was found in eastern Georgian Bay. We feel that this is a severe underrepresentation of large wetlands of ecological importance in Georgian Bay, and that Herdendorf's list should be updated with information from this study. We speculate that size alone is not a sufficient criterion for determining ecological significance. The Ontario Wetland Evaluation System [12] indicates that small coastal wetlands (i.e. <2 ha) can be grouped together to form complexes if there is a biological or hydrological rationale for doing so. For Georgian Bay, many of the smaller wetlands could be grouped into complexes since they are often found close together (within 750 m). With completion of the MCWI, we are now in the position to create the complexes, once we have a better understanding of the role that small wetlands play in terms of ecosystem functions, such as providing suitable nursery and spawning habitat for the Lake Huron fishery.

Water levels in the Great Lakes, specifically in Lake Michigan-Huron, are expected to decline as a result of climate change [19, 20]. These changes will alter the distribution, areal coverage and vegetation structure in the coastal wetlands of eastern Georgian Bay. Wetland habitat in the MCWI has been classified according to three unique habitat zones: Low Marsh, which is critical habitat for fish [6], High Marsh which is critical habitat for marsh birds and turtles [7, 32], and Upstream habitat, that plays a critical role in controlling water quality in downstream habitats (UP)[33]. Both Wei and Chow-Fraser [34] and Midwood and Chow-Fraser [35] utilized IKONOS satellite imagery to map different types of fish habitat in the LM portion of coastal wetlands. In the HM zone, Rokitnicki-Wojcik and Chow-Fraser [36] developed a method that can provide detailed maps for HM vegetation. These methods can now be applied to map all LM and HM habitat in the MCWI that was delineated with IKONOS imagery. This mapping should produce consistent, baseline maps of fish habitat as well as meadow habitat for majority of coastal wetlands in eastern and northern Georgian Bay. With the acquisition of new satellite imagery, changes in vegetation coverage can be monitored and linked to the observed changes in water level in Lake Michigan-Huron.

DeCatanzaro et al. [16] were able to use road density as a surrogate for human development. They found poorer water quality (i.e. increased nutrients, conductivity and suspended solids) in coastal wetlands adjacent to quaternary watersheds that were associated with high road density. Majority of coastal wetlands in Georgian Bay occur within watersheds that have low road density except for two (Sturgeon River and Coldwater River) [16]. The highest density occurs in the Coldwater River watershed (16.1 m/ha), and this watershed also contains the largest single area of LM habitat, and this may mean that one of the largest fish spawning areas in Georgian Bay is currently threatened by human development. Eleven of the 32 watersheds that are not currently being impacted by human development have large chains of islands. These island watersheds represent ideal conservation sites since they have limited human access, except for some cottage development. In addition, despite the fact that these islands cover less than 5% of the total watershed area within the basin, they account for nearly a quarter of all coastal habitats in eastern and northern Georgian Bay. A first step towards conserving critical wetland habitat in Georgian Bay should be to protect these islands and currently the Georgian Bay Land Trust (GBLT) has managed to acquire and protect islands covering over 250 ha (www.GBLT.org).

In creating the MCWI, we have provided a consistent and accurate inventory of coastal wetlands in eastern and northern Georgian Bay under low water level conditions. This project took three years to complete with help of many GIS technicians. At all times we tried to ensure that each technician was delineating at a consistent level of accuracy. We do acknowledge that small differences in the date of image acquisition, which created slight differences in ground feature colour, may result in some discrepancy in wetland delineation. This type of error in image interpretation is unavoidable and we believe the resulting error does not significantly alter the accuracy of the MCWI. The incorporation of the OBM wetland layer into the MCWI was necessary in order to complete the project in a timely fashion. While we found that this layer provided a sufficient level of coverage for the UP portion of our inventory, manual delineation designed specifically to identify upland habitat which was connected via surface water may provide a more accurate inventory.

While we believe that the MCWI in its current form provides a useful and comprehensive tool that should be adopted and utilized by conservation managers, we know that it can be improved with further enrichment. First, the image gap in a small portion of the French River Delta needs to be filled and all wetlands in this area need to be delineated. Secondly, wetlands identified by the inventory need to be grouped into ecologically relevant complexes in accordance with the complexing rules outlined in the OWES [12] or with suitable modifications. Finally, we recommend that satellite imagery be acquired every five years for a statistically valid subset of the MCWI. This will allow researchers and managers to track general trends in areal wetland coverage change as water levels fluctuate.

Here we show how to create a complex habitat-based inventory of coastal wetlands for a large expanse of eastern and northern Georgian Bay, Ontario, Canada. The application of this project beyond the scope of the Laurentian Great Lakes is widespread as managers are continually in need of cost-effective methods to produce high quality and ecologically relevant geospatial data. Although aerial photography is considered the gold standard for habitat identification, in the context of the Great Lakes, the MCWI is able to provide a static view of habitat conditions across a significant portion of the entire basin, which has proven to be too costly in past projects (GLCWCI). We recommend that managers undertaking mapping projects at similar spatial scales as the MCWI and GLCWCI consider the benefits of having contiguous data coverage within a time scale where geographic comparisons are valid and regional differences due to succession are minimized.

Conclusions

The use of IKONOS and Quickbird satellite imagery in the MCWI ensured that the entire shoreline of eastern Georgian Bay and large parts of the northern shoreline were available for delineation. The MCWI clearly provides the most detailed delineation of wetlands in the coastal zone in eastern and northern Georgian Bay. We therefore recommend that the MCWI be integrated into the GLCWCI to provide a more complete inventory of coastal wetlands for the

Georgian Bay region. With the creation of this comprehensive inventory, researchers will be able to conduct statistically valid research using a randomly selected wetland dataset and have accurate data of the distribution of coastal wetland habitat basin-wide. This inventory will also be useful to environmental managers and land-use planners to ensure that future development takes coastal wetland habitat into account.

Acknowledgements

We thank Georgian Bay Forever for providing a research grant to PC-F to undertake this three-year project, and for giving us access to the IKONOS imagery. Many GIS technicians contributed to the final inventory, but Kristina Cimaroli undoubtedly made the most important contributions. We acknowledge additional funding in the form of an NSERC scholarship to DR-W and an OGS scholarship to JM.

References

[1] Keddy, P. A., Fraser, L.H., Solomeshch, Junk, W.J., Campbell, D.R., Arroyo, M.T.K. and Alho, C.J.R., 2009. Wet and Wonderful: the world's largest wetlands are conservation priorities. Bioscience **59**(1): 39-51.

[2] Li, J. and Chen, W., 2005. A rule-based method for mapping Canada's wetlands using optical radar and DEM data. International Journal of Remote Sensing **26**(22): 5051-5069.

[3] Jones, K., Lanthier, Y., van der Yoet, P, van Valkengoed, E., Taylor, D. and Fernandex-Prieto, D., 2008. Monitoring and assessment of wetlands using Earth Observation: the GlobWetland project. Journal of Environmental Management, doi:10.1016/j.jenvman.2007.07.037.

[4] Herendorf, C.E., 2004. Morphometric factors in the formation of Great Lakes coastal wetlands. Aquatic Ecosystem Health and Management. **7**(2):179-197

[5] Chow-Fraser, P. and Albert, D. 1999. Identification of Eco-Reaches of Great Lakes Coastal Wetlands that have high biodiversity values. Discussion paper for SOLEC '98. Env Canada-USEPA Publication, 88 pp.+ appendices

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

[7] Smith, L.A. and Chow-Fraser, P. 2010. Implications of the species-area relationship on sampling effort and conservation priorities of wetland birds in southern Ontario. Wetlands **30**: 553-563.

[8] Snell, E.A. 1987. Wetland Distribution and Conversion in Southern Ontario. Canada Land Use Monitoring Program. Working Paper No. 48. Inland Waters and Lands Directorate, Environment Canada.

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

[10] Ingram, J., Holmes, K., Grabas, G., Watton, P., Potter, B., Gomer, T. and Stow, N., 2004. Development of a coastal wetland database for the Great Lakes Canadian Shoreline Final Report to: The Great Lakes Commission, 1-51.

[11] Environment Canada and Ontario Ministry of Natural Resources. 2003. The Ontario Great Lakes Coastal Wetland Atlas: A summary of information (1983-1997). 49 pp.

[12] OMNR 1993. Ontario Wetland Evaluation System. Northern Manual. Ontario Ministry of Natural Resources (OMNR), No.50254.

[13] Chow-Fraser, P. 2006. Development of the wetland water quality index for assessing the quality of Great Lakes coastal wetlands. *In* Coastal wetlands of the Laurentian Great lakes: health, habitat, and indicators.

Edited by T.P. Simon and P.M. Stewart. Bloomington, Indiana: Indiana Biological Survey. pp. 137-166.

[14] Croft, M. V. and Chow-Fraser, P. 2007. Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of Great Lakes coastal marshes. Journal of Great Lakes Research **33** (Special Issue 3): 172-197.

[15] Seilheimer, T. S. and Chow-Fraser, P. 2007. Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. Journal of Great Lakes Research **33**(Special Issue 3): 154-172.

[16] DeCatanzaro, R., Cvetkovic, M. and Chow-Fraser, P. 2009. The relative importance of road density and physical watershed features in determining coastal marsh water quality in Georgian Bay. Environmental Management **44**: 456-467.

[17] Croft, M. V. and Chow-Fraser, P. 2009. Non-random sampling and its role in habitat conservation: a comparison of three wetland macrophyte sampling protocols. Biodiversity Conservation **8**(9): 2283-2305.

[18] EPA (Environmental Protection Agency). 2007. Biocriteria – Aquatic Life Use Support. Available at: [http://www.epa.gov/waterscience/biocriteria/alus/ref2.html].

[19] Mortsch, L. D. and Quinn, F.H. 1996. Climate change scenarios for Great

Lakes Basin ecosystem studies. Limnology and Oceanography **41**: 903-911.

[20] Sellinger, C. E., Stow, C.A., Lamon, E.C. and Qian, S.S. 2008. Recent water level declines in the Lake Michigan-Huron system. Environmental Science and Technology **42**: 367-373.

[21] Keddy, P. A. and Reznicek, A.A., 1986. Great lakes vegetation dynamics: the role of fluctutating water levels and buried seeds. Journal of Great Lakes Research **12**: 25-36.

[22] Quinlan, C., and Mulamoottil, G. 1987. The effects of water level fluctuations on three Lake Ontario shoreline marshes. Canadian Water Resource Journal **12**: 64-77.

[23] Jude, D.J. and Pappas, J., 1992. Fish utilization of Great Lakes coastal wetlands. J. Great Lakes Res. **18**(4):651-672.

[24] Randall, R. G., Minns, C.K., Cairns, V.W., and Moore, J.E. 1996. The relationship between an index of fish production and submerged macrophyte and other habitat features at three littoral areas in the Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences **53**(Suppl 1): 35-44.

[25] Wang, L., Sousa, W.P., Gong, P. and Biging, G.S. 2004. Comparison of IKONOS and QuickBird images for mapping mangrove species on the Caribbean coast of Panama. Remote Sensing of Environment **91**: 432-440.

[26] Hardisky, M. A., Gross, M.F. and Klemas, V. 1986. Remote sensing of coastal wetlands. Bioscience **36**(7): 453-460.

[27] Maynard, L. and Wilcox, D., 1997. Coastal wetlands of the Great Lakes. State of the Lakes Ecosystem Conference 1996. U.S. Environmental Protection Agency. 99 pp.

[28] OMNR 2000. Ontario Base Map (Wetland polygons, eastern and northern Georgian Bay coast)[computer file]. Ontario Ministry of Natural Resources. http://www.geographynetwork.ca/website/obm/viewer.htm. Accessed April 1st, 2009.

[29] GLCWC 2004 Coastal Wetland inventory (Polygon data). [electronic file] Great Lakes Coastal Wetland Consortium http://www.glc.org/wetlands/inventory.html Accessed June 1st 2007

[30] Wilcox, D. A. and Meeker, J.E. 1991. Disturbance effects on aquatic vegetation in regulated lakes in northern Minnesota. Canadian Journal of Botany **69**(7): 1542-1551.

[31] Hudon, C., 2004. Shift in wetland plant composition and biomass following low-level episodes in the St. Lawrence River: looking into the future. Can. J. Fish. Aquat. Sci. **61**:603-617.

[32] DeCatanzaro, R. and Chow-Fraser, P. 2010. Relationship of road density and marsh condition to turtle assemblage characteristics in the Laurentian Great Lakes. J. Great Lakes Research. **36**: 357-365.

[33] DeCatanzaro, R., 2010. Factors influencing coastal marsh water quality in Georgian Bay, Ontario. Master's Thesis. McMaster University, Hamilton, Ontario, Canada.

[34] Wei, A. and Chow-Fraser, P. (2007) Use of IKONOS imagery to map coastal wetlands of Georgian Bay. Fisherie*s* **32**: 167-173.

[35] Midwood, J.D. and Chow-Fraser, P. (2010) Mapping floating and emergent aquatic vegetation in coastal wetlands of eastern Georgian Bay, Lake Huron, Canada. Wetlands **30**: 1-12.

[36] Rokitnicki-Wojcik, D., Wei, A. and Chow-Fraser, P. 2011 Transferability of object-based rule sets for mapping coastal high marsh habitat among different regions in Georgian **Bay,** Canada. Wetlands Ecol Manage*.* **19**: 223-236*.*

Table 1. List of OMNR watershed codes and the assigned name to reflect the major tributary or geographic feature. Watershed area, the total number of marshes and the amount of marsh area within each watershed are provided.

Quaternary Watershed ID	Watershed Name	Watershed Area (ha)	Wetland Number	Total Marsh Habitat (ha)	Number of marshes >2ha
2CE-01	La Cloche	27269	82	132.4	15
2CE-11	La Cloche Islands	2542	34	12.6	0
2CF-02	Philip Edward Island	4909	367	433.2	56
2CF-18	MacGregor-Sampson Islands	2179	266	350.7	32
2CG-06	Great La Cloche Island	9643	136	394.2	35
2CG-32	Bedford-Rous Islands	1937	29	7.9	0
2CG-33	Strawberry-Heywood Islands	1595	12	72.8	8
2CH-01	Beaverstone River	12957	196	500.1	39
2CH-03	Killarney	62878	594	915.5	87
2CH-04	Whitefish River	26640	1162	841.1	78
2CH-17	Killarney Islands	2640	524	233.7	15
2DD-01	French River	126103	1225	792.8	91
2DD-03	Pickerel River	105176	660	385.0	41
2DD-26	Outer French-Pickerel Islands	3484	749	257.7	16
2EA-01	Henvey Inlet-Key River	19669	392	751.3	90
2EA-03	Sandy Bay	6057	118	280.2	23
2EA-04	Giroux River	10949	225	285.9	37
2EA-05	Point au Baril	11554	566	939.6	116
2EA-06	Shebeshekong River	19720	275	880.7	83
2EA-07	Parry Island	7666	249	586.4	40
2EA-08	Spider Lake	8816	280	297.9	36
2EA-09	Still River	23649	14	38.2	3
2EA-10	Magnetawan-Naiscoot Rivers	92622	235	448.5	49

2EA-11	Naiscoot Lake	21331	9	5.5	0
2EA-13	Shawanaga River	30979	201	329.5	48
2EA-14	Parry Sound	59479	241	434.2	55
2EA-15	Seguin River	8330	27	41.2	7
2EA-24	Eastern Coast Islands	11854	1835	1386.8	102
2EB-01	Moon-Musquash Islands	4392	1088	572.0	39
2EB-02	Moon-Musquash Rivers	71731	916	2394.0	179
2EB-03	Muskoka Lakes	18578	6	7.4	1
2EC-17	Severn River	70445	423	973.7	77
2EC-18	Beausoleil-Severn Islands	2913	798	746.4	52
2ED-04	Sturgeon River	18887	19	135.2	7
2ED-05	Coldwater River	21727	149	1131.0	29
2ED-06	North River	32214	31	97.3	4
2ED-17	Giants Tomb Island*	564	0	0.0	0

* Currently no satellite image coverage for the watershed in the MCWI

Table 2. Comparison of the total area of Low Marsh, High Marsh and Upstream wetlands for eastern and northern Georgian Bay identified in the GLCWC and the MCWI.

	GLCWC Area (ha)	**GLCWC Polygon #**	**GLCWC Mean Size (ha)**	**MCWI Area (ha)**	**MCWI Polygon #**	**MCWI Mean Size (ha)**
Total Low Marsh	297.8	170	1.8	5376.1	3771	1.4
Total Low Marsh >2ha	297.8	170	1.8	4043.9	414	9.8
Total High Marsh	586.7	234	2.5	3297.5	6255	0.5
Total High Marsh >2ha	586.7	234	2.5	1842.1	289	6.4
Total Upstream	1762.4	379	4.7	8676.1	2603	3.3
Total Upstream >2ha	1762.4	379	4.7	7381.2	883	8.4
Total Wetland	3660.8	696	5.3	17349.7	12629	1.4
Total Wetland >2ha	3660.8	696	5.3	13267.1	1586	8.4

Table 3. Number and areal coverage of Low Marsh, High Marsh and Upstream wetlands for each of the 37 quaternary watersheds along eastern and northern Georgian Bay (see location of each in Figure 5).

Watershed Name	Number of Low Marsh units	Area of Low Marsh (ha)	Number of High Marsh Units	Area of High Marsh (ha)	Number of Upstream Wetlands	Area of Upstream Wetlands (ha)
Beausoleil-Severn Islands	303	484.7	492	254.2	3	7.5
Beaverstone River	44	139.6	103	116.7	50	247.4
Bedford-Rous Islands	14	5.6	15	2.4	0	0.0
Coldwater River	49	796.7	100	334.4	0	0.0
Eastern Coast Islands	629	631.5	1035	403.5	169	353.1
French River*	413	164.4	554	175.1	260	462.7
Giants Tomb*	0	0.0	0	0.0	0	0.0
Giroux River	66	77.4	92	97.9	70	121.2
Great La Cloche Island	14	25.5	16	2.9	106	365.8
Henvey Inlet-Key River	77	108.2	134	95.5	183	594.8
Killarney	194	314.0	239	174.3	162	430.1
Killarney Islands	280	143.4	224	56.5	20	33.8
La Cloche	25	63.2	34	25.0	23	44.2
La Cloche Islands	10	3.0	24	9.7	0	0.0
MacGregor-Sampson Islands	81	72.4	162	91.9	23	186.4
Magnetawan-Naiscoot Rivers	27	60.8	49	83.5	167	347.7
Moon-Musquash Islands	426	293.3	620	135.9	42	161.9
Moon-Musquash Rivers	271	420.6	487	268.1	159	1707.6
Muskoka Lakes	0	0.0	0	0.0	6	7.4
Naiscoot Lake	0	0.0	0	0.0	9	5.5
North River	6	0.9	25	96.4	0	0.0
Outer French-Pickerel Islands*	359	125.4	343	80.0	46	51.8

Parry Island	83	108.1	126	67.7	40	410.7
Parry Sound	64	104.9	70	97.6	114	275.8
Philip Edward Island	125	169.7	150	65.4	92	198.0
Pickerel River	235	132.7	257	80.7	170	180.4
Point au Baril	105	126.3	193	109.2	270	708.5
Sandy Bay	29	41.8	52	20.8	39	254.0
Seguin River	1	4.4	9	2.6	17	34.2
Severn River	131	447.3	249	223.4	43	303.0
Shawanaga River	39	60.5	57	29.3	109	271.1
Shebeshekong River	42	200.5	96	95.1	137	585.0
Spider Lake	78	75.7	162	45.1	40	177.1
Still River	7	0.8	2	0.3	5	37.1
Strawberry-Heywood Islands	1	0.1	1	0.0	10	72.7
Sturgeon River	6	100.6	13	34.6	0	0.0
Whitefish River	410	256.9	677	231.5	75	352.8

* indicate incomplete wetland identification due to incomplete imagery coverage within the coastal zone

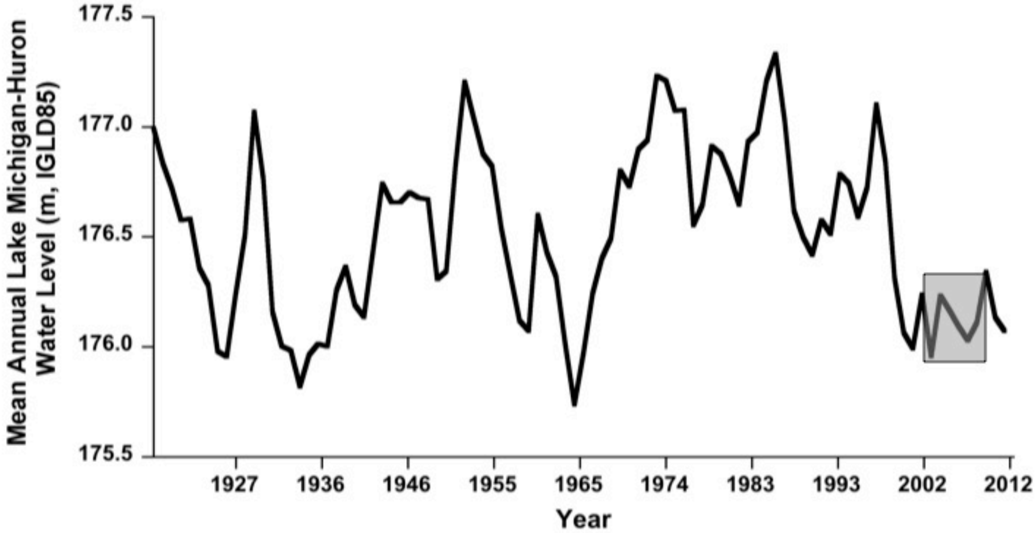


Figure 1. Change in water levels of Lake Huron from 1918 to 2011 (Data from the Canadian Hydrographic Service, Department of Fisheries and Oceans). Highlighted section covers water levels during the period of image acquisition for the MCWI.

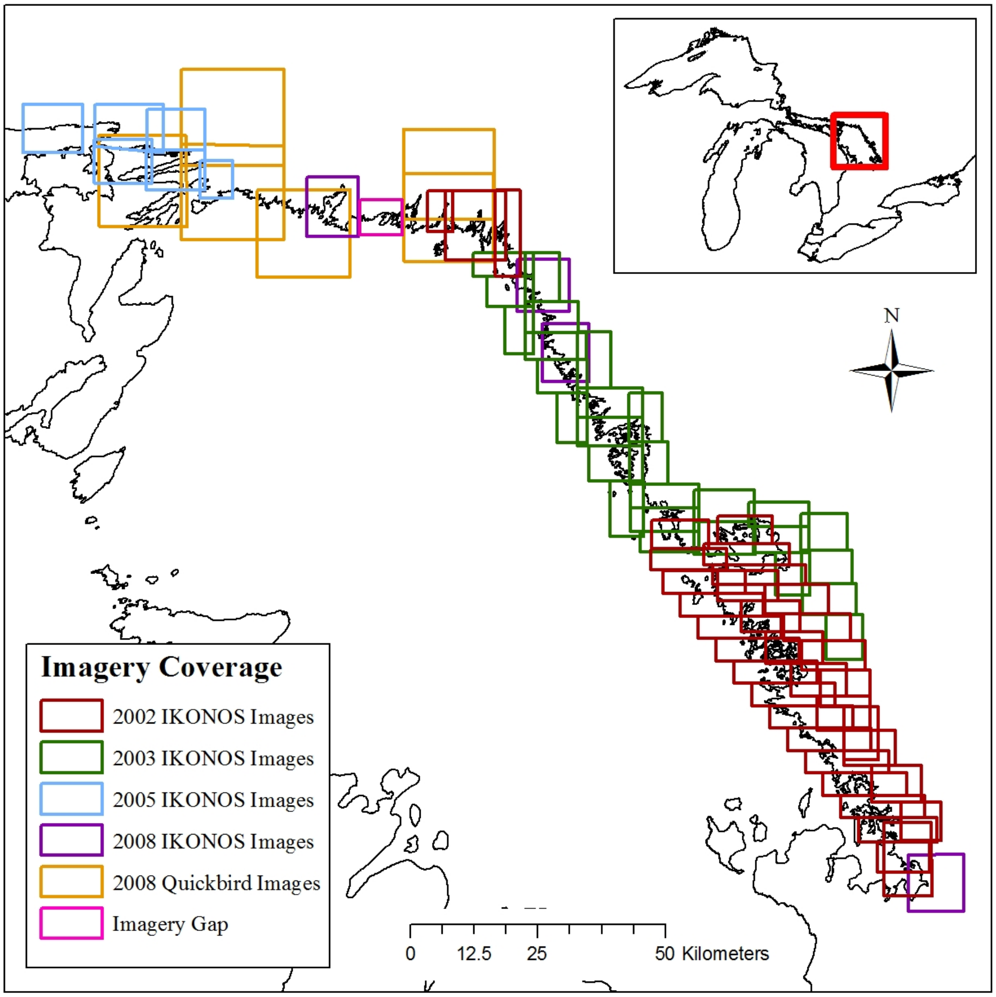


Figure 2. Boundaries of the satellite imagery for eastern and northern Georgian Bay acquired for this project in 2002, 2003, 2005 and 2008. To complete the coverage of northern Georgian Bay, it was necessary to acquire six Quickbird images (indicated in orange). One gap still exists along the northern shore of Georgian Bay as indicated (in pink).

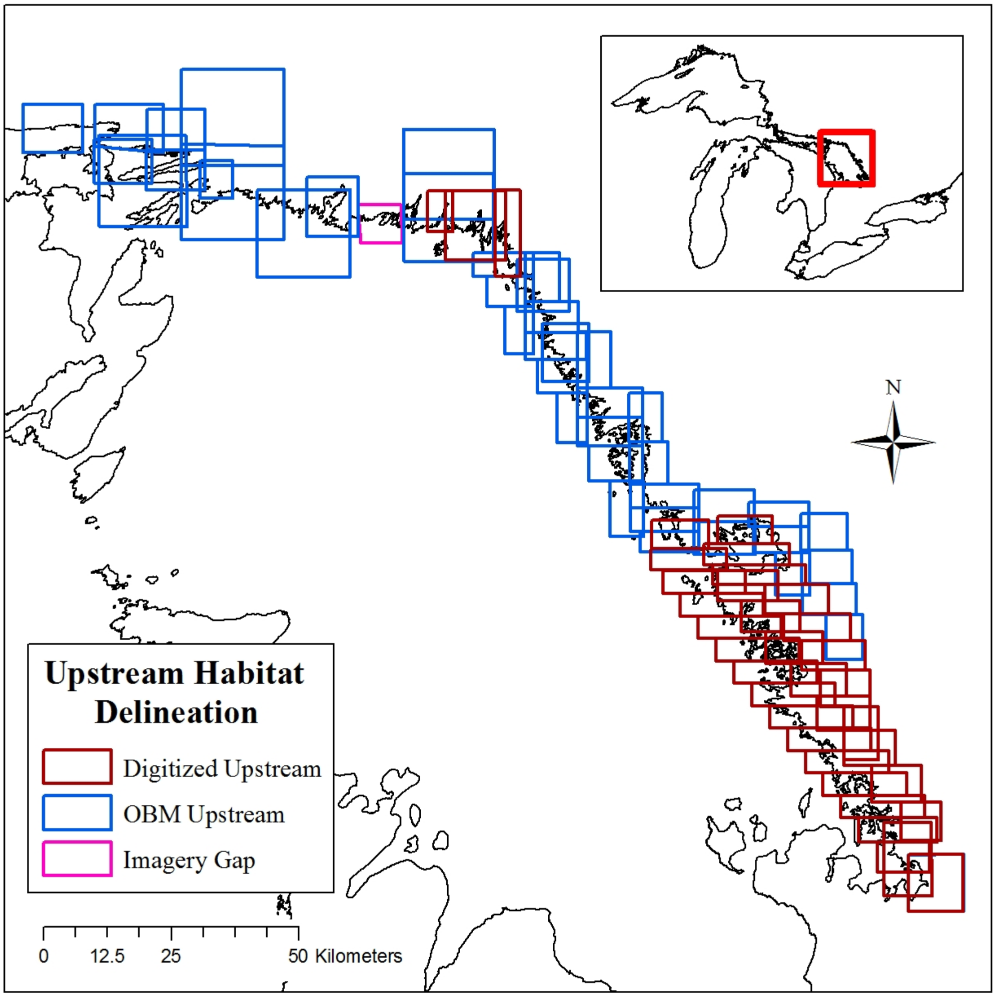


Figure 3. Map showing the boundaries of imagery that was used to digitize upstream habitat (indicated in red). For all other areas, upstream habitat was obtained from corresponding Ontario Base Maps (obtained from OMNR; indicated in blue).

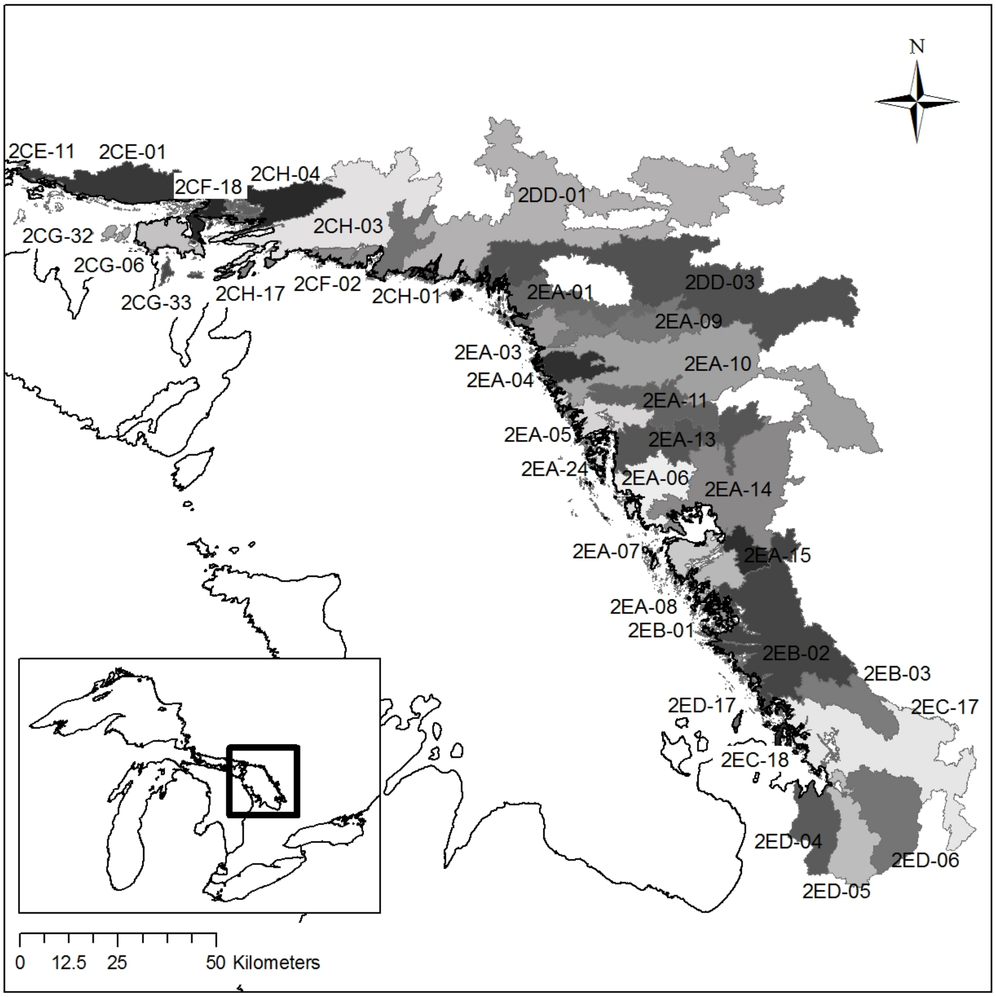


Figure 4. Quaternary watershed location for eastern and northern Georgian Bay, obtained from OMNR. See Table 1 for list of names corresponding to each OMNR code. The shading of the watersheds is for illustrative purposes only.

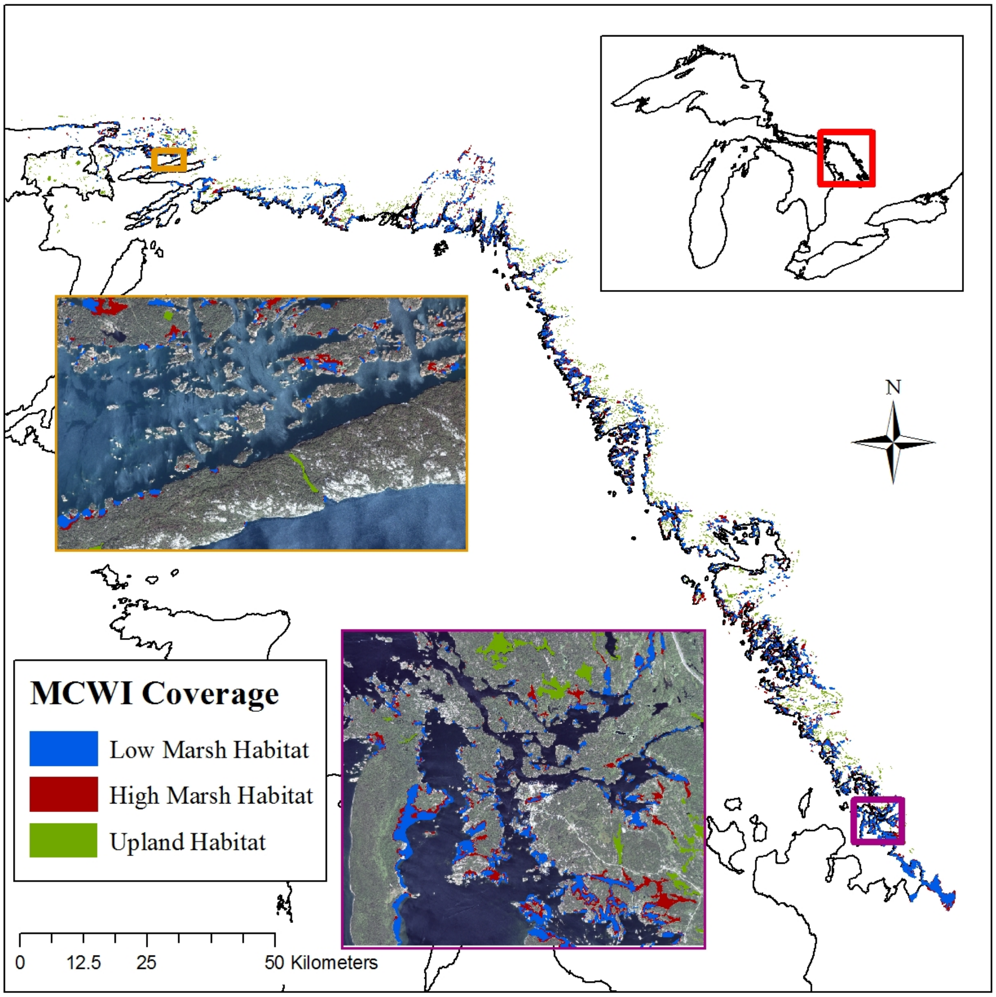


Figure 5. Overview map of the McMaster Coastal Wetland Inventory, covering eastern and northern Georgian Bay within 2-km of the shoreline. The two insets provide a close-up of the region near MacGregor Bay (top; orange) and the Honey Harbour (bottom; purple).

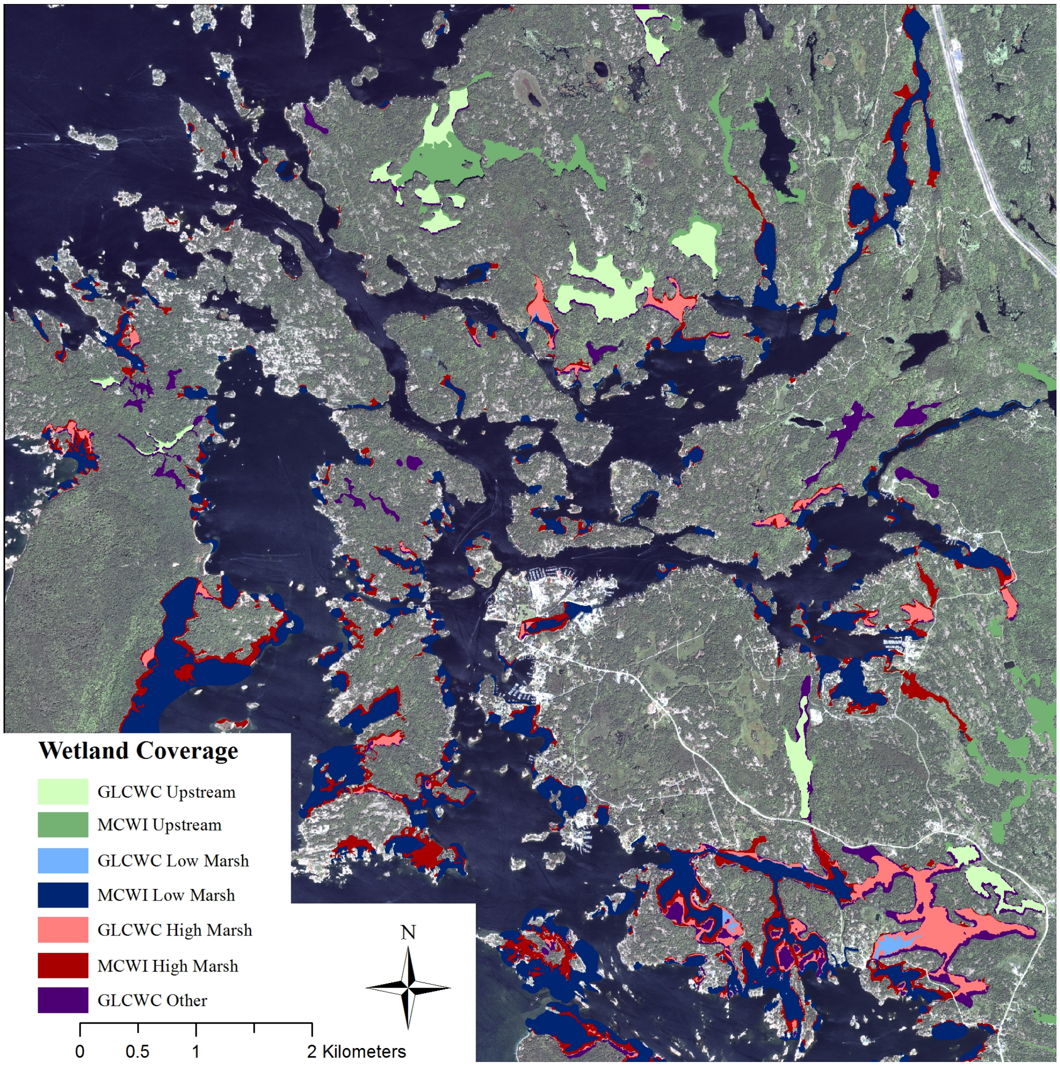


Figure 6. Comparison of coverage between McMaster Coastal Wetland Inventory (MCWI) and Great Lake Coastal Wetland Consortium (GLCWCI) inventory in the Honey Harbour region. Areas identified as Upstream, Low Marsh and High Marsh in the GLCWCI are also in the MCWI; however, areas identified as Upstream, Low Marsh and High Marsh in the MCWI do not occur in the GLCWCI. Areas in purple represent habitat that was included in the GLCWCI but not in the MCWI.

Chapter 2:

Mapping floating and emergent aquatic vegetation in coastal wetlands of eastern Georgian Bay, Lake Huron, Canada.

Reprinted with permission from:

Midwood JD and Chow-Fraser P (2010) Wetlands 30(6):1141-1152.

Abstract

Expansion and contraction of floating and emergent vegetation due to fluctuating water levels has a direct impact on the amount of critical fish habitat in the coastal marshes of Georgian Bay, Lake Huron (Canada). Traditional mapping approaches developed for site-specific studies are too expensive to quantify such changes at the scale of Georgian Bay. Here, we use IKONOS images to develop a classification method (process-tree classification (PTC)), an automated, object-based, image-analysis approach that can produce regional maps of wetland habitat for southeastern Georgian Bay (1466.7 Km). PTC discriminated among six wetland habitat classes (emergent, high-density floating, low-density floating, meadow, water, and rock) in four IKONOS satellite images with a mean accuracy of 87.4%. The PTC was then applied without modification to 17 other IKONOS images collected concurrently in 2002. Based on analysis of 50 randomly chosen wetlands in these images, we estimate that at 2002 water levels, at least 25% of an average wetland (6.5 ha) contains potential fish habitat. Although the PTC developed is specific to the 21 IKONOS images used in this study, the framework is transferable to satellite images acquired in other regions of Georgian Bay, and the approach itself could be applied to other large lakes.

Key Words: coastal wetlands, remote sensing, mapping, habitat, aquatic vegetation, Georgian Bay

Introduction

Wetlands have been globally recognized for their cultural, economic and ecological value (Brander et al. 2006). Barbier et al. (2008) estimate that 50% of the world's marshes have already been lost or degraded. In North America, close to 70% of the wetlands in settled areas of the Laurentian Great Lakes basin have been lost (Snell 1987), and the remaining are still threatened by human development (Niemi et al. 2007) and declining water levels (Mortsch 1998). In the Great Lakes, coastal wetlands help maintain good water quality, regulate watershed hydrology, and provide essential habitat for a number of organisms, especially Great Lakes fishes that use these marshes for spawning and feeding (Wei et al. 2004), and as shelter from predation (Randall et al. 1996).

Government agencies in both Canada and the U.S. have recognized the need to create a comprehensive wetland inventory as a first step to conserve remaining Great Lakes coastal wetlands (Lawson 2004; Fournier et al. 2007). Ingram et al. (2004) used aerial photographs to delineate most of the coastal marshes of Ontario to create an inventory, but due to incomplete coverage of aerial photography, they were unable to identify all marshes along the eastern and northern shores of Georgian Bay (eastern bay of Lake Huron), where some of the most pristine systems occur (Chow-Fraser 2006; Cvetkovic 2008). The Georgian Bay wetlands have a wide spatial distribution and are rarely road-accessible (De Catanzaro et al. 2009), making them too time-consuming and expensive to map using traditional ground surveys and aerial photography (illustrated in Wei & Chow-Fraser 2007).

Since coastal marshes are directly connected to open water of lakes, wetland vegetation responds rapidly to changes in water level and water quality (Lougheed et al. 2001; Hudon, 2004; Chow-Fraser 2006). Mortsch and Quinn (1996) predicted that the increase in temperature resulting from a two-fold increase in atmospheric CO_2 could lead to decreased frequency of precipitation and increased evaporation, in turn causing water levels in Lake Huron to drop by as much as 2.5 m from base case. Between 1999 and 2008, water levels in Georgian Bay fluctuated at approximately 50 cm below the long-term average, and this has led to major shifts in the wetland plant community, from emergent and floating vegetation to increased meadow vegetation (Rokitnicki-Wojcik 2009). Because floating, emergent and submergent vegetation are essential components of fish habitat, wetland managers must be able to identify these critical habitat types and map them. Such maps developed with a method that can be applied consistently would provide wetland managers a means to track changes in fish habitat at regular intervals.

Earlier studies focusing on wetland mapping relied predominantly on two methods: aerial photography, which provides high resolution at a fine spatial scale, or Landsat imagery, which provides low resolution at a coarse spatial scale (e.g. Poulin et al. 2002; Leahy et al 2005). IKONOS satellite imagery, by comparison, has a much higher spatial resolution than Landsat imagery (1 m pan-sharpened), a wider spatial coverage when compared to aerial photography (~100 km^2 per scene), as well as four distinct spectral bands (red, green, blue, and near-infrared) that are useful for automated classification procedures (Lillesand et al. 2004; Wei & Chow-Fraser 2007).

IKONOS imagery has been used to identify wetlands (Fuller et al. 2005) and to produce vegetation maps from multi-temporal images (Dechka et al. 2002) and pixel-based spectral reflectance (Sawaya et al. 2003). Within a Great Lakes context, Wei and Chow-Fraser (2007) successfully used IKONOS imagery and a maximum-likelihood classification (MLC) to map aquatic vegetation in Fathom Five National Marine Park, Canada, and one wetland in eastern Georgian Bay with accuracies greater than 85%. Although this MLC can provide an accurate wetland-specific classification, the need for local ground truth samples (GTS) limits its application at the regional scale, especially for eastern Georgian Bay, where most wetlands are only accessible by boat.

The MLC used by Wei & Chow-Fraser (2007) is also a pixel-based classification, which may limit its usefulness for classifying wetland systems that have a large degree of variation in pixel values (Fuller et al. 2005). Chubey et al. (2006) and Fournier et al. (2007) have shown that an object-based classification system can yield improved accuracy over traditional pixel-based classification systems because image objects combine spectral properties with additional information provided to the user (e.g. shape, size, area, and mean spectral response) and thus minimize errors induced by local variability in pixel values (Navular 2007). This image-object-based approach has been used in terrestrial systems (Laliberte et al. 2004; Silva et al. 2008, Zhou et al. 2008), marine systems (Wang et al. 2004), upland coastal habitats (Grenier et al. 2007; Rokitnicki-Wojcik 2009) and riparian marshland (Dillabaugh & King 2008) but to our knowledge has not yet been used to map fish habitat in freshwater coastal wetlands.

The overall goal of this study is to characterize areal vegetation coverage in southeastern Georgian Bay. First we develop a regionally applicable classification system that minimizes the need for ground truth samples but produces habitat maps of coastal wetlands with an overall accuracy of at least 85%, which is the level of accuracy achieved in Wei and Chow-Fraser (2007). We next apply this classification and identify the dominant types of vegetation and consequently the composition of fish habitat in these coastal marshes. Since declining water level is one of the most serious threats to pristine Georgian Bay coastal marshes, our results should greatly enhance the ability of environmental agencies to track changes in the amount of fish habitat as water levels fluctuate with climate change. Classification of satellite images acquired at different water-level scenarios could also facilitate development of empirical relationships between areal cover of wetland vegetation and water-level, and these could be used to model how further declines in water levels may affect fish habitat quantity and quality.

Methods

Study Sites

Eastern Georgian Bay, Ontario, Canada contains two distinct geographic regions, the limestone Niagara escarpment in the southeast and the Canadian Shield (granite) along the remainder of the coast. Due to complex local geography, most areas of Georgian Bay are not road-accessible, and are therefore relatively undisturbed by human activities. Marshes that have formed along the coast have retained their naturally low nutrient levels, and are characterized by clear oligotrophic water so long as there is adequate exchange between the bay and the marsh; however, when connectivity with Georgian Bay is restricted, runoff from the Canadian Shield can make the water highly coloured with dissolved organic carbon (i.e. dystrophic) (De Catanzaro 2010). The rocky substrate of the Canadian Shield and exposure to wind and wave action limit the amount of sediment deposition in these wetlands and, as a result, meadow development is very limited. The dominant vegetation types tend to be floating (e.g. *Nymphaea*, *Nuphar*, *Brasenia,* and *Zizania*) and submerged herbaceous vegetation (e.g. many species of *Potamogeton* and *Myriophyllum*, etc.), with a narrow fringe of emergent macrophytes (e.g. *Schoenoplectus* and *Eleocharis*).

We collected samples from 16 wetlands for use in the creation and validation of the classification method in this study. Data from five wetlands were used for creation and eleven for validation. There were six wetlands in Tadenac Bay, six in North Bay, two in Severn Sound, and one each in Go Home Bay and Sans Souci (Figure 1; Appendix 1). Tadenac Bay is owned and managed by the Tadenac Club, which has left this property essentially undeveloped since 1896. Wetlands in this bay receive minimal disturbance from human activities. By comparison, many wetlands in North Bay and Severn Sound in the southern region have been subject to recreational and cottage development (2340 year round inhabitants, 2006 Census, Statistics Canada) as well as high levels of boat traffic. The Sans Souci and Go Home Bay wetlands were selected because they are associated with intermediate levels of disturbance when compared to the other study sites. Wetlands used in the application (50) of the classification were randomly selected from a group of 144 marshes that had been manually delineated by interpretation of IKONOS images covering the entire coast of Georgian Bay from Severn Sound in the south to Parry Sound in the north (Figure 1; Appendix 2).

1 – Pre-processing

IKONOS images were acquired in 2002 by Georgian Bay Forever (formerly Georgian Bay Foundation), an environmental non-profit organization, and licensed to McMaster University. Twenty-one images covered the shoreline from Severn Sound to southern Parry Sound (Figure 1). For each image, three bands were available in the visible spectrum (red (RE), green (GR) and blue (BL)) and one band in the near-infrared (NIR). The images were cloud-free, collected at approximately 11:30 am on July 1st, 2002 (EST). This date was sufficiently late in the season to ensure majority of the vegetation had matured.

Images were pre-processed by GeoEye (Dulles, VA, U.S.A.) based on a standard, proprietary, geometrically corrected procedure. They were projected into UTM N17 using the WGS84 datum. All four spectral bands were also pansharpened with the 1 m panchromatic band during this preprocessing phase and the resulting bands had 1 m spatial resolution (GeoEye; Dulles, VA, U.S.A.). Sawaya et al. (2003) suggested that images collected during a single pass would share similar spectral properties and could therefore be used for regional mapping purposes. A preliminary comparison of spectral properties among our 21 IKONOS images was conducted by Rokitnicki-Wojcik (2009) and no significant differences were found. Based on this preliminary analysis and the fact that the imagery had been collected contemporaneously, we assume that the spectral properties of ground features in the five images used in this study are representative of features observed in all 21 images and can theoretically be used to create a model applicable to all (see sample image in Appendix 3).

2 – Image Segmentation

Masking

The PTC was created in Definiens Developer 7.0 software (Definiens®AG, Munchen, Germany). This software uses a decision-tree framework with image objects. Our first step was to isolate the wetland from the surrounding onshore (or upland) vegetation (trees, shrubs) that might share similar spectral properties. We isolated wetland areas using a manually-derived mask layer. A small band of onshore vegetation remained outside of our mask layer to ensure that all wetland vegetation was included; this onshore vegetation will later be identified as meadow vegetation. The lakeward edge of the mask was delineated to include a conservative estimate of submerged aquatic vegetation (SAV) based on SCUBA observations during our field surveys.

Remaining Unclassified

A multiresolution segmentation was used to aggregate the remaining unclassified pixels into image objects. This method employs a user-defined resolution to minimize the average heterogeneity of neighbouring pixels. Chubey et al (2006) used a visual inspection of the image objects created by a multiresolution segmentation to maximize the creation of homogeneous groups. We followed the same process to determine the ideal segmentation parameters. In our final segmentation, we selected the layers associated with the RE, GR and BL bands with which to create the segmentation. The NIR band was excluded due to its coarser pixel size.

We selected a scale factor of 10 within which the heterogeneity would be minimized. This means that large-homogenous regions would be grouped together to form objects greater than 10 m^2 while small heterogeneous regions would be grouped together into objects smaller than 10 m^2. The colour or shape of the input pixels can be used to help identify objects based on composition and degree of homogeneity of neighbouring pixels. Since information our imagery was derived from spectral or colour data, there is no expectation that our classes would be predicted based on shape such as agricultural fields or land plots. As such, we opted to use colour as the main determinant in our segmentation. We set the influence of shape to 10% and colour was used for the remaining 90%.

3 – Training/Testing Sample Selection

During 2007 and 2008 (June to August inclusive), we collected GTSs in 15 of the 16 wetlands (none were collected in Roseborough Bay). In each of the 15 wetlands, all homogeneous ground cover with an area > 4 m^2 were sampled for meadow vegetation (“M”), emergent vegetation (“E”), high-density floating vegetation (“HD”; > 50% coverage within the quadrat), low-density floating vegetation (“LD”; < 50% coverage), rock (“R”) and water (“W”) (see Appendix 1). On average, 26 GTS were collected per wetland. Samples of water and rock were not always collected at each site since these two classes are easily recognizable in imagery. Portions of the 2002 IKONOS images for Tadenac Bay and North Bay were printed off and used in the field to manually delineate all coverage types in 13 wetlands. For three of our wetlands (Garden Channel, Roseborough Bay and Oak Bay) maps were drawn by hand in the field, showing the distribution of aquatic vegetation within the wetlands. Since there had been a 5- to 6-year difference between image acquisition and GTS collection, we did not rely on a direct overlay of the GTS when selecting sample objects (SO). Instead, the GTSs were used to help guide the selection of representative SOs in the IKONOS images (Wei & Chow-Fraser 2007). The use of representative points allowed us to use a comparatively small number of GTSs (n=385) and maps to collect a larger number of SOs (n=1845; Appendix 4).

4 – Sample Analysis

Using a combination of GTS and field-derived maps for 5 wetlands (Black Rock Bay, Coffin Rock, Garden Channel, Oak Bay and North Bay 1), we selected 1076 SOs that corresponded to the 6 habitat classes in the IKONOS images: “E” (n=192), “LD” (n=141) “HD” (n=202), “M” (n=230), “R” (n=158), “W” (n=153) (Appendix 4). We exported the mean values of IKONOS bands (RE, GR, BL and NIR) as well as NIR divided by RE, hue, intensity and saturation associated with each SO. The hue, saturation, intensity (HSI) transformation for hue represents a gradient of colour among the IKONOS bands. HSI transformation for saturation used in this study is an expression of the maximum level of intensity in either the RE, GR, and BL spectrum minus the minimum intensity level in the same bands divided by the original maximum value. Finally, the HSI transformation for intensity uses the largest value in either the RE, NIR or NIR/RE bands (Schowengerdt 1997; Definiens 2007). For each feature an ANOVA was performed in SAS JMP IN 5.1 (SAS Institute, Cary, North Carolina, U.S.A.) to determine if there were significant differences in the value of the feature among the six habitat classes. Once significance was established, we used a Tukey-Kramer analysis to identify differences among the ground-cover classes (data not shown). While this technique provided us with distinct SO properties for water (mean NIR < 250) and rock (RE, GR, BL saturation < 0.23), the remaining four vegetation classes shared too many similar properties for them to be separated solely on the basis of spectral responses. Hence, the Tukey-Kramer analysis was only used as a starting point to identify potentially separable features for vegetation classes. In order to identify the four vegetation classes, we selected new samples from GTSs and a field derived map ("E" (n=37, "LD" (n=16), "HD" (n=92), "M" (n=20), "R" (n=70), "W" (n=31); Appendix 4) in one wetland (Black Rock Bay). For most features, there was a considerable amount of overlap among all vegetation classes. For our final classification, we selected the feature that provided the least amount of overlap and then used relational features to refine the classification.

5 – Process Tree Creation

The process tree was created in a hierarchical manner such that the input IKONOS image bands could be substituted for any of the 21 images (Figure 2). In the first step of the classification, we identified image objects that corresponded to “W” and “R”. An SO was assigned to “W” if it had a mean digital value of ≤ 250 in the NIR band. We classified the object as “R” when the saturation value of the HSI transformation had a mean value <0.23. We grouped the vegetation classes “E” and “LD” first into a category called “wet” vegetation when the HSI transformation for intensity < 0.007. The remaining vegetation (“HD” and “M”) was grouped as “dry” vegetation. The next step was to separate the “dry” and “wet” vegetation into their constituent classes. “E” was separated from “wet” vegetation when the HSI transformation for hue was > 0.8. The remaining “wet” vegetation was classified as “”LD vegetation. “M” was separated from “Dry” vegetation when the mean digital number in the blue band was < 380. The remaining SOs were classified as HD vegetation.

Non-spectral class separation features or “relational” features were also used to correct for misclassifications. Objects identified as “M” that were completely surrounded by water or other aquatic vegetation (“HD”, “LD” or “E”) were assigned to the "HD" class. Conversely, "HD" was converted to "M" if it was in contact with the mask or if it was surrounded by other meadow classes.

6 – Image Classification

The PTC (Figure 2) was used to classify all coastal wetlands in the 21 IKONOS images that were > 2 ha in size (the minimum size for a wetland to be evaluated in the Ontario Wetland Evaluation System (OWES, OMNR 1993)). The RE, GR, BL and NIR and NIR/RE bands along with a mask layer were used for the classification. All classified files were exported into a GIS for further analysis.

7 – Validation

We used maps and ground control points to select a unique set of SOs in eleven independent wetlands to validate the accuracy of the PTC (Figure 1; Appendix 4). Since we used samples to select representative objects, we were once again able to use a small number of GTSs and maps to collect a larger number of SOs. We calculated user- and producer- accuracy as well as the Kappa statistic to test overall and class-specific accuracies. The user-accuracy represents the ratio of correctly classified objects in a class to the total number of objects assigned to that class. By comparison, the producer-accuracy represents the ratio of correctly classified objects in a class to the actual number of ground-truth objects for that class. The Kappa statistic, ranging from 0 to 1, represents the expected agreement between ground truth and classification results, after accounting for the fact that some of the agreement will happen purely by chance (Congalton 1991). Although there is no consensus in the remote sensing community concerning acceptable Kappa thresholds, Kappa values greater than 0.80 are preferred, although values from 0.5 to 0.79 are still desirable (Cohen 1960).

8 – Application

In total, 144 wetlands (>2 ha in size) were identified in the IKONOS imagery that covers the shoreline from Severn Sound to Parry Sound. We used the PTC to classify all 144 wetlands and then imported areal cover associated with all 6 habitat classes into ArcMap 9.2 (ESRI Inc., Redlands, California, U.S.A., 2006). To illustrate how this approach can be used to estimate quantity of habitat classes at a regional scale, we randomly selected 50 of the 144 wetlands (see Figure 1) to estimate fish habitat. As an estimate of generic fish habitat, the "E", "LD" and "HD" classes were merged together to form what we will call "visible fish habitat" (VFH), which does not include habitat containing SAV. By combining VFH with an estimate of open water containing SAV, we produced an estimate of "potential fish habitat" (PFH). We feel that applying the approach to 35% of the wetlands would be sufficient for demonstrating the usefulness of our approach. Total time spent on the application and areal assessment of these 50 wetlands, excluding time spent on the creation of the PTC, was approximately 6.5 hours (8 minutes per wetland); however, this time may vary according to users' familiarity of the software and prior experience with classification.

Results

The overall accuracy for our 11 wetlands was 87.4%. For each wetland, with the exception of West Black Rock Bay, Alexander Bay, North Bay River and Treasure Bay, the overall accuracy was greater than our minimum benchmark accuracy of 85% (Table 1). The overall Kappa statistic for the wetlands ranged from 0.75 to 1.00, with the majority of the sites above 0.8. Only Alexander Bay and North Bay River had Kappa values below 0.8 (0.75 and 0.76 respectively). "W" had the highest overall class accuracy (Kappa = 0.96), with 98.5% of image objects identified correctly. While accuracies associated with "R" and "M" were higher, "HD" was only slightly lower (92.4%, Kappa = 0.89; 93.9%, Kappa = 0.96; 88.4%, Kappa = 0.91, respectively); "E" and "LD" had the lowest overall accuracies (77.9%; Kappa = 0.71 and 74.6%; Kappa = 0.72, respectively). An example of a classified wetland can be found in Figure 3.

The Kappa statistic was used to evaluate the PTC and to determine the degree to which our classification accuracy occurred purely by chance. The majority of our sites fell within either the "excellent agreement" category suggested by Cohen (1960; 0.8-1.0) or the “almost perfect agreement” category suggested by Landis & Koch (1977; 0.81-1.00). Only two sites, Alexander Bay and North Bay River had Kappa values < 0.8 (0.75 and 0.76, respectively). Nevertheless, these wetlands still fall within the “reasonable agreement” category of Cohen (1960; 0.5-0.79) or the “substantial agreement” category of Landis & Koch (1977; 0.61-0.80).

Of the 50 wetlands (mean size 6.6 ±7.9 ha; Appendix 2) randomly selected from the inventory of 144 wetlands distributed along the shoreline of south-eastern Georgian Bay from Severn Sound to Parry Sound (Figure 1), the most common type of vegetation coverage was “LD” (1.1 ha, ±1.3 ha), followed by “E” (0.32 ±0.38 ha), “HD” (0.23 ±0.42 ha) and “M” (0.18 ±0.33 ha). We calculated that on average, the coastal wetlands in southeastern Georgian Bay contained 1.6 ha (±2.0 ha) of VFH, representing 25% of the total wetland area. The wetland masks we created contained a conservative estimate of SAV, and we determined that each wetland contained approximately 6.3 ha (±7.7 ha) of PFH. Assuming that the 50 wetlands that were randomly sampled are representative of all 144 wetlands, we estimate that approximately 230.4 ha of VFH and 907.2 ha of PFH existed in coastal wetlands of the southern half of Georgian Bay during 2002.

Discussion

This is the first time that an approach based on image objects has been used to map aquatic vegetation in Great Lakes coastal wetlands. The PTC represents an accurate and regionally applicable approach to map coastal marsh habitat, especially for the Great Lakes fish community. Using a combination of spectral and relational image object features we are now able to identify and quantify fish habitat in eastern Georgian Bay wetlands. Vegetation is known to have higher reflectance in the NIR spectrum, compared to the visible spectrum. Conversely, water has low reflectance in both the visible spectrum and the NIR (Swain & Davis 1978). Following an initial segmentation process, we identified open water where there were low levels of reflectance in the NIR spectrum. Water had the highest accuracy at the class level largely due to naturally high absorption in the NIR, which limited confusion with other classes.

Eastern Georgian Bay and the coastal wetlands along its shore are located on the Canadian Shield, a large, ancient, granitic rock formation extending across central Canada. Rock is therefore prevalent in many of our wetlands, but probably not as pertinent a feature to classify in other Great Lakes coastal wetlands. Because there is a high reflectance in the three bands of the visible spectrum (RE, GR, and BL) associated with this land-cover feature, there were very low values for the HSI saturation transformation, and consequently, there was minimal confusion with other land-cover classes, and "R" was the second most accurately classified feature.

Schmidtt & Skidmore (2003) were able to use hyperspectral data to discriminate among vegetation classes to the species level in salt-marshes. This was likely achievable because of a greater number of bands with narrower bandwidth in their study. By comparison, the four bands of the IKONOS imagery in this study only allowed for a modest degree of separation. Two of our vegetation classes ("E" and "LD") were mixtures of both vegetation and water and this combination allowed for an accurate separation of this "wet" vegetation from "dry" vegetation. Vegetation typically has high reflectance in the NIR spectrum but, when vegetation is combined with water, this reflectance is diminished; therefore "wet" vegetation was classified when there were lower values in the HSI intensity transformation.

Ullah et al. (2000) compared spectral reflectance in three different species of emergent vegetation and found that *Schoenoplectus* spp. (the dominant emergent species in Georgian Bay) had the lowest reflectance. They also noted that vertical vegetation (emergent) decreased spectral interference from substrates since long stems can block or intercept electromagnetic radiation from reaching or reflecting off the substrate. We used this differential influence of the substrate (water in the case of "wet" vegetation) to isolate "E" from "LD" vegetation. Due to a greater influence of water in "LD" vegetation, we classified image objects as

"E" when there was a high HSI hue transformation. The "LD" and "E" vegetation classes had the lowest overall accuracy. The majority of the error in their classification was due to confusion with the other "wet" vegetation type, which we attribute to the mixture of both water and vegetation, and this will be discussed in more detail later.

We considered both "M" and "HD" vegetation to be "dry" vegetation. This is despite the presence of water in "HD" (defined as greater than 50% vegetation coverage) because the dense vegetation made this type of aquatic vegetation more spectrally similar to "M" than to either "E" or "LD". In preliminary analyses, we tried to use the NIR/RE ratio to further separate "HD" from "M" classes since we anticipated that meadow vegetation would have a higher ratio due to greater vegetation density and less influence from water. Unfortunately, this approach did not prove useful, and instead, we were able to separate out "M" using low values in the BL band.

Overlap in spectral signature, especially between "HD" and "M" classes, made it necessary to use additional logic-based features to correct for any misclassifications that arose during spectral separation. Based on our knowledge of wetland zonation, we know that wetlands progress lakeward from terrestrial vegetation (not inundated), to meadow vegetation ("M"; seasonally inundated), to aquatic vegetation (permanently inundated; "HD", "LD" and "E") and finally to open water ("W") with submerged aquatic vegetation (SAV). The transitional zone from meadow to aquatic vegetation is usually dominated by “E” and as water depth increases, “LD” and “HD” begins to take over. We have visited dozens of wetlands in Georgian Bay, and we rarely observe open water bordered by meadow vegetation without the presence of a transitional aquatic zone. It is equally rare for us to find dense floating patches mixed with meadow vegetation. Finally, we know that meadow vegetation does not exist in patches surrounded by any other habitat classes within the aquatic zone or in isolation without contact with the mask layer. We were able to use this type of logic to create rule sets that could correct for misclassifications of the various land-cover classes.

Using a combination of spectral and relational features, the PTC was able to separate our six classes with an overall accuracy greater than the benchmark of 85% that we had set out as a goal for this study. Four of the eleven wetlands did not meet our target overall accuracy of 85% (i.e. Alexander Bay, North Bay River, West Black Rock Bay and Treasure Bay; see Table 1). In all cases, the low accuracy associated with “LD” and “E” classes decreased the overall accuracy for these wetlands. The lowest overall accuracy was found for North Bay River (77%). This site was also associated with low user accuracy, which was attributed to a higher incidence of “E” being misclassified as “M”. This site was one of the few wetlands that contained *Typha* spp. (cattails), which is atypical for other wetlands in Georgian Bay (Croft and Chow-Fraser 2007). Although this species is clearly emergent vegetation, it is known to have higher reflectance in all four

IKONOS bands compared with *Schoenoplectus* spp. (bulrushes; Ullah et al. 2000), which is the more typical dominant emergent plant in Georgian Bay wetlands. It is probable that the presence of *Typha* spp. in parts of the emergent zone increased reflectance and caused the incorrect classification.

While there was some variation among sites in terms of both overall accuracy and Kappa values, the PTC in general performed extremely well for four of the classes, but only moderately well for "E" and "LD". Dillabaugh and King (2008) also had low accuracy when classifying emergent vegetation. Since it occurs in a transitional zone, emergent vegetation shares spectral properties with both onshore and aquatic vegetation. In this study, the similar spectral properties of LD and E prevented accurate discrimination of these two classes. We could have improved our overall accuracy by keeping these two classes together in the "wet" vegetation class, but, since these two vegetation types represent distinct types of fish habitat (Cvetkovic 2008), we opted to keep them separate. Incorporation of bathymetric data or a digital elevation model (DEM) into the PTC should decrease confusion between these two classes. This combination of spectral data with water depth would allow a more precise classification since our field observations indicate that emergent vegetation is typically found in shallower water than are low-density floating vegetation. Even though "M" and "HD" are already associated with high accuracies, the DEM may also improve the discrimination between these vegetation classes.

In this study, we were not able to map SAV, which is a large portion of fish habitat. In the clear waters of Fathom Five National Marine Park, Wei and Chow-Fraser (2007) were able to map SAV. In a preliminary study, we found that the dystrophic waters of eastern Georgian Bay prevented us from accurately mapping SAV using IKONOS imagery. We have observed, however, that SAV tends to be found in most open-water areas immediately adjacent to LD or HD vegetation, and extend lakeward to at least 5m depth. Since many fish species are dependent on SAV for spawning and nursery habitat (Randall et al. 1996), it is important that future mapping efforts incorporate this component so that total potential fish habitat can be quantified for Georgian Bay.

To map aquatic vegetation in this study, we utilized IKONOS satellite imagery. This satellite launched in 1999 and was one of the first satellites to provide fine-resolution multispectral data. In the intervening years, other satellites have been launched that offer finer-resolution (Quickbird) and a greater number of multispectral bands (WorldView-2). Although these new satellites may offer better discrimination among wetland vegetation classes, there is value to continuing to use IKONOS at the regional scale in order to track changes in vegetation cover to support long-term monitoring programs because of availability of archival images.

Hardisky et al. (1986) first documented the need for a rapid, cost-effective approach to map coastal wetlands nearly 25 years ago. Because of pressures from human development, the majority of mapping efforts by Canadian environmental agencies in the 1990s have focused on wetlands of Lakes Erie and Ontario (Ball et al. 2003), leaving unmapped the most pristine coastal wetlands in remote areas of eastern Georgian Bay. Using the PTC, we have mapped a subset of some of the larger wetlands (>2 ha) in the southern half of Georgian Bay in order to establish a baseline for vegetation coverage during a period of sustained low water levels. If these maps are updated at regular intervals, we can begin to develop a relationship between water level and aquatic vegetation coverage. This relationship will help managers take actions to cope with a forecasted decline in water level of up to 2.5 m by 2050 (Mortsch & Quinn 1996). From field surveys conducted between 2002 and 2006, we know that these Georgian Bay wetlands contain some of the most diverse fish and macrophyte communities in the Great Lakes basin (Croft & Chow-Fraser 2007; Seilheimmer & Chow-Fraser 2007).

Since fish are known to associate with different densities of vegetation (Jude & Pappas 1992; Jacobus & Webb 2005) as well as different morphological forms (Dibble et al. 1997; Cvetkovic 2008), we have mapped “E”, “HD” and “LD” vegetation separately. These are important fish habitat classes, accounting for 25% of the average wetland in this region. We estimated a total of 230.4 ha of VFH and 907.2 ha of PFH, and these data could be used in conjunction with fish community surveys to develop species-area relationships for fish species in coastal wetlands of Georgian Bay. Future investigations should use this approach to identify wetlands that may be vulnerable to biodiversity loss when they become diminished in size because of human or natural disturbance.

Coastal wetlands in south-eastern Georgian Bay have high biodiversity, but the average size is small (8.7 ha) compared with wetlands in Lake Erie (15.9 ha) and Lake Superior (39.2 ha) (Ingram et al. 2004). While these wetlands are small, there are many more of them. For instance, there are 568 wetland units in the south-eastern shoreline of Georgian Bay alone, compared with only 881 for the whole of Lake Ontario (Ingram et al. 2004). It would have been too costly and difficult to map the many small wetlands in Georgian Bay using the traditional approach involving field-truthing and aerial photography, and may explain why the inventory prepared by the Great Lakes Coastal Wetland Consortium omitted many of these (Midwood et al. unpub. data). We speculate that these small wetlands are important to the fish community of Georgian Bay because the oligotrophic nature of these wetlands (DeCatanzaro et al. 2009) may force the fish community to forage by constantly migrating from wetland to wetland. Hence, small proximate wetlands may function as alternate habitats for metapopulations of fish, and this hypothesis should be properly addressed in future studies.

Wei & Chow-Fraser (2007) demonstrated that broad vegetation groups in wetlands can be separated using a pixel-based method, provided the plants have unique spectral properties. We have extended their research to utilize a process tree classification to accurately map broad vegetation groups over a large region with minimal field surveys and using an object-based approach. This regionally applicable PTC can provide biologists with a tool to rapidly map wetland vegetation and characterize coastal fish habitat. In view of the anticipated drop in water level in the Great Lakes due to global climate change, we encourage environmental agencies to adopt this or similar methods to continue to map wetlands in Georgian Bay to track losses and gains in fish habitat.

Acknowledgements

This study could not have been carried out without funding and access to IKONOS imagery provided by Georgian Bay Forever (GBF). We thank all members of the Georgian Bay Association (GBA), especially Mary Muter, who helped us gain access to the coastal wetlands and generously provided their cottages for us to stay in. Other funding for this research came from the Ontario Ministry of Natural Resources and the Natural Sciences and Engineering Research Council of Canada. Thank you to Maja Cvetkovic, Daniel Rokitnicki-Wojcik, Lyndsay Smith, Henry Heiser and Anhua Wei for their helpful suggestions and comments. Finally, thank you to everyone in the Chow-Fraser lab who helped collect the field data.

References

Ball H, Jalava J, King T, Maynard L, Potter B, Pulfer T (2003) The Ontario Great Lakes coastal wetland atlas: a summary of information (1983–1997). Environment Canada

Barbier E B, Koch E W, Silliman B R, Hacker S D, Wolanski E, Primavera J, Granek E F, Polasky S, Aswani S, Cramer L A, Stoms D M, Kennedy C J, Bael D, Kappel C V, Perillo G M E, Reed D J (2008) Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319:321-323

Brander L M, Florax R J G M, Vermaat J E (2006) The empirics of wetland valuation: a comprehensive summary and a meta-analysis of the literature. Environmental & Resource Economics 33:223-250

Chow-Fraser P (2006) Development of the Water Quality Index (WQI) to assess effects of basin-wide land-use alteration on coastal marshes of the Laurentian Great Lakes. In: Coastal wetlands of the Laurentian Great Lakes: health, habitat and indicators. In: Simon, T.P., Stewart, P.M. (Eds), Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat, and Indicators. Authorhouse, Bloomington, IN, pp. 137-166

Chubey M S, Franklin S E, Wulder M A (2006) Object-based analysis of Ikonos-2 imagery for extraction of forest inventory parameters. Photogrammetric Engineering & Remote Sensing 72(4):383-394

Cohen J (1960) A coefficient of agreement for nominal scales, Educ. Psychol. Measurement 20(1): 37-46

Congalton R G (1991) A review of assessing the accuracy of classifications of remotely sensed data. Remote Sensing of Environment 37: 35-46

Croft M V, Chow-Fraser P (2007) Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of Great Lakes coastal marshes. Journal of Great Lakes Research 33(Special Issue 3): 172-197.

Cvetkovic M (2008) Factors affecting macrophyte and fish distribution in coastal wetlands of Georgian Bay. Master's Thesis. McMaster University

Dechka J A, Franklin S E, Watmough M D, Bennett R P, Ingstrup D W (2002) Classification of wetland habitat and vegetation communities using multi-temporal IKONOS imagery in southern Saskatchewan. Canadian Journal of Remote Sensing 28(5): 679-685

DeCatanzaro R, Cvetkovic M, Chow-Fraser P (2009) The relative importance of road density and physical watershed features in determining coastal marsh water quality in Georgian Bay. Environmental Management. 44(3):456-467

DeCatanzaro R (2010) Factors influencing coastal marsh water quality in Georgian Bay, Ontario. Master's Thesis. McMaster University

Definiens (2007) Definiens Developer 7.0 Reference Book. Definiens®AG, Munchen, Germany

Dibble E D, Killgore K J Harrel S L (1997). Assessment of fish-plant interactions. Miscellaneous Paper A-97-6. U.S. Army Engineer Waterway Experiment Station, Vicksburg, MS

Dillabaugh K A, King D J (2008). Riparian marshland composition and biomass mapping using Ikonos imagery. Canadian Journal of Remote Sensing 34(2): 143-158

Fournier R A, Grenier M, Lavoie A, Helie R (2007) Towards a strategy to implement the Canadian Wetland Inventory using satellite remote sensing. Canadian Journal of Remote Sensing 33, Suppl. 1. S1-S16

Fuller L M, Morgan T M, Aichele S S (2005) Wetland Delineation with IKONOS High-Resolution Satellite Imagery, Fort Custer Training Center, Battle Creek, Michigan. U.S. Geological Survey, Scientific Investigations Report 2006-5051, 8 p

Grenier M, Demers AM, Labrecque S, Benoit M, Fournier R A, Drolet B (2007) An object-based method to map wetland using RADARSAT-1 and Landsat ETM images: test case on two sites in Quebec, Canada. Canadian Journal of Remote Sensing 33(Suppl. 1): S28-S45

Hardisky M A, Gross M F, Klemas V (1986) Remote sensing of coastal wetlands. Bioscience 36(7): 453-46

Hudon C (2004) Shift in wetland plant composition and biomass following low-level episodes in the St. Lawrence River: looking into the future. Canadian Journal of Fisheries and Aquatic Sciences 61:603-617

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

Jacobus J, Webb P W (2005) Using fish distribution and behaviour in patchy habitats to evaluate potential effects of fragmentation on small marsh fishes: a case study. Journal of Great Lakes Research 31(Supplement 1): 197-211

Jude D J, Pappas J (1992) Fish utilization of Great Lakes coastal wetlands. Journal of Great Lakes Research 18(4): 651-672

Laliberte A S, Rango A, Havstad K M, Paris J F, Beck R F, McNeely R, Gonzalez A L (2004) Object-oriented image analysis for mapping shrub encroachment from 1937 to 2003 in southern New Mexico. Remote Sensing of Environment 93: 198-210

Landis J R, Koch G G (1977) An application of hierarchical Kappa-type statistics in the assessment of majority agreement among multiple observers. Biometrics 33(2): 363-374

Lawson R (2004) Coordinating coastal wetlands monitoring in the North American Great Lakes'. Aquatic Ecosystem Health & Management 7(2):215-221

Leahy M G, Jollineau M Y, Howarth P J, Gillespie A R (2005) The use of Landsat data for investigating the long-term trends in wetland change at Long Point, Ontario. Canadian Journal of Remote Sensing 31(3):240-254

Lillesand T M, Kiefer R W, Chipman J W (2004) Remote Sensing and Image Interpretation. 5th Edition. John Wiley & Sons Publishing. Hoboken, New Jersey

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

Mortsch L (1998) Assessing the impact of climate change on the Great Lakes shoreline wetlands. Climate Change 40:391-416

Mortsch L, Quinn F H (1996) Climate change scenarios of Great Lakes Basin ecosystem studies. Limnololgy and Oceanography 41(5):903-911

Navulur K (2007) Multispectral Image Analysis Using the Object-Oriented Paradigm. CRC Press, Taylor & Francis Group. Boca Raton, FL, USA

Niemi G J, Kelly J R, Danz N P (2007) Environmental indicators for the coastal region of the North American Great Lakes: introduction and prospectus. Journal of Great Lakes Research 33(sp3):1-12

OMNR (1993) Ontario Wetland Evaluation System. Northern Manual. Ontario Ministry of Natural Resources (OMNR), No.50254

Poulin M, Careau D, Rochefort L, Desrochers A (2002) From satellite imagery to peatland vegetation diversity: how reliable are habitat maps?. Conservation Ecology 6(2):16

Randall R G, Minns C K, Cairns V W, Moore J E (1996) The relationship between an index of fish production and submerged macrophyte and other habitat features at three littoral areas in the Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 53 (Suppl 1):35-44

Rokitnicki-Wojcik D (2009) Use of remote sensing and GIS for wetland monitoring and assessment. Master's Thesis. McMaster University

Sawaya K E, Olmanson L G, Heinert N J, Brezonik, P L, Bauer M E (2003) Extending satellite remote sensing to local scales: land and water resource monitoring using high-resolution imagery. Remote Sensing of Environment 88: 144-156

Schowengerdt R (1997) Remote sensing: models and methods for image processing. Academic Press, Burlington, Massachusetts

Scmidt K S, Skidmore A K (2003) Spectral discrimination of vegetation types in a coastal wetland. Remote Sensing of Environment 85: 92-108

Seilheimer T S, Chow-Fraser P (2007) Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. Journal of Great Lakes Research 33(Special Issue 3): 154-172

Silva T S F, Costa M P F, Melack J M, Novo E M L M (2008) Remote sensing of aquatic vegetation: theory and applications. Environmental Monitoring Assessment 140:131-145

Snell E A (1987) Wetland distribution and conservation in southern Ontario. Inland Waters and Land Directorate, Environment Canada, Burlington, Ontario

Statistics Canada (2006) Georgian Bay Township 2006 Community Profile. Available via http://www12.statcan.gc.ca/census-recensement/2006/dp-pd/prof/92-591/index.cfm?Lang=E Accessed 2 Oct 2009

Swain P H, Davis S M (1978) Remote Sensing: The Quantitative Approach. McGraw-Hill, Inc. West Lafayette, Indiana

Ullah A, Rundquist D C, Derry D P (2000) Characterizing spectral signatures for three selected emergent aquatic macrophytes: a controlled experiment. Geocarto International 15(4): 29-39

Wang L, Sousa W P, Gong P (2004) Integration of object-based and pixel-based classification for mapping mangroves with IKONOS imagery International Journal of Remote Sensing 25(4): 5655-5668

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

Wei A, Chow-Fraser P (2007) Use of IKONOS imagery to map coastal wetlands of Georgian Bay. Fisheries 32(4): 167-173

Zhou W, Troy A, Grove M (2008) Object-based land cover classification and change analysis in the Baltimore metropolitan area using multitemporal high resolution remote sensing data. Sensors 8: 1613-1636

Table 1: Error matrix for the 12 wetlands used to test the accuracy of our process tree. Kappa is expressed as a value from 0 to 1. The label "N/A" means that for a specific wetland, no features of that class were present.

		Class						
Wetland	**Accuracy Method**	**Meadow (%)**	**High-Density Floating (%)**	**Low-Density Floating (%)**	**Emergent (%)**	**Rock (%)**	**Water**	
West Black Rock Bay	Producer	92.6	59.6	81.9	53.3	100	97.2	
	User	82.8	100	49.5	86.7	84.4	99.0	
	Kappa	0.92	0.59	0.80	0.51	1.00	0.96	
							Overall Accuracy (%)	82.3
							Average Kappa	0.80
Treasure Bay	Producer	77.3	83.9	79.7	71.3	100	95.4	
	User	66.5	32.8	89.2	80.4	100	100	
	Kappa	0.75	0.83	0.77	0.69	1.00	0.89	
							Overall Accuracy (%)	81.4
							Average Kappa	0.82
Miners Creek	Producer	100	100	98.4	100	N/A	95.5	
	User	100	100	100	98.3	N/A	100	
	Kappa	1.00	1.00	0.98	1.00	N/A	1.00	
							Overall Accuracy (%)	99.2
							Average	1.00

							Kappa	
Thunder Bay	Producer	100	N/A	89.4	80.8	80.5	100	
	User	98.1	N/A	86.5	86.8	100	100	
	Kappa	1.00	N/A	0.88	0.79	0.80	1.00	
							Overall Accuracy (%)	92.2
							Average Kappa	0.89
Alexander Bay	Producer	100	N/A	52.8	36.4	90.2	100	
	User	100	N/A	15.7	100	100	98.6	
	Kappa	1.00	N/A	0.49	0.34	0.90	1.00	
							Overall Accuracy (%)	79.4
							Average Kappa	0.75
Roseborough Bay	Producer	100	100	78.0	93.0	83.0	97.2	
	User	95.0	100	86.0	85.0	100	99.0	
	Kappa	1.00	1.00	0.77	0.92	0.82	0.90	
							Overall Accuracy (%)	93.0
							Average Kappa	0.90
North Bay 2	Producer	100	89.3	66.6	59.0	100	100	
	User	75.1	82.9	90.6	59.7	100	100	
	Kappa	1.00	0.88	0.63	0.57	1.00	1.00	
							Overall Accuracy (%)	85.3
							Average	0.85

								Kappa	
	Producer	94.6	94.3	74.2	100	98.2	100		
	User	100	71.0	94.1	100	100	100		
	Kappa	0.94	0.94	0.73	1.00	0.98	1.00		
North Bay 3								Overall Accuracy (%)	93.9
								Average Kappa	0.93
	Producer	100	100	79.6	98.0	53.2	100		
	User	98.3	100	86.0	78.0	100	100		
	Kappa	1.00	1.00	0.77	0.98	0.53	1.00		
North Bay 5								Overall Accuracy (%)	91.1
								Average Kappa	0.88
	Producer	95.4	100	68.3	38.2	N/A	95.0		
	User	57.5	83.4	64.6	68.0	N/A	100		
	Kappa	0.95	100	0.65	0.36	N/A	0.86		
North Bay River								Overall Accuracy (%)	77.0
								Average Kappa	0.76
	Producer	100	100	47.2	63.0	100	98.6		
	User	100	94.1	63.4	78.9	100	91.6		
	Kappa	1.00	0.99	0.42	0.61	1.00	0.97		
South Bay 2								Overall Accuracy (%)	86.4
								Average	0.83

							Kappa	
All Wetlands	Producer	96.4	91.9	74.2	72.1	89.5	98.1	
	User	88.5	84.9	75.1	83.8	98.3	98.9	
	Overall	92.4	88.4	74.6	77.9	93.9	98.5	
	Kappa	0.96	0.91	0.72	0.71	0.89	0.96	
							Overall Accuracy (%)	87.4

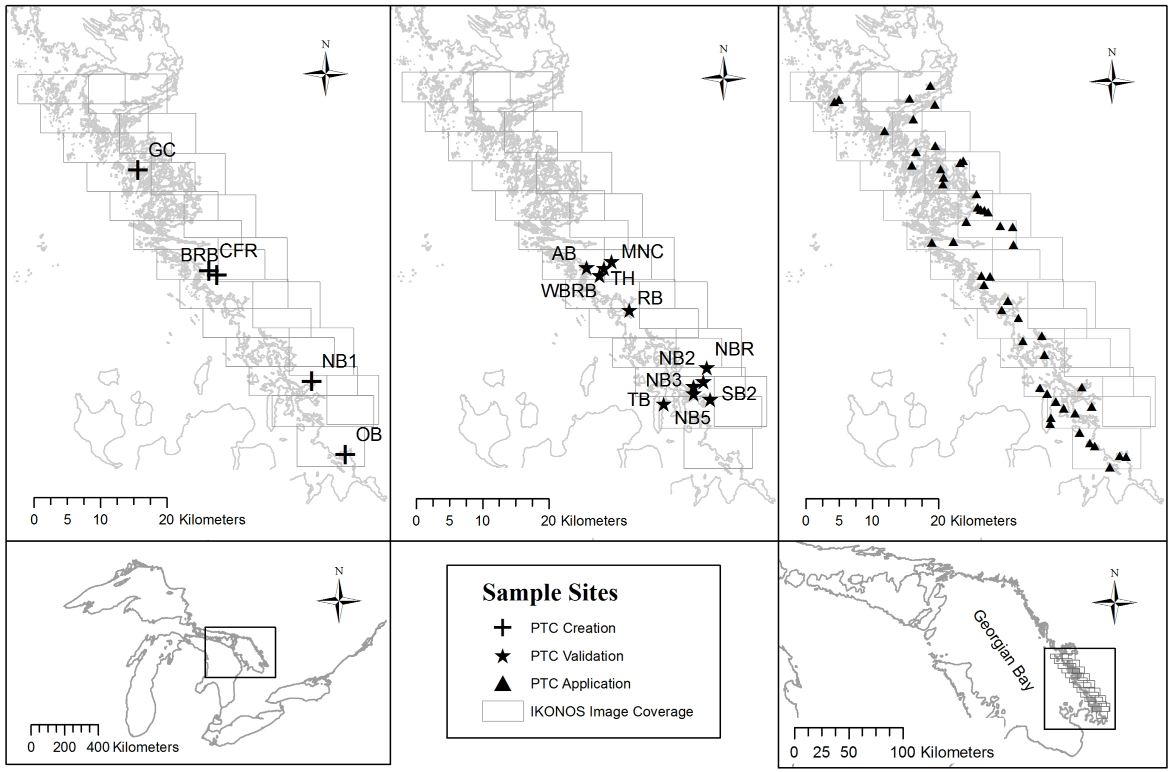


Figure 1: Map of eastern Georgian Bay identifying coastal wetlands used for creating (crosses) and validating (stars) the process tree classificaiton (PTC). The PTC was then applied to 50 wetlands located along the south-eastern shore of Georgian Bay (triangles). The square boxes outline the coverage of the 21 IKONOS images used in this study.

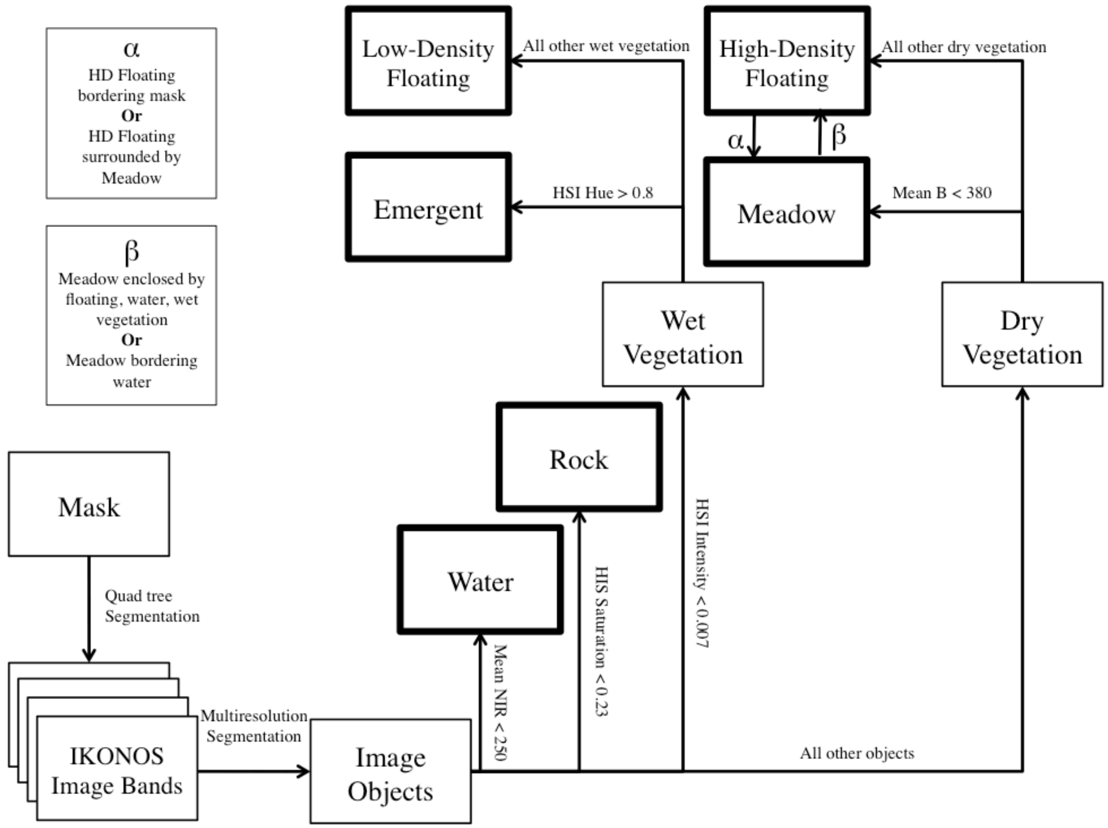


Figure 2: Flow chart illustrating the steps in the Process Tree Classification (PTC). Small text next to each arrow indicates the feature(s) that were used to separate the image into 6 different land cover classes.

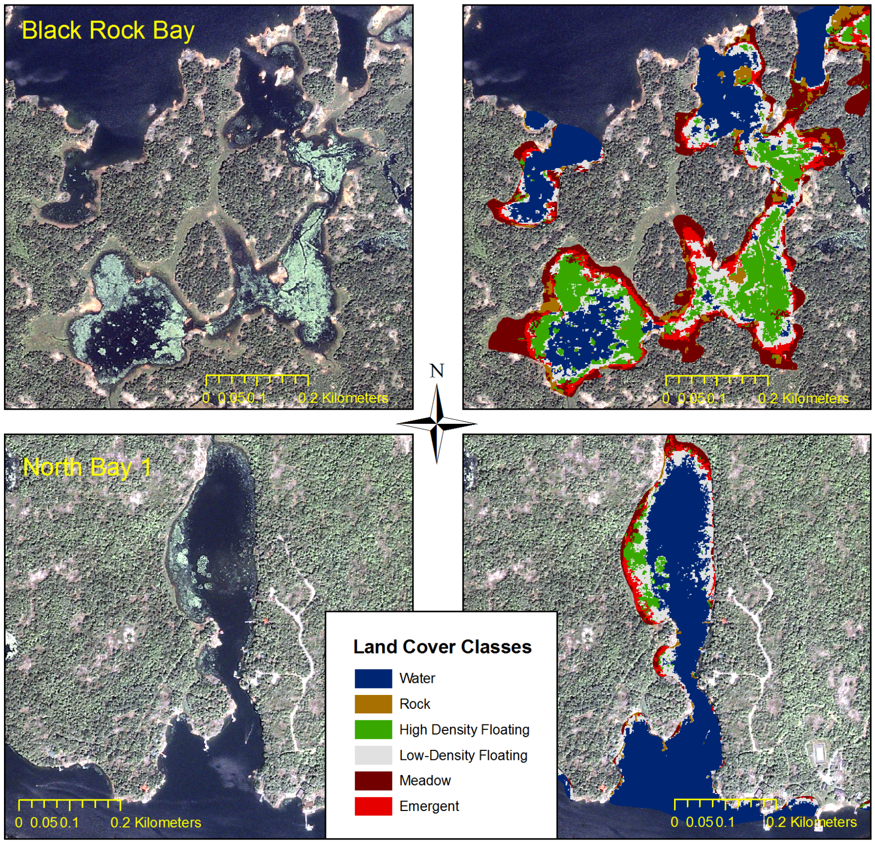


Figure 3: Comparison of a) original IKONOS image with b) classified image using **PTC** method. Red=emergent vegetation, green = dense floating vegetation, grey=sparse floating vegetation, maroon=meadow vegetation, blue=water and brown=rock. This image was taken in July 2002 of the Tadenac Bay region of eastern Georgian Bay.

Chapter 3:

Predicting the response of submerged aquatic vegetation to low water levels in coastal wetlands of Georgian Bay, Lake Huron.

Midwood JD and Chow-Fraser P (2012) (In submission, Journal of Great Lakes Research).

Abstract

Distribution of submersed aquatic vegetation (SAV) in coastal wetlands of large lakes varies annually due to changing lake levels. Photo-interpretation of a time series of images has been used to track changes in SAV in wetlands with clear water, but in dystrophic wetlands, underwater vegetation is not discernable. We tested the applicability of a published relationship between light extinction coefficient (EXT) and the maximum depth of colonization (Z_{max}) in the dystrophic wetlands of eastern and northern Georgian Bay, and found that it underestimated Z_{max}. This necessitated development of a rule-based model to estimate potential SAV habitat over a range of water levels for a wetland using a digital elevation model. Using only EXT and exposure as parameters, the GIS model predicted SAV habitat with an accuracy of 83.5%; the accuracy dropped slightly to 83.3% when depth was substituted for EXT. With this substitution, we applied the model to six wetlands in Georgian Bay, to determine how SAV coverage would change with a decline in water levels from 176.4 to 174.4 (m, asl). For four sites, the areal extent of SAV declined with a decrease in water level, while for the remainder, aquatic plant beds remained unchanged or increased slightly in cover. These differences can be attributed to variation in wetland geomorphology, and we recommend that this approach be used throughout Georgian Bay to provide an estimate of underwater fish habitat as a function of different water levels in Lake Huron.

Keywords: Great Lakes, Water Level, Light Extinction, Exposure, Depth, GIS

Introduction

Submerged aquatic vegetation (SAV) is an essential component of coastal wetland habitat. It helps to regulate water temperature, provide food for waterfowl, create habitat for fishes, provide fish with protection from predation, and regulate water movements (Orth and Moore, 1984; Jude and Pappas, 1992). Due to their central role in coastal wetland habitat creation and maintenance, environmental factors which control SAV distribution have been extensively studied and reviewed (Hudon et al., 2000; Madsen et al., 2001; Lacoul and Freedman, 2006). The maximum depth of colonization (Z_{max}) by SAV tends to be a function of the amount of light that can penetrate to the bottom, which is quantified by the light extinction coefficient (EXT*;* rate at which light is absorbed/attenuated) (Duarte and Kalff, 1987; Middelboe and Markager, 1997; Hudon et al., 2000; Squires et al., 2002). Light attenuates predictably below the water surface because of absorption and scattering by organisms and suspended inorganic and organic particles (algae, bacteria, sediment) as well as absorption by water molecules themselves (Dennison et al., 1993).

Clear water naturally absorbs light in the near-infrared wavelength (Chambers and Prepas, 1988). By contrast, dystrophic water (containing dissolved organic compounds) not only absorbs light in the near-infrared range of the spectrum, it also increases absorption in the visible spectrum (Chambers and Prepas, 1988). As water becomes enriched with organic matter, it becomes darker and prevents light from penetrating to the bottom, thus decreasing Z_{max} (Chambers and Prepas, 1988; Houser, 2006). Previous investigators have attempted to identify a generalized Z_{max} for SAV growth (Chambers and Kalff, 1985; Sand-Jensen and Madsen, 1991; Schwarz et al., 2000), but the exact value seems to vary by species (Canfield et al., 1985; Middelboe and Markager, 1997), lake trophic states, and geographic location (Chambers and Kalff, 1985). Despite regional variation, however, equations have been developed that relate Z_{max} to EXT (Canfield et al., 1985; Duart and Kalff, 1987; Hudon et al., 2000; Zhu et al., 2007); however, Z_{max} seems to be consistently under-estimated in clear water sites (Canfield et al., 1985), and the only reliable way to obtain Z_{max} in some environments is to measure it directly.

Submerged aquatic vegetation can be prevented from establishing in shallow water (<1 m) by exposure to wind and waves. Studies have found a strong correlation between the minimum depth of SAV colonization (Z_{min}) and stream velocity (Chambers, 1987; Chambers et al., 1991; Grace and Pugesek, 1997; Riis and Hawes, 2003; Capers and Les, 2005). The force of waves affects SAV by physically damaging or completely removing plants and by removing seedlings (Keddy, 1982, 1983, 1985). Similarly, ice can create a "scour zone" in water less than 1 m deep (Stewart and Freedman, 1989). These physical disturbances are prime factors in limiting SAV development in shallow water. Murphey and Fonseca (1995) developed the Relative Exposure Index (REI) to estimate wind and wave exposure at all depths using remotely sensed imagery or maps. The REI combines the average amount of time the wind travels along 1 of 16 compass bearings, the mean wind speed in each of the 16 directions, and the fetch, where fetch represents the distance the wind travels across an open body of water.

While exposure and light are two of the strongest abiotic factors that can influence the distribution of SAV, there are other important factors that cannot be ignored. Similar to terrestrial species, aquatic plants take in a majority of nutrients, particularly phosphorus and nitrogen (Bini et al., 1999; Riis and Biggs, 2001; Xie et al., 2005) from sediment or substrate in wetlands (Barko et al., 1991; Xie et al., 2005). Ideally, sediment must be less than 75% sand (Barko and Smart, 1986) and less than 20% organic matter (Capers and Les, 2005), containing a mixture of clay (particle size <2 μm), silt (particle size >2 μm, <50 μm), and sand (particle size >50 μm) (Gafny and Gasith, 1999; Xie et al., 2005). Sand provides a solid foundation for SAV to establish roots, and silt and clay contain essential nutrients like phosphorus and nitrogen (Chambers et al. 1991; Gafny and Gasith, 1999; Istvanovics et al., 2008). Temperature (Dale, 1986) and hydrostatic pressure (Kautsky, 1988; Wetzel, 2001) can be limiting factors in water depths greater than 10 m, but in shallower depths, changes in temperature through the season can only control life-history traits such as onset of flowering and senescence (Dale, 1986).

Coastal wetlands are naturally dynamic systems and the macrophytes within them rely on natural water-level variation to maintain species diversity (Keddy and Reznicki, 1986; Gathman et al., 2005) and community structure (Quinlan and Mulamoottil, 1987; Wei and Chow-Fraser, 2005). In many wetlands of the Great Lakes, human activities have degraded water quality and lowered overall biodiversity (Chow-Fraser, 1998). By contrast, coastal wetlands of eastern and northern Georgian Bay have been spared such disturbances (Chow-Fraser, 2006; Cvetkovic and Chow-Fraser, 2011), and are known to support high biodiversity of macrophytes (Croft and Chow-Fraser, 2007) and fishes (Seilheimer and Chow-Fraser, 2007). The main threat is a decline in water levels that began in the 1970s (Sellinger et al., 2008) and which is predicted to drop even farther over the next decades by up to 1.75 m (Mortsch and Quinn, 1996; Magnuson et al., 1997; Angel and Kunkel, 2010). Since SAV is such a critical component of fish habitat, there is an urgent need to model the impact of future water-level declines on the areal extent of SAV in these pristine wetlands of eastern and northern Georgian Bay.

Wei and Chow-Fraser (2007) used an approach involving interpretation of satellite imagery to map SAV communities in several clear-water wetlands of

Fathom Five National Marine Park near Tobermory, Lake Huron. Midwood and Chow-Fraser (2010) attempted to apply the same remote-sensing approach to map SAV in many wetlands of eastern Georgian Bay, but the method was unsuccessful because of the dystrophic nature of the water. The large geographic extent of eastern and northern Georgian Bay makes it impractical to map wetland aquatic vegetation using traditional and time-consuming field surveys alone. One approach that is gaining momentum is the use of empirical relationships based on a limited number of physical parameters to predict SAV distribution in order to map critical fish habitat at a particular site (Havens et al., 2002; Cho and Poirrier, 2005; Jin et al., 2007).

The overall goal in this study is to develop a regionally applicable method for mapping SAV distribution in Georgian Bay wetlands that can be used to determine the impact of declining water levels on SAV colonization. The literature is unanimous in identifying light as the most limiting factor for colonization of SAV, especially in deep-water habitat. Although there are published relationships between EXT and Z_{max} (e.g. Hudon et al., 2000), we were uncertain they could be applied directly to our study without validation because of the dystrophic nature of the water in Georgian Bay. Therefore, we had to first collect data to test the applicability of published relationships and if necessary develop a relationship that can be applied regionally to predict Z_{max} for SAV using light extinction coefficients measured in Georgian Bay. Since light is only one of several variables that influence where plants can be established in a coastal wetland, we developed an approach that would integrate all the factors into a single model, including the effect of water levels and exposure on plant establishment in shallow depths. We therefore developed a rule-based Geographic Information System (GIS) to estimate amount of SAV habitat over a range of water levels for any given wetland with appropriate bathymetric information. In developing the rules, we deliberately chose variables that are relatively easy to measure or derive to maximize usefulness of our approach. Finally, we applied this rule-based model to predict future distributions of SAV in six wetlands of eastern and northern Georgian Bay under different water-level scenarios. Results of this study will provide a better understanding of how declining water levels may impact the quantity of fish habitat in a region as large as Georgian Bay, Lake Huron, and help to identify appropriate management actions to adapt to these changes.

Materials and Methods

Light Extinction

Replicate light readings were collected with a spherical quantum submersible light sensor (Li-Cor, Lincoln, Nebraska USA) in nine wetlands (see Table 1) to generate mean light readings at 50-cm intervals from the surface to at least 5 m depth. The negative slope of $\log_e$-transformed light readings against depth was an estimate of the extinction coefficient (EXT). At the same time we measured the light readings, we used SCUBA to determine the maximum depth of colonization by macrophytes.

Macrophyte Surveys

Data used to develop the rule-based model to predict the distribution of SAV were collected during macrophyte surveys conducted in summer 2008 and 2009. In total, 18 wetlands were sampled in both years, and of these, five were sampled in both 2008 and 2009 (Figure 1; Table 2). We established 218 ground control points (GCPs). At each point, all aquatic macrophytes occurring in a 1-m^2 quadrat were identified to species, water depth was measured, and geographic coordinates (decimal degrees) were noted. In shallow water (<1 m), depth was measured with a ruler; in water deeper than 1 m, either a depth meter or a calibrated rope was used. We used snorkel and mask or SCUBA to complete macrophyte surveys in water deeper than 1 m.

Variables Absent in the Rule-Based Model

Several studies have suggested that sediment composition (Gafny and Gasith, 1999; Xie et al., 2005) and nutrient content (Bini et al., 1999; Riis and Biggs, 2001; Xie et al., 2005) play important roles in determining SAV distribution. We conducted a preliminary study to evaluate the relative importance of substrate type and found no clear relationship between the presence of SAV and surface sediment composition (top 20 cm) in terms of sand (particle size > 50 µm), silt (2 µm < particle size < 50 µm), clay (particle size < 2 µm), or organic content (Midwood, unpub. data). Of the 82 samples collected in 14 wetlands in summer 2008, none were growth-limiting (i.e., >75% sand; Barko and Smart, 1986). Samples in wetlands tended to be dominated by silt (41.1% ± 21.5%), with some sand (36.5% ± 32.6%) and clay (22.2% ± 15.9%; Table 3). The organic content was uniformly low (8.1% ± 7.9%), and below the 20% threshold deemed by Capers and Les (2005) as being too high for SAV colonization. There was also little variation in nutrient level in sediment samples. Total phosphorus content in these sediment samples varied from 0.04 mg/g to 0.72 mg/g, with a mean of 0.42 mg/g (± 0.14 mg/g), while that of total ammonia ranged from 0.001 mg/g to 0.27 mg/g, with a mean of 0.039 mg/g (±0.048 mg/g), both low concentrations compared with sediments in wetlands elsewhere in the Great Lakes (see Crosbie and Chow-Fraser, 1999; McNair and Chow-Fraser, 2003). Due to a lack of any discernable trends in our sediment data as well as the challenges associated with creating sediment maps for multiple wetlands, we excluded sediment composition and nutrient content from further consideration. Based on these observations, we concluded that while substrate type and nutrient content could influence plant growth in terms of growth rate or biomass of SAV (Barko and Smart, 1986; Capers and Les, 2005), these factors were not as important as light and wave exposure for delineating the maximum and minimum depths of colonization.

Rule-Based Model Development

Since SAV in nature occurs within a vegetation matrix together with emergent and floating vegetation, we built the rule-based model on a previously classified GIS vegetation map that had been created from IKONOS satellite imagery (Midwood and Chow-Fraser, 2010), and overlaid depth measurements to produce a DEM. Depth was used as a surrogate for light to estimate the maximum depth of plant colonization (Z_{max}), since light is not a variable that can easily be applied in a GIS. To evaluate the appropriateness of this substitution, we compared the accuracy of our model (SAV_{GB}) with a similar model where Z_{max} was determined based on each wetland's unique light extinction coefficient ($SAV_{GB\text{-}EXT}$). To create $SAV_{GB\text{-}EXT}$, equation 3 was substituted into our SAV_{GB} in place of a single depth value for Z_{max}. Extinction data were available for 194 of the 218 GCPs discussed previously. We applied both models to these GCPs to determine the relative accuracy of each model.

The degree of wave exposure was used to delineate the upper limit of SAV colonization. We used a modified version of the Relative Exposure Index (REI; Murphey and Fonseca, 1995) to estimate exposure for each GCP we obtained in the field. In a GIS, the GCP was used as a centroid from which 16 lines were drawn in the four cardinal directions as well as 12 equally spaced intermediate bearings (Figure 2). Lines were terminated when they reached land or dense vegetation, as inferred from the imagery. Average wind speed and the amount of time the wind traveled in a specific direction were acquired from a buoy located offshore in Georgian Bay (Integrated Science Data Management, Fisheries and Oceans Canada, Buoy C45143, 2008). By combining these metrics into equation 1 (adapted from Murphey and Fonseca, 1995, where v = average wind speed (m/s), p = proportion of time wind blows in a given direction, and f = fetch (m)), we were able to estimate exposure for each GCP. All values of REI presented in this paper refer to the REI for each GCP.

$$REI = Mean \sum_{n=16}^{i} v_i p_i f_i$$

Eq. 1

A third variable, the ratio REI/Depth (REI/D), was used to account for the disproportional influence of exposure in shallow water areas (Keddy, 2000). Higher values represent areas with either high exposure in deep water or any level of exposure in shallow waters. Typically though, REI values associated with shallow water are lower than those associated with deeper water since shallow sites are located adjacent to the shore and will thus have minimal exposure in at least one of the 16 bearings.

All data from field surveys and REI calculations were entered into SAS JMP 8.0 (SAS Institute, Cary, North Carolina, U.S.A.). Two-thirds of the database

(146 GCPs) were randomly selected for use in the development of our model. A recursive partitioning approach (also known as a decision tree) was adopted. In this approach, data were split based on groupings of variables (depth, REI, and REI/D) that best predicted the response (binary presence/absence of SAV). These splits are made recursively to form a decision-tree structure. A drawback to this approach is that it can over-fit data, which results in erroneous groupings, especially as the number of data points decreases (Hothorn et al., 2006). The three variables, depth, REI, and REI/D (all of which can be calculated with a DEM and satellite imagery), were entered into the partition analysis.

Model accuracy was tested with the remaining one-third of the database (72 GCPs) and a contingency analysis. Overall accuracy was calculated as the percent of correctly identified points (0,0 or 1,1). The Kappa statistic was also calculated, incorporating the fact that some of our observed agreements will occur purely by chance (Congalton, 1991). Kappa values from 0.5 to 0.79 are considered reasonable, but values greater than 0.80 are ideal (Cohen, 1960). From the entire database (all 218 points) we randomly selected 1/3 of the GCPs in an iterative approach to estimate the standard errors of the accuracy of the model. Ten replications were used to provide confidence intervals for the Kappa statistic as well as overall accuracy.

Rule-Based Model Application

Our model, SAV_{GB}, was applied to six wetlands in Georgian Bay: Alexander Bay (AB), Coffin Rock (CFR), North Bay 1 (NB1), North Bay 5 (NB5), Red Sand Beach (RSB), and Treasure Bay (TB; Figure 1). In order to apply the model in a spatial context, rasters or grids had to be developed for each wetland that provided both bathymetric (DEM) and exposure data. In a GIS, rasters are used to create a grid where each cell has a unique value (Lillesand et al., 2004), in our case for depth and exposure.

In summer 2009, depth GCPs were collected in six wetlands when water levels were at an elevation of 176.4 m (asl). In shallow water (< 1 m), we measured depth with a ruler; for deep water, we employed a calibrated rope or a depth meter. We recorded the geographic coordinates at each location where depth measurements were taken. We digitized the shoreline (0 m) of each wetland in GIS using 2008 IKONOS imagery. This shoreline was combined with the depth GCPs collected in the field and a Kriging method (Oliver and Webster, 1990) was used to create a DEM with 1-m cells and depth expressed as metres above sea level (m, asl).

To produce areal maps of potential SAV coverage, we needed to know the REI for every part of the wetland. For each 1-m cell, we would then know the depth, the REI, and ratio of REI/D. Our rule-based model could then be applied to each individual cell to evaluate whether SAV could potentially grow there.

Exposure rasters were created so that every point in the DEM could be linked to a specific REI value. Within each wetland, a random subset of 20 locations was selected from the DEM (Beyer 2004). The REI for each of these locations was then calculated. To create the raster, we interpolated these points using the Kriging method (Oliver and Webster, 1990).

SAV_{GB} was applied to each of the six wetlands where depth was derived from the DEM and REI was derived from the exposure raster. REI/D was calculated for each cell in the wetland based on the DEM and the exposure raster. The result was a map showing areas of potential SAV habitat within each wetland. These maps were then exported into a GIS and the area of potential SAV habitat for each wetland was calculated. To estimate SAV habitat under lower water level scenarios, the depth value in the DEM was successively lowered from an elevation of 176.4 m (asl) at intervals of 0.1 m. This allowed us to simulate water level declines in each wetland from 176.4 to 174.4 (m, asl). A 2-m change was selected to represent the maximum decline predicted by 2080 (Angel and Kunkel, 2010). The potential area of SAV habitat for each wetland at each elevation was entered into JMP 8.0 for further analysis. Potential area was expressed as a proportion of the area of SAV habitat at an elevation of 176.4 m (asl).

Results

Wetland descriptions

In 2008, we surveyed SAV in 82 quadrats from 14 wetlands, whereas, during 2009, we surveyed 136 quadrats in only 9 wetlands (Table 2). Majority (71.0%) of these data were collected from four wetlands that were sampled extensively in both 2008 and 2009: CFR (35), NB1 (45), AB (50), and TB (25). Water depth at each sampling point ranged from 0.2 m to 10.0 m with a median value of 1.5 m (mean = 2.3 m ± 1.9 m). Within this range, SAV occurred between depths of 0.2 m and 6.1 m. The deepest occurrence was found in AB, a clear–water site, largely influenced by Georgian Bay waters. In majority of sites, however, Z_{max} was observed between depths of 4.0 m and 4.7 m (Table 1).

We calculated exposure for all 218 points based on a modification of the equation provided by Murphey and Fonseca (1995; Eq. 1). On average, REI was 51.8 (±69.6) with majority of the values falling between 42.5 and 61.1. The most exposed site was found in Sturgeon Bay (STB) in the North Channel region, where REI was 810.1. Two other sites, Silver Island (SLI) and RSB, also located in northern Georgian Bay, had high levels of exposure with REI values of 216.9 and 199.9, respectively.

Light Equation

Light data and Z_{max} were collected in nine wetlands (Figure 1). EXT ranged from lows of 0.25 and 0.26 in two clear water sites, RSB and AB, respectively, to a high of 1.26 in Waterfall Bay (WFB; Table 1). Similarly, Z_{max} was deepest in both RSB and AB at 8.49 m, while the shallowest Z_{max} of 4.24 m was recorded in Roseborough Bay (RB; Table 1).

Hudon et al. (2000) sampled Z_{max} in the St. Lawrence and Ottawa Rivers across a gradient of water clarity to develop an equation to predict Z_{max} from EXT for a large region (Eq. 2). We chose this relationship to use in our comparison since it was based on data collected at a comparable latitude and included similar species such as *Vallisneria americana*.

$$Z_{\max} = 3.08(EXT)^{-0.675} \qquad \text{Eq. 2}$$

Using a paired t-test, we compared the Z_{max} values predicted by this equation to the observed Z_{max} in nine wetlands in eastern Georgian Bay (Table 1). The predicted Z_{max} values were significantly lower than the observed Z_{max} values by more than a meter (paired t-test, $P < 0.01$, df = 8). We then used data from the nine wetlands to create an empirical relationship between Z_{max} and EXT specific to Georgian Bay as follows:

$$Z_{\max} = \left(\frac{1.25}{EXT}\right) + 3.32 \qquad \text{Eq. 3}$$

where EXT is wetland specific and Z_{max} is expressed in meters (R^2=0.879; p=0.0002).

Rule-Based Model

SAV_{GB} was designed in a decision tree structure with a threshold value for a specific variable being used to separate data into two categories at each node. In total, there were seven nodes and each node, with the exception of two, ends with either SAV being present or absent (Table 4; Figure 3).

Rule-Based Model Accuracy

Based on 194 GCPs, there was little difference in accuracy between the SAV_{GB} model (83.3%) and the $SAV_{GB\text{-}EXT}$ model (83.5%). Using an independent database comprised of 1/3 of the original SAV database, we found that SAV_{GB} had an overall accuracy of 83.3% and a Kappa of 0.626 (Table 5). This Kappa value was within the range suggested by Cohen (1960) that is considered to have a reasonable level of agreement. An iterative approach that re-sampled from the entire database ten times to estimate the accuracy had an overall accuracy of 82.0% (± 3.0%) and a Kappa of 0.581 (± 0.073).

Model Application

The rule-based SAV model was applied to six wetlands in eastern Georgian Bay where bathymetry data and exposure were available. The result was an areal estimate of potential SAV habitat that ranged from 2.1 ha in NB5 to 15.8 ha in TB. By modeling decreased depth in each of these sites at 10-cm intervals, we were able to estimate how a drop in water level would influence SAV distribution. In four of the six wetlands (AB, CFR, NB1, and TB) there was an inverse relationship between potential SAV habitat and water levels (Table 6; Figure 4). The two remaining wetlands, RSB and NB5, showed variable responses but never dropped below 70% of the maximum areal extent of potential SAV habitat, suggesting that they were not as severely impacted. Of the six wetlands, RSB was the only one that showed a net increase in SAV habitat with a decline in water levels. In NB5, there was a steady decline in SAV habitat to an elevation of 175.3 m (asl), which was followed by an increase in habitat to an elevation of 174.4 m (asl) where potential SAV habitat was nearly equal to that at 176.4 m (asl) (Table 6; Figure 4).

Discussion

With forecasted declines in water levels in Lake Huron of up to 1.75 m over the coming decades, it is important to understand and predict how coastal habitat will respond. In eastern and northern Georgian Bay there are over 3500 coastal wetlands, majority of which are small (<2 ha; Midwood et al. 2012). The large number of coastal wetlands makes site-specific surveys impractical and therefore a regionally applicable model provides the most efficient method for mapping habitat. Midwood and Chow-Fraser (2010) produced a method for mapping emergent and floating vegetation, but they were unable to map SAV due to dystrophic water; models presented here offer a feasible approach to quantify fish habitat in coastal marshes on a regional basis, provided there are available DEM data.

Many studies have used the relationship between light extinction and maximum depth of colonization to identify the depth limits of SAV (Chambers and Kalff, 1985; Chambers and Prepas, 1988; Middelbøe and Markager, 1997; Hudon et al., 2000). We determined that an equation developed by Hudon et al. (2000) consistently underestimated Z_{max} in Georgian Bay wetlands by greater than 1 meter. Therefore, we developed an equation to predict the maximum depth of SAV colonization specifically for wetlands in northern and eastern Georgian Bay. This type of equation provides a useful means for assessing the effective surface area colonized by SAV (Hudon et al., 2000), but since it does not consider factors that limit SAV establishment in shallow water where light is not limiting (i.e., exposure), we needed to incorporate other parameters to generate a model that provides a more accurate distribution of SAV (Cho and Poirrier, 2005). Another drawback of using EXT is that it requires *in situ* measurement of light from each wetland of interest, and this limits its usefulness across a region. Since majority of coastal wetlands in eastern and northern Georgian Bay have not been surveyed, the $SAV_{GB\text{-}EXT}$ model, which was developed with EXT, would have been largely irrelevant. Fortunately, we found that the $SAV_{GB\text{-}EXT}$ model was only marginally better than the SAV_{GB} model, which only relied on depth information (83.5% vs 83.3% accuracy), and we were therefore able to apply this model across wetlands that had appropriate bathymetric information (e.g., a DEM) without the need for field surveys.

Despite the wide breadth of literature that suggests that sediment composition (Gafny and Gasith, 1999; Xie et al., 2005) and nutrient levels (Bini et al., 1999; Riis and Biggs, 2001; Xie et al., 2005) are important for SAV growth and establishment, we did not find any relationship between surface sediment composition and macrophyte distribution. While this is in agreement with Gafny and Gasith (1999), they did find a clear link between SAV and sediment structure in terms of subsurface sediment composition, a variable that we did not measure for this study. We believe that the lack of a clear link between SAV and sediment in this study is due to lack of variability in sediment composition and nutrients levels within our wetlands. Had we focused on SAV coverage or biomass, which have been linked to sediment composition (Barko and Smart, 1986; Capers and Les, 2005), we might have found it necessary to include sediment as a parameter in our models. Therefore, we concluded that light and exposure are likely more important in determining presence or absence of vegetation, while nutrient and sediment composition may be responsible for the biomass of macrophyte found in Georgian Bay wetlands.

Exposure is known to play a major role in limiting macrophyte growth, especially in shallow waters (Chambers, 1987; Chambers et al., 1991; Grace and Pugesek, 1997; Riis and Hawes, 2003; Capers and Les, 2005). Not only does it limit macrophyte establishment and dispersal of propagules (Keddy, 1982, 1983, 1985), it also affects the composition of sediment and nutrient levels (Madsen et al., 2001). Therefore, in this study, exposure indirectly incorporated multiple variables including: sediment composition, nutrient content, wave scouring, and physical removal. Since exposure can be estimated using satellite imagery (Murphey and Fonseca, 1995; Wei, 2007; Cvetkovic, 2008), it proved to be the best variable for the development of this regional model.

Based on our regional model (SAV_{GB}), we have established empirical relationships linking SAV to environmental variables specifically for eastern and northern Georgian Bay. This is an essential link in forecasting how SAV will respond to future changes in water level fluctuations (Hudon et al., 2000). SAV_{GB} represents the area of a wetland where SAV could potentially grow (i.e., their potential niche). Currently, the specific relationship between SAV-potential and SAV-realized niches is unknown. However, a study by Wei and Chow-Fraser

(2008) in Lake Ontario found a significant positive relationship between potential habitat and occupied habitat for emergent vegetation.

Threats from declining water levels could drastically alter pristine coastal habitat in eastern Georgian Bay. Sustained low water levels in this region have already altered floating and emergent vegetation and caused an overall loss of fish habitat (Midwood and Chow-Fraser, 2012). With continued declines and the projected loss of interannual water level variation (Magnuson et al., 1997; Angel and Kunkel, 2010), coastal habitat will continue to change, making it important to document current distributions and forecast how the quantity and quality of habitat will respond to a different water-level regime. Through the application of SAV_{GB} to six wetlands in eastern and northern Georgian Bay we have demonstrated that the response of SAV to declining water levels is variable but often results in a loss of potential SAV habitat, with four of the six wetlands showing a decline in potential SAV habitat of up to 76.7%.

As water levels decline, it is often possible for SAV to migrate into deeper water where new habitat has become available (i.e., no longer limited by low light levels; Gradual slope - Figure 5). The variable results observed in our application of SAV_{GB} are consistent with this theory, but the degree of adaptation is dependent on the geomorphology of the wetland (Albert et al., 2005). While no studies have specifically documented the typical geomorphology of Georgian Bay wetlands, in our experience, they are typically small (<2 ha; Midwood et al. 2012)

Press) and fall into two main groups, embayments (both protected and open) and fringing habitat. The main difference between these two groups is the degree of connectivity to Georgian Bay and the degree of exposure, whereby fringing habitats form along exposed coastal areas and protected embayments in the numerous bays and inlets along the extensive shoreline of eastern and northern Georgian Bay. Within these two groups, factors such as size and dominant substrate type vary depending on local watershed characteristics (DeCatanzaro and Chow-Fraser, 2011).

In our study, the two wetlands that did not show a decline in SAV habitat were a small protected embayment (NB5) and a large fringing wetland (RSB). For these wetlands, a decline in water level of 2 m resulted in an increase (RSB) or no change (NB5) in potential SAV habitat. Red Sand Beach is a highly exposed wetland, where the SAV in shallow water was highly influenced by wind and wave action and where maximum SAV habitat could not become established until water levels dropped another 2 m relative to the 2009 levels. At 175.7 m asl, SAV habitat reached its minimum because there is a steep drop-off that limits the size of the potential habitat zone in 2009 (Steep slope – Figure 5). As water levels declined and this steep zone was no longer covered by water, the potential area of SAV habitat expanded, and this accounted for the observed increase following a decline in water level.

In the case of NB5, the small protected embayment was occupied by SAV, emergent, and floating vegetation in 2009, and this would dry up as water levels initially declined, leading to an initial loss of potential SAV habitat; however, because there is a small channel that connects this embayment with Georgian Bay, we must assume that SAV can migrate as water levels decline from 176.4 (m, asl) to 174.4 (m, asl) and thus result in an overall net loss of only 4% of the initial SAV habitat. Nevertheless, it is important to note that the NB5 wetland shifted from being primarily a protected embayment to a fringing wetland. While exposure is known to influence aquatic vegetation (Chambers, 1987; Riis and Hawes, 2003), studies have documented corresponding changes to both the fish and invertebrate communities (Randall et al., 1996; Burton et al., 2002), with richness for both maximized at intermediate levels of exposure. In eastern Georgian Bay, Cvetkovic (2008) identified fish species that both increased (rock bass, *Ambloplites rupestris*) and decreased (pumpkinseeds, *Lepomis gibbosus*) with increasing exposure. Cvetkovic (2008) linked the observed correlation between species and exposure as an indirect influence of changes in aquatic vegetation composition caused by exposure levels. Therefore, a 2-m drop in water levels for NB5 could result in wholesale changes in the fish community even without any substantial loss of SAV habitat.

Similar to NB5, NB1, and TB are also protected embayments but they are approximately four and six times larger on an areal basis, respectively. As a result, as SAV habitat migrated into open water, area of the new habitat was not sufficiently large to compensate for lost habitat within the protected embayment. In a similar manner, AB and CFR are open embayments that would allow SAV to migrate as water levels declined. Despite this migration, however, the model predicted an overall loss of potential SAV habitat because colonization could only occur in a relatively smaller area.

The variable influence of geomorphology makes any regional prediction difficult unless wetlands are selected randomly so as to represent a statistically significant subset of all wetlands in eastern and northern Georgian Bay. To illustrate the differential response of potential SAV habitat to declining water levels in different wetland types, we have provided an illustrative set of cases including; small/large protected embayments, open embayments, and fringing habitats (Figure 6). While these four cases do not represent all possibilities and notably exclude estuaries and river systems, in our experience they reflect majority of cases for wetlands in eastern and northern Georgian Bay (J. Midwood Pers. Obs.). We recommend that future studies be initiated to test the theoretical response of the different wetlands proposed in this study.

As with most regionally applicable models, there are some weaknesses in ours. The most prevalent one deals with prediction of the Z_{max} in our rule-based model, SAV_{GB}. Since this model takes the approach of a decision tree with specific thresholds or cut-off values, one single value was applied to all coastal wetlands for Z_{max}. While derived empirically using a recursive partitioning approach, this threshold typically overestimates Z_{max} in wetlands with dystrophic water and, conversely, underestimates Z_{max} of SAV in clear water sites. As one might expect, our light extinction equation better represents the true Z_{max} since light is more quickly attenuated in dystrophic water then it is in clear water (Chambers and Prepas, 1988; Houser, 2006). Despite this limitation, the total area underestimated is quite small since majority of sites in eastern and northern Georgian Bay are dystrophic and SAV have not been observed beyond a depth 8.5 m. When light extinction data are available, SAV_{GB} can easily be modified to incorporate a wetland-specific Z_{max}. Due to time constraints and availability of data, we were unable to validate the models with a randomly selected set of wetlands, and cannot extrapolate our findings to the entire region. It is important that a proper study be initiated to evaluate our findings with a representative set of Georgian Bay wetlands.

In this study, both the $SAV_{GB\text{-}EXT}$ and the SAV_{GB} models had comparable performance when applied across Georgian Bay, but for an individual wetland, it may be preferable to develop a model based on an EXT value calculated for the wetland in question. The advantage of the SAV_{GB} model is that it requires no field sampling, provided that remotely sensed imagery and bathymetry data are available. Since IKONOS satellite imagery and digital aerial photos exist for all of eastern and northern Georgian Bay, the lack of detailed bathymetry (sub-meter resolution) is the major impediment to application of this approach throughout the region. Light Detection and Ranging (LiDAR) technology is the perfect method for creating regional DEMs for the Georgian Bay coast line (Irish and White, 1998; Wozencraft and Millar, 2005), and this should be a top priority for environmental agencies that need to identify impacts of climate change on availability of fish habitat in the Great Lakes.

Conclusion

Prior to this study, methods were not available to map the submerged component of coastal wetland habitat in Georgian Bay. We have presented an approach that can now be used to map and predict how SAV habitat will change under changing water level conditions. Despite limitations mentioned previously, our light extinction equation can be used to provide a simple estimate of SAV distribution on a wetland-specific basis. Where data are available, SAV_{GB} can be used to map and model SAV distributions. When SAV_{GB} was applied to six wetlands, the response was quite variable and largely driven by the geomorphology of the wetland. We recommend that future studies apply SAV_{GB} to a group of randomly selected wetlands over a large spatial scale to determine the impact of low water levels on available SAV habitat.

Acknowledgements

Funding for this project came from Georgian Bay Forever and the Ontario Ministry of Natural Resources through the Canada Ontario Agreement. J. Midwood received funding from the Ontario Graduate Scholarship. We are indebted to numerous summer students and other graduate students in the Chow-Fraser lab who helped to collect field samples. We also greatly appreciate the work of both Maja Cvetkovic and John Paul Leblanc who helped to edit earlier drafts of this manuscript.

References

Albert, D.A., Wilcox, D.A., Ingram, J.W., Thompson, T.A., 2005. Hydrogeomorphic classification for Great Lakes coastal wetlands. J. Great Lakes Res., 31, 129–146.

Angel, J.R., Kunkel, K.E., 2010. The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. J. Great Lakes Res., 36, 51–58.

Barko, J., 1991. Sediment interactions with submersed macrophyte growth and community dynamics. Aquatic Botany, 41, 41–65.

Barko, J. and Smart, R. (1983) Effects of organic matter additions to sediment on the growth of aquatic plants. The Journal of Ecology, 71, 161–175.

Barko, J., Smart, R., 1986. Sediment-related mechanisms of growth limitation in submersed macrophytes. Ecology, 67, 1328–1340.

Beyer, H.L., 2004. Hawth's Analysis Tools for ArcGIS. http://www.spatialecology.com/htools. Accessed January 10th, 2011.

Bini, L., Thomaz, S., Murphy, K., Camargo, A., 1999. Aquatic macrophyte distribution in relation to water and sediment conditions in the Itaipu Reservoir, Brazil. Hydrobiologia, 415, 147–154.

Burton, T., Stricker, C., Uzarski, D., 2002. Effects of plant community composition and exposure to wave action on invertebrate habitat use of Lake Huron coastal wetlands. Lakes and Reservoirs Res. Manage., 7, 255–269.

Canfield, D., Langeland, K., Linda, S., Haller, W., 1985. Relations between water transparency and maximum depth of macrophyte colonization in lakes. J. Aquat. Plant Manage., 23, 25–28.

Capers, R., Les, D., 2005. Plant community structure in a freshwater tidal wetland. Rhodora, 107, 386–407.

Chambers, P., Kalff, J., 1985. Depth distribution and biomass of submersed aquatic macrophyte communities in relation to secchi depth. Can. J. Fish. Aquat. Sci., 42, 701–709.

Chambers, P., 1987. Nearshore occurrence of submersed aquatic macrophytes in relation to wave action. Can. J. Fish. Aquat. Sci., 44, 1666–1669.

Chambers, P., Prepas, E., 1988. Underwater spectral attenuation and its effect on the maximum depth of angiosperm colonization. Can. J. Fish. Aquat. Sci., 45, 1010–1017.

Chambers, P., Prepas, E., Hamilton, H., Bothwell, M., 1991. Current velocity and its effect on aquatic macrophytes in flowing waters. Ecol. Appl., 1, 249–257.

Cho, H., Poirrier, M. 2005. A model to estimate potential submersed aquatic vegetation habitat based on studies in Lake Pontchartrain, Louisiana. Restor. Ecol., 13, 623–629.

Chow-Fraser, P., 1998. A conceptual ecological model to aid restoration of Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario, Canada. Wetlands Ecol. Manage., 6, 43–57.

Chow-Fraser, P., 2006 Development of the Water Quality Index (WQI) to Assess Effects of Basin-wide Land-use Alteration on Coastal Marshes of the Laurentian Great Lakes. In: Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Authorhouse, Bloomington, IN, pp 137–166.

Cohen, J., 1960. A Coefficient of Agreement for Nominal Scales. Educ. Psychol. Meas., 20, 37–46.

Congalton, R., 1991. A review of assessing the accuracy of classifications of remotely sensed data. Remote Sens. Environ., 37, 35–46.

Croft, M., Chow-Fraser, P., 2007. Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of Great Lakes coastal marshes. J. Great Lakes Res., 33(SP3), 172-197.

Crosbie, B., Chow-Fraser, P., 1999. Percentage land use in the watershed determines the water and sediment quality of 22 marshes in the Great Lakes basin. Can. J. Fish. Aquat. Sci., 56, 1781–1791.

Cvetkovic, M., 2008. Factors affecting macrophyte and fish distribution in coastal wetlands of Georgian Bay. Master's Thesis, McMaster University.

Cvetkovic, M., Chow-Fraser, P., 2011. Use of ecological indicators to assess the quality of Great Lakes coastal wetlands. Ecol. Indic., 11, 1609-1622.

Dale, H., 1986. Temperature and light: the determining factors in maximum depth distribution of aquatic macrophytes in Ontario, Canada. Hydrobiologia, 133, 73–77.

DeCatanzaro, R., Chow-Fraser, P., 2011. Effects of landscape variables and season on reference water chemistry of coastal marshes of eastern Georgian Bay. Can. J. Fish. Aquat. Sci., 68, 1009-1023.

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Kollar, S., Bergstrom, P.W., Batiuk, R.A., 1993. Assessing water quality with submersed aquatic vegetation. BioScience, 43, 86–94.

Duarte, C., Kalff, J., 1987. Latitudinal influences on the depth of maximum colonization and maximum biomass of submerged angiosperms in lakes. Can. J. Fish. Aquat. Sci., 44, 1759–1764.

Gafny, S., Gasith, A., 1999. Spatially and temporally sporadic appearance of macrophytes in the littoral zone of Lake Kinneret, Israel: taking advantage of a window of opportunity. Aquat. Bot., 62, 249–267.

Gathman, J., Albert, D., Burton, T., 2005. Rapid plant community response to a water level peak in northern Lake Huron coastal wetlands. J. Great Lakes Res., 31, 160–170.

Grace, J.B., Pugesek, B.H., 1997. A structural equation model of plant species richness and its application to a coastal wetland. Am. Nat., 149, 436–460.

Havens, K., Harwell, M., Brady, M., Sharfstein, B., East, T., Rodusky, A., Anson, D., Maki, R., 2002. Large-scale mapping and predictive modeling of submerged aquatic vegetation in a shallow eutrophic lake. The Scientific World Journal, 2, 949–965.

Hothorn, T., Hornik, K., Zeileis, A., 2006. Unbiased recursive partitioning. J. Comput. Graph Stat., 15, 651–674.

Houser, J., 2006. Water color affects the stratification, surface temperature, heat content, and mean epilimnetic irradiance of small lakes. Can. J. Fish. Aquat. Sci., 63, 2447–2455.

Hudon, C., Lalonde, S., Gagnon, P., 2000. Ranking the effects of site exposure, plant growth form, water depth, and transparency on aquatic plant biomass. Can. J. Fish. Aquat. Sci., 57, 31–42.

Irish, J., White, T., 1998. Coastal engineering applications of high-resolution lidar bathymetry. Coastal Eng., 35, 47–71.

Istvánovics, V., Honti, M., Kovács, Á., Osztoics, A., 2008. Distribution of submerged macrophytes along environmental gradients in large, shallow Lake Balaton (Hungary). Aquat. Bot., 88, 317–330.

Jin, K., Ji, Z., James, R., 2007. Three-dimensional water quality and SAV modeling of a large shallow lake. J. Great Lakes Res., 33, 28–45.

Jude, D., Pappas, J., 1992. Fish utilization of Great Lakes coastal wetlands. J. Great Lakes Res., 18, 651–672.

Kautsky, L., 1988. Life strategies of aquatic soft bottom macrophytes. Oikos, 53, 126–135.

Keddy, P., 1982. Quantifying within-lake gradients of wave energy: interrelationships of wave energy, substrate particle size and shoreline plants in Axe Lake, Ontario. Aquat. Bot., 14, 41–58.

Keddy, P., 1983. Shoreline vegetation in Axe Lake, Ontario: effects of exposure on zonation patterns. Ecology, 64, 331–344.

Keddy, P., 1985. Wave disturbance on lakeshore and within-lake distribution of Ontario's Atlantic Coastal plain flora. Can. J. Bot., 63, 656–660.

Keddy, P., 2000. Wetland Ecology: Principles and Conservation. Cambridge University Press, Cambridge UK.

Keddy, P., Reznicek, A., 1986. Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. J. Great Lakes Res., 12, 25–36.

Lacoul, P., Freedman, B., 2006. Environmental influences on aquatic plants in freshwater ecosystems. Environ. Rev., 14, 89–136.

Lillesand, T.M., Kiefer, R.W., Chipman, J.W., 2004. Remote Sensing and Image Interpretation, 5th edn. Wiley, Hoboken, NJ.

Madsen, J., Chambers, P., James, W., Koch, E., Westlake, D., 2001. The interaction between water movement, sediment dynamics and submersed macrophytes. Hydrobiologia, 444, 71–84.

Magnuson, J., Webster, K., Assel, R., Bowser, C., Dillon, P., Eaton, J., Evans, H., Fee, E., Hall, R., Mortsch, L., 1997. Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian shield region. Hydrol. Processes, 11, 825–871.

McNair, S., Chow-Fraser, P., 2003. Change in biomass of benthic and planktonic algae along a disturbance gradient for 24 Great Lakes coastal wetlands. Can. J. Fish. Aquat. Sci., 60, 676–689.

Middelboe, A., Markager, S., 1997. Depth limits and minimum light requirements of freshwater macrophytes. Freshwater Biol., 37:553–568.

Midwood, J.D., Chow-Fraser, P., 2010. Mapping Floating and Emergent Aquatic Vegetation in Coastal Wetlands of Eastern Georgian Bay, Lake Huron, Canada. Wetlands*,* 3, 1–12.

Midwood, J.D., Chow-Fraser, P., 2012. Changes in aquatic vegetation and fish communities following 5 years of sustained low water levels in coastal marshes of eastern Georgian Bay, Lake Huron. Glob. Change Biol*.,* 18, 93-105.

Midwood, J.D., Rokitnicki-Wojcik, D. & Chow-Fraser, P. 2012. Development of an inventory of coastal wetlands for eastern Georgian Bay, Lake Huron. ISRN Ecology vol. 2012, Article ID 950173, 13 pages. doi:10.5402/2012/950173.

Mortsch, L., Quinn, F., 1996. Climate change scenarios for Great Lakes Basin ecosystem studies. Limnol. Oceanogr., 41, 903–911.

Murphey, P., Fonseca, M., 1995. Role of high and low energy seagrass beds as nursery areas for *Penaeus duorarum* in North Carolina. Mar. Ecol. Prog. Ser., 121, 91–98.

Oliver, M.A., Webster, R., 1990. Kriging: a method of interpolation for geographical information systems. Int. J. Geogr. Inf., 4, 313-332.

Orth, R., Moore, K.A., 1984. Distribution and abundance of submerged aquatic vegetation in Chesapeake Bay: an historical perspective. Estuaries*,* 7, 531-540.

Quinlan, C., Mulamoottil, G., 1987. The effects of water level fluctuations on three Lake Ontario shoreline marshes. Can. Water Res. J., 12, 64–77.

Randall, R., Minns, C., Cairns, V., Moore, J., 1996. The relationship between an index of fish production and submerged macrophytes and other habitat features at three littoral areas in the Great Lakes. Can. J. Fish. Aquat. Sci., 53, 35–44.

Riis, T., Biggs, B., 2001. Distribution of macrophytes in New Zealand streams and lakes in relation to disturbance frequency and resource supply—a synthesis and conceptual model. New Zeal. J. Mar. Fresh., 35, 255–267.

Riis, T., Hawes, I., 2003. Effect of wave exposure on vegetation abundance, richness and depth distribution of shallow water plants in a New Zealand lake. Freshwater Biol., 48, 75–87.

Sand-Jensen, K., Madsen, T., 1991. Minimum light requirements of submerged freshwater macrophytes in laboratory growth experiments. J. Ecol., 79, 749–764.

Schwarz, A., Howard-Williams, C., Clayton, J., 2000. Analysis of relationships between maximum depth limits of aquatic plants and underwater light in 63 New Zealand lakes. New Zeal. J. Mar. Fresh., 34, 157–174.

Seilheimer, T., Chow-Fraser, P., 2007. Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. J. Great Lakes Res., 33, 154–171.

Sellinger, C., Stow, C., Lamon, E., Qian, S., 2008. Recent Water Level Declines in the Lake Michigan− Huron System. Environ. Sci. Technol., 42, 367–373.

Squires, M., Lesack, L., Huebert, D., 2002. The influence of water transparency on the distribution and abundance of macrophytes among lakes of the Mackenzie Delta, Western Canadian Arctic. Freshwater Biol., 47, 2123–2135.

Stewart, C., Freedman, B., 1989. Comparison of the macrophyte communities of a clearwater and a brownwater oligotrophic lake in Kejimkujik National Park, Nova Scotia. Water Air Soil Poll., 46, 335–341.

Wei, A., 2008. Forecasting the response of coastal wetlands to declining water levels and environmental disturbances in the Great Lakes. PhD Thesis, McMaster University.

Wei, A. and Chow-Fraser, P. (2005) Untangling the confounding effects of urbanization and high water level on the cover of emergent vegetation in Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario. Hydrobiologia, 544:1–9.

Wei, A., Chow-Fraser, P., 2007. Use of IKONOS imagery to map coastal wetlands of Georgian Bay. Fisheries, 32, 167-173.

Wei, A., Chow-Fraser, P., 2008. Testing the transferability of a marsh-inundation model across two landscapes. Hydrobiologia, 600, 41–47.

Wetzel, R., 2001. Limnology - Lake and River Ecosystems. Academic Press, San Diego, California.

Wozencraft, J., Millar, D., 2005. Airborne lidar and integrated technologies for coastal mapping and nautical charting. Mar. Technol. Soc. J., 39, 27–35.

Xie, Y., An, S., Wu, B., 2005. Resource allocation in the submerged plant *Vallisneria natans* related to sediment type, rather than water-column nutrients. Freshwater Biol., 40, 391–402.

Zhu, B., Fitzgerald, D., Hoskins, S., Rudstam, L., Mayer, C., Mills, E., 2007. Quantification of historical changes of submerged aquatic vegetation cover in two bays of Lake Ontario with three complementary methods. J. Great Lakes Res., 33, 122–135

Table 1. Summary of light characteristics at sites associated with data used to calculate light extinction coefficients (EXT) and maximum depth of colonization (Z_{max}; m). Water Colour is in Pt units. TURB = Turbidity. Also provided are: (a) Observed Z_{max} values for each wetland as well as those predicted from (b) Hudon et al.'s (2000) equation and (c) the Georgian Bay-specific equation. Differences (m) between observed and that predicted by Hudon's equation (d) and by the Georgian Bay-specific equation (e) are also presented.

Wetland	Latitude	Longitude	EXT	Water Colour	TURB	(a) Z_{max}	(b) Predicted Z_{max}	(c) Predicted Z_{max}	(d)	(e)
Alexander Bay *	45.05715	-80.00856	0.26	23.3	0.34	8.48	5.56	8.09	2.93	0.39
Coffin Rock	45.04711	-79.98246	0.62	41.7	0.22	4.54	4.46	5.33	0.08	-0.79
North Bay 1	44.89437	-79.79425	0.85	80.3	1.23	5.45	4.20	4.78	1.26	0.68
North Bay 5	44.88156	-79.80388	0.82	35.7	0.19	5.45	4.22	4.83	1.23	0.63
North Go Home *	45.00593	-79.96083	0.42	18.3	0.62	5.45	4.87	6.28	0.58	-0.82
Red Sand Beach *	46.03148	-81.65944	0.25	16.0	0.06	8.48	5.65	8.36	2.83	0.12
Roseborough	44.99698	-79.92521	0.97	55.4	0.19	4.24	4.10	4.60	0.14	-0.36
Scow Bay *	46.07687	-81.61002	0.65	22.3	0.22	5.15	4.42	5.25	0.73	-0.09
Water Fall Bay	45.55872	-80.34222	1.26	—	—	4.54	3.93	4.30	0.61	0.24

* Clear Water Sites

Table 2. Wetland location (Decimal Degrees) and number of samples collected in 2008 and 2009. The number of samples from each wetland that were used for the Development of the Rule-Based model and the number used to test the model are also shown.

Wetland	Wetland Code	Latitude	Longitude	Samples Collected 2008	Samples Collected 2009	Total Samples	Model Development	Model Testing
Alexander Bay	AB	45.05366	-80.00375	10	40	50	33	17
Black Rock Bay	BRB	45.04772	-79.97418	2	—	2	2	0
Coffin Rock	CFR	45.04728	-79.98523	10	25	35	24	11
Duncanson's Bay	DBB	45.01929	-79.98394	2	—	2	1	1
Ganyon Bay	GB	44.92254	-79.82300	—	5	5	3	2
Miner's Creek Bay	MNC	45.06074	-79.95384	2	—	2	2	0
North Bay 1	NB1	44.89967	-79.79426	12	33	45	27	18
North Bay 5	NB5	44.88158	-79.80312	2	5	7	4	3
North Go Home	NGH	45.00638	-79.96106	2	—	2	1	1
Red Sand Beach	RSB	46.03069	-81.65873	5	—	5	2	3
Roberts Bay	RB	44.85511	-79.83085	—	5	5	3	2
Roseborough Bay	RS	44.99659	-79.92490	6	—	6	6	0
Sand Bay	SB	45.93209	-80.91606	—	7	7	6	1
Sawdust Bay	SAW	46.00114	-81.21019	6	—	6	4	2
Scow Bay	SCB	46.07717	-81.60730	7	—	7	5	2
Silver Island	SI	45.95868	-81.29566	6	—	6	5	1
Sturgeon Bay	STB	46.05902	-81.85726	—	1	1	0	1
Treasure Bay	TB	44.87119	-79.85882	10	15	25	18	7
			Total	82	136	218	146	72

Table 3. Average depth and sediment composition information for all 82 samples collected and processed. Sand is defined as particles greater than 50 µm diameter. Clay particles were smaller than 2 µm and silt ranged from >2 µm to <50 µm.

Sediment Property	Mean	Median	Standard Deviation	Upper 95%	Lower 95%
Proportion Sand (%)	36.5	17.9	±32.6	43.6	29.3
Proportion Clay (%)	22.2	15.9	±17.7	26.1	18.3
Proportion Silt (%)	41.1	39.6	±21.5	45.8	36.4
Proportion Organic (%)	8.1	5.5	±7.9	9.8	6.3
Depth (cm)	285	268	±206	330	240
Total Phosphorus (mg/g)	0.42	0.43	±0.14	0.45	0.39
Total Ammonia (mg/g)	0.039	0.022	±0.048	0.049	0.028

Table 4. Detailed description of each node in the rule-based submerged aquatic vegetation model.

Node Number	Description
1	The first node in the model removed areas that were considered too deep for SAV colonization (Depth ≥ 4.88 m). SAV limitation at these depths is driven by availability of light and areas shallower than 4.88 m are further refined at node 2.
2	At this node, shallow water areas are separated from deeper regions of SAV habitat. Deeper areas are considered to be regions with depths greater than or equal to 0.82 m and progress to node 3. Areas with depths less than 0.82 m represent the shallow water zone and are further differentiated at node 5.
3	For this model, potential SAV habitat is considered to exist between depths of 0.82 m and 4.10 m. Depths ranging from 4.10 m to 4.88 m are further differentiated at node 4.
4	For this node, exposure plays a role to help separate limits to Z_{max}. Areas with higher REI values (REI>=79.9) have SAV extending out a depth of 4.88 m compared with areas with lower REI values, which have SAV out to a depth of only 4.10 m. In our experience, water clarity in areas with high exposure or REI are typically more influenced by water from Georgian Bay, which tends to be clear. By comparison, sites that are more influenced by their watershed with less flushing from Georgian Bay water typically are more dystrophic and therefore have a lower value for Z_{max}.
5	The remaining nodes deal solely with shallow water parts of the wetlands (Depth < 0.82 m). At node 5, the combined influence of REI/D is used to isolate areas that have low levels of exposure (REI/D >= 2.86) but still occur in shallow water. Majority of the remaining sites did not have SAV.
6	Once again using the REI/D ratio, sites with intermediate REI/D values were separated and considered to not contain SAV. When the REI/D ratio was equal to or greater than 50.52, we proceed to node 7.

7	In shallow water sites with high REI/D ratios, SAV were considered present when REI values were low. This was typically linked with sites that occurred in shallow water with low exposure or deep water with proportionally higher REI. Conversely, SAV were not present at sites with high REI/D values where the REI value was also greater than or equal to 43.2.

Table 5. Contingency table of the predicted versus observed presence (1) and absence (0) of submerged aquatic vegetation using the Rule-Based model and an independent set of 71 sample points. The overall accuracy of the model as well as the Kappa statistic are also provided.

		Observed	
		0	1
Predicted	0	18	4
	1	8	42
	Kappa	0.626	
	Overall Accuracy	83.3	

Table 6. Potential submerged aquatic vegetation habitat (ha) for each wetland at a given water level elevation. Potential SAV habitat declined in Coffin Rock, North Bay 1, Alexander Bay, and Treasure Bay as the water level also declined. North Bay 5 and Red Sand Beach had variable responses.

	Potential SAV Habitat (ha)					
Elevation (m, asl)	**Coffin Rock**	**North Bay 1**	**North Bay 5**	**Alexander Bay**	**Red Sand Beach**	**Treasure Bay**
176.4	8.98	8.93	2.09	7.60	5.42	15.83
176.3	8.91	8.45	1.98	7.15	5.29	15.48
176.2	8.67	7.93	1.82	6.86	5.21	14.71
176.1	8.21	7.30	1.72	6.44	4.87	13.63
176.0	7.90	6.68	1.72	6.09	4.95	12.40
175.9	7.64	5.89	1.67	5.93	5.07	11.70
175.8	7.60	5.52	1.66	5.69	5.16	11.13
175.7	7.49	5.34	1.64	5.61	5.30	10.60
175.6	7.23	5.04	1.62	5.24	5.44	10.09
175.5	6.93	4.93	1.57	4.91	5.58	9.58
175.4	6.51	4.94	1.51	4.65	5.72	8.96
175.3	6.21	4.95	1.48	4.28	5.85	8.57
175.2	5.94	4.73	1.52	3.99	5.99	8.22
175.1	5.57	4.18	1.57	3.73	6.12	7.78
175.0	5.31	3.57	1.62	3.51	6.27	7.22
174.9	5.12	3.22	1.68	3.25	6.39	6.00
174.8	5.02	2.88	1.73	3.13	6.53	5.55
174.7	4.96	2.83	1.76	3.00	6.61	5.15
174.6	4.98	2.76	1.81	2.95	6.66	4.54
174.5	5.15	2.65	1.88	2.86	6.69	4.08
174.4	5.50	2.54	2.00	2.78	6.61	3.69

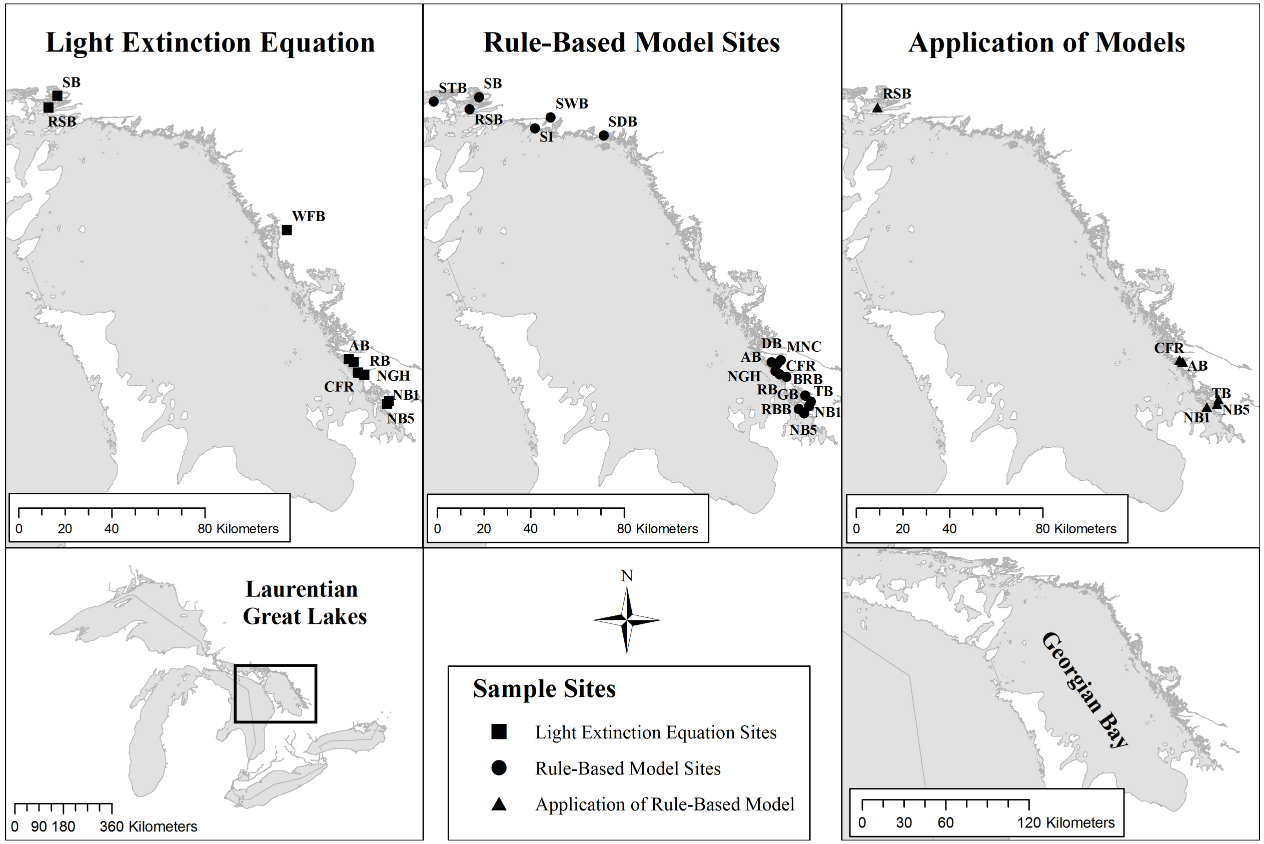


Figure 1. Location of wetlands used in the development of the light extinction equation, rule-based model, and the application of the rule based model. Wetlands were sampled in 2008 and 2009.

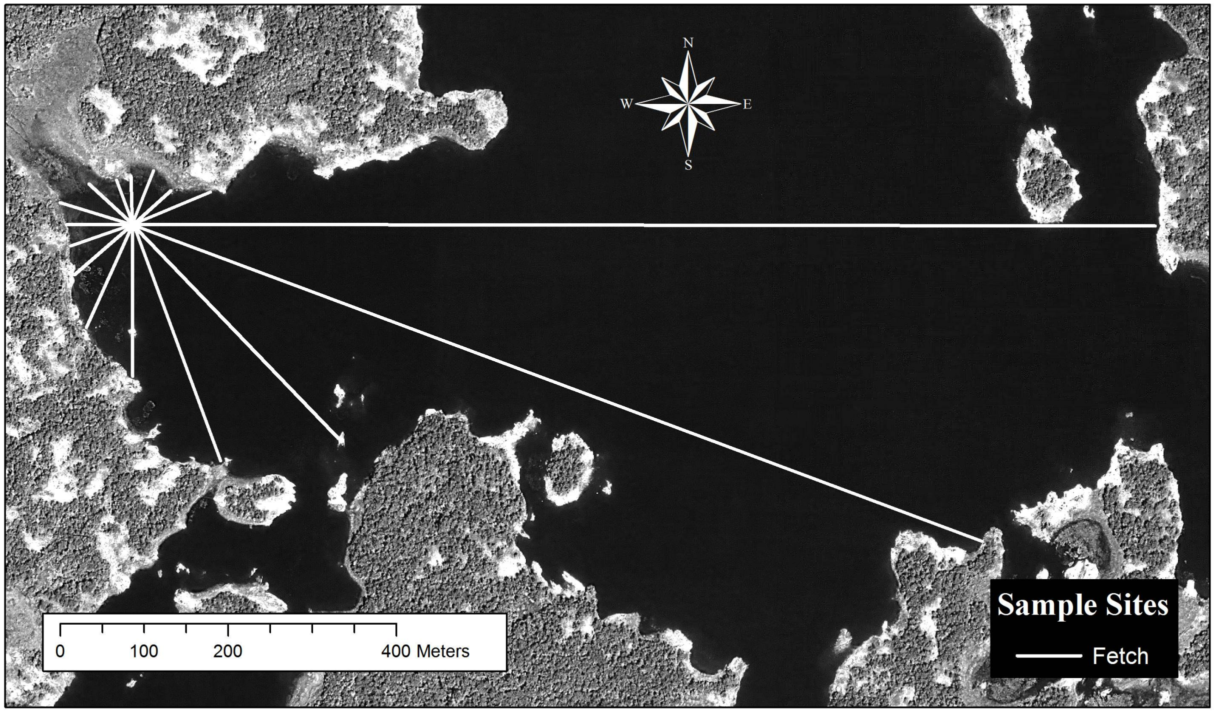


Figure 2. Example calculation of the modified Relative Exposure Index (REI) for a sample point in Coffin Rock, Tadenac Bay. Fetch (white lines) was measured in each of the four cardinal directions as well as 12 equally spaced intervals. Fetch estimates the distance the wind travels in a given direction before reaching the sample point. This value, when combined with the percentage of time the wind blows in a given direction and the average wind speed was used to calculate the REI, an estimate of disturbance from wind and wave energy.

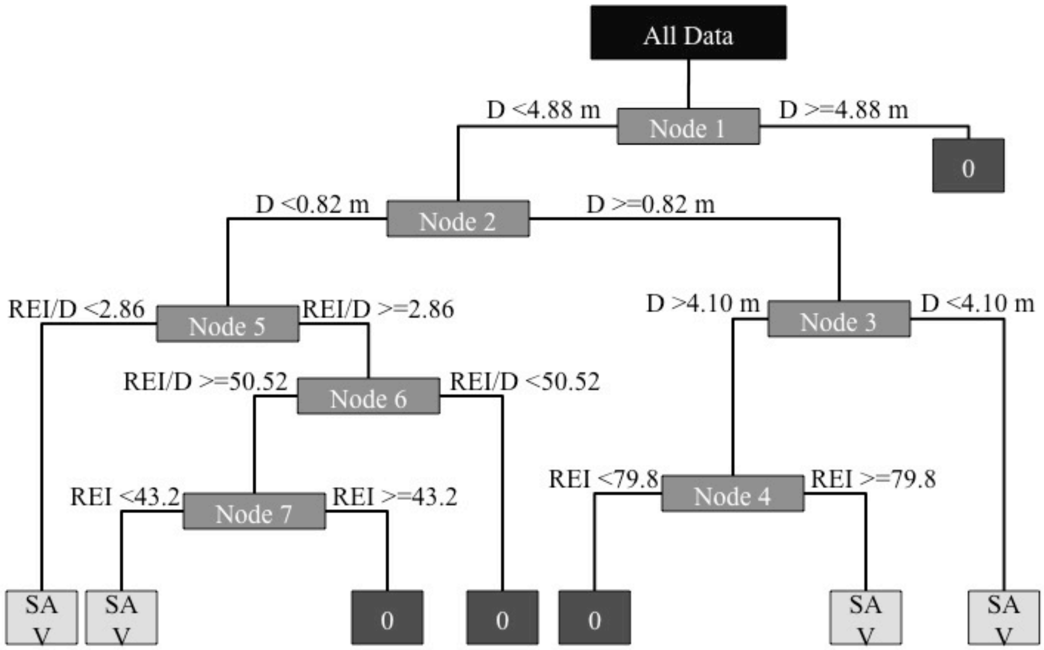


Figure 3. Schematic of the rule-based model (SAV_{GB}). A detailed description of the threshold at each node is found in Table 3.

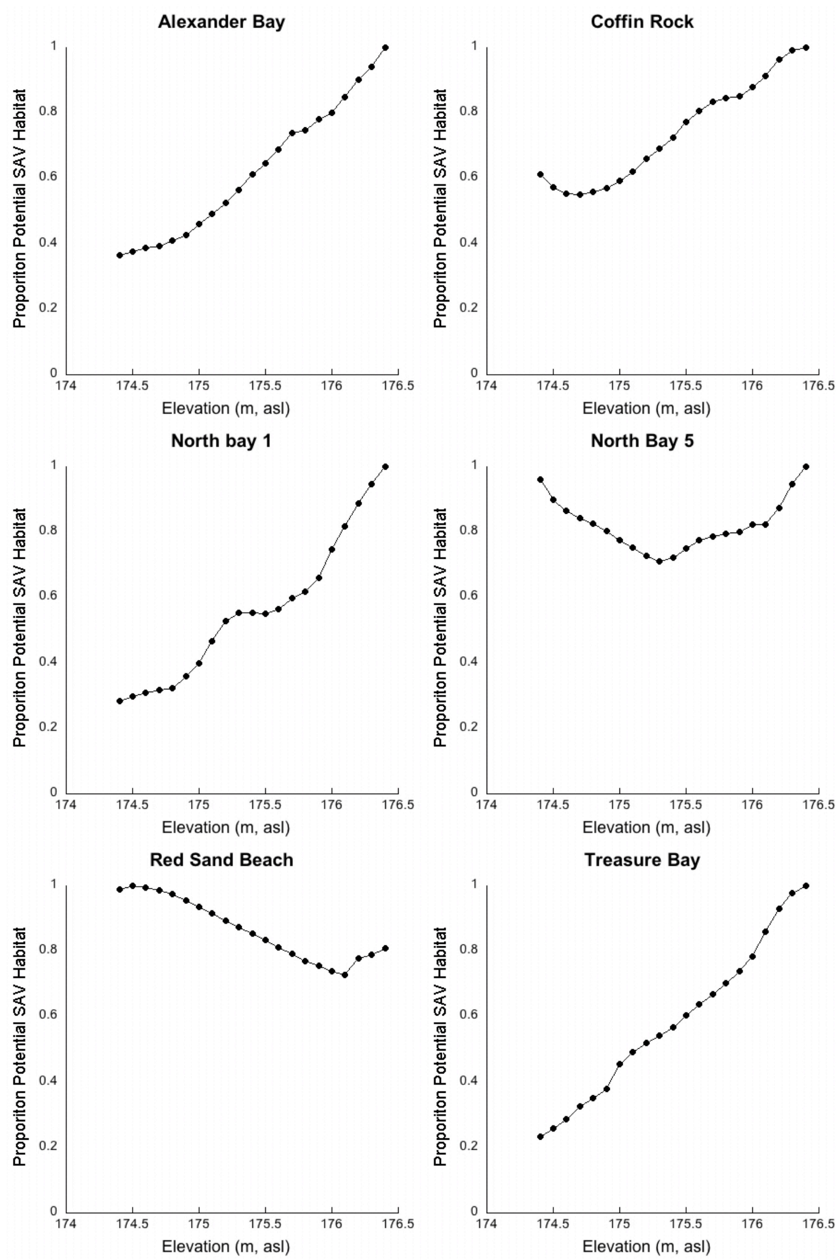


Figure 4. Results for the application of SAVGB to six wetlands. Four of the six wetlands show a decline in SAV with lower water levels (AB, CFR, NB1, and TB). NB5 shows no long-term changes and RSB shows an increase in potential SAV habitat.

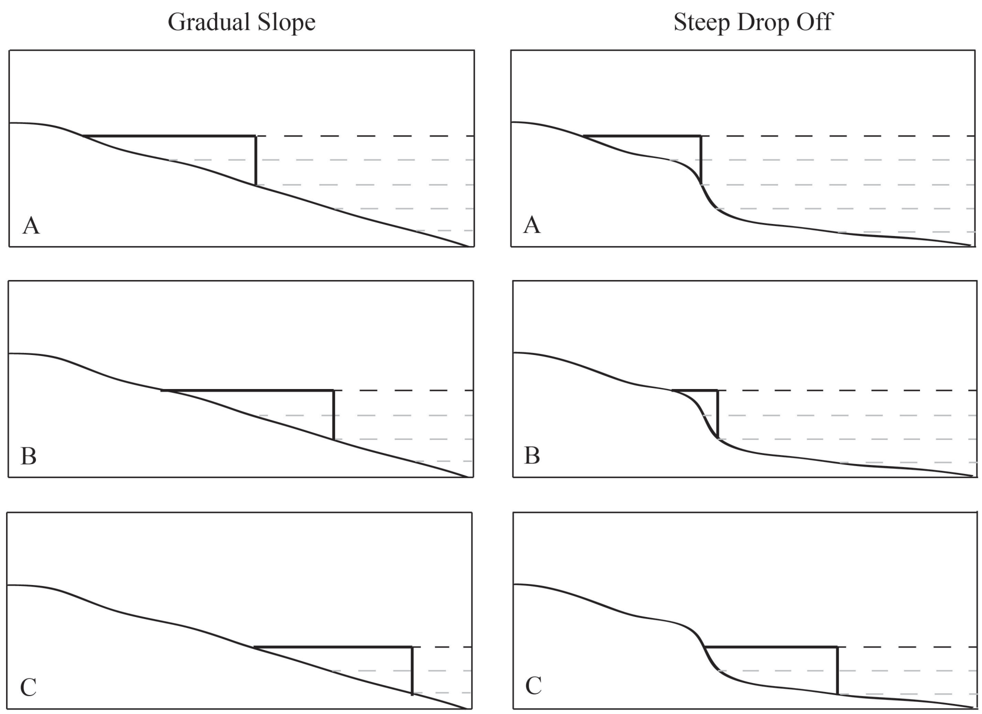


Figure 5. Conceptual response of potential SAV habitat under progressively lower water levels (A-C), for two different wetland geomorphologies. In the "Gradual Slope" situation, potential SAV habitat changes little as it migrates into the bay. In the "Steep Drop Off" situation (reminiscent of RSB in this study), potential SAV habitat reaches a minimum in case B as it transitions through the steep zone and is actually maximized under the lowest water level scenario (case C).

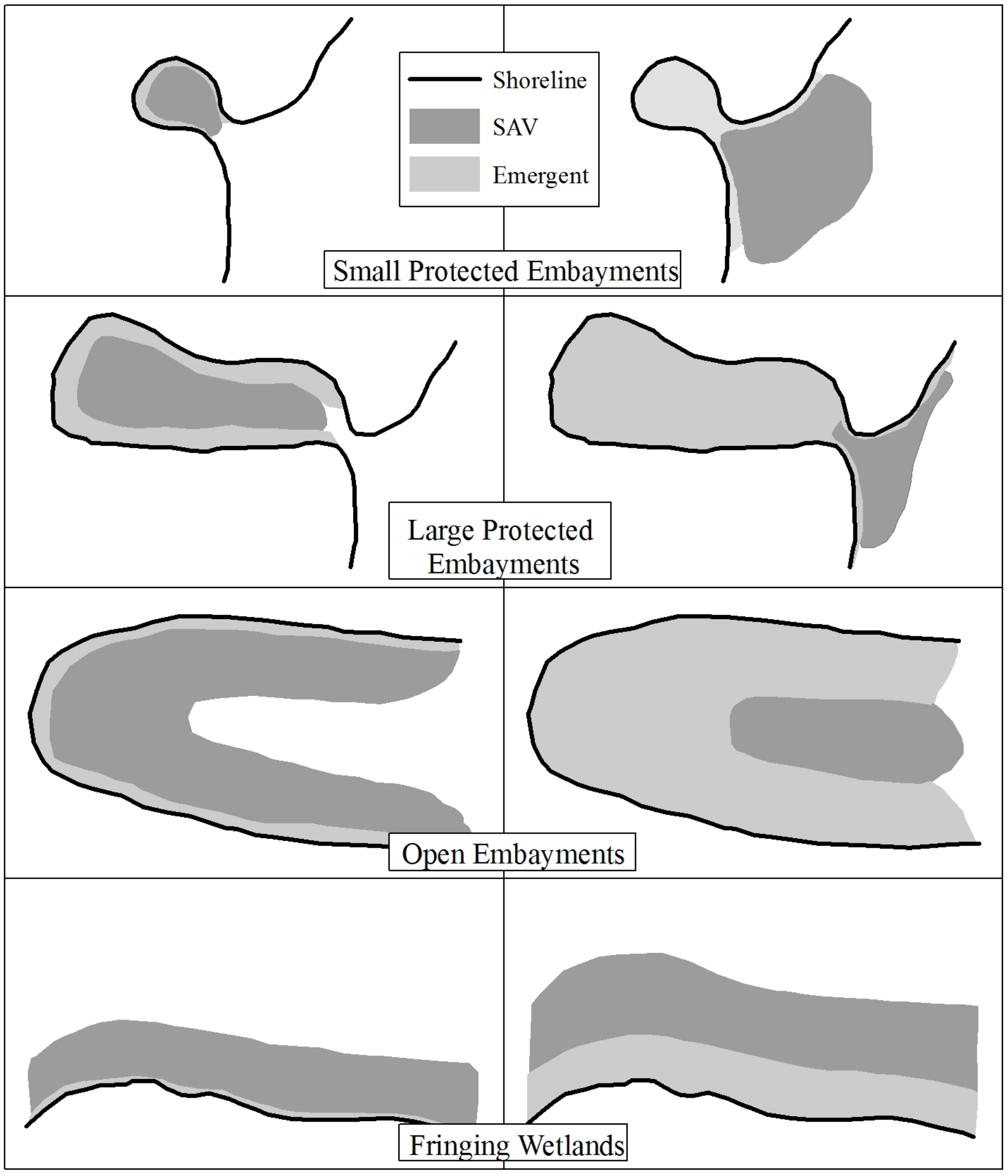


Figure 6. Conceptual response of potential SAV habitat under low water levels for four different types of coastal wetland environment. Images on the left show SAV distribution at an initial time period. Images on the right show the response of SAV in the same wetland following a decline in water levels.

Chapter 4:

Changes in aquatic vegetation and fish communities following 5 years of sustained low water levels in coastal marshes of eastern Georgian Bay, Lake Huron.

Reprinted with permission from:

Midwood JD and Chow-Fraser P (2012) Global Change Biology 18:93-105.

Abstract

Aquatic vegetation in the relatively pristine coastal wetlands of eastern Georgian Bay provides critical habitat for a diverse fish community. Declining water levels in Lake Huron over the past decade, however, have altered the wetland plant assemblages in favour of terrestrial (emergent and meadow) taxa and have thus reduced or eliminated this important ecosystem service. In this study, we compared IKONOS satellite images for two regions of eastern Georgian Bay (acquired in 2002 and 2008) to determine significant changes in cover of 4 distinct wetland vegetation groups (meadow [M], emergent [E], high-density floating [HD] and low-density floating [LD]) over the six years. While LD decreased significantly (mean -2995.4 m^2), M and HD increased significantly (mean +2020.9 m^2 and +2312.6 m^2, respectively) between 2002 and 2008. Small patches of LD had been replaced by larger patches of HD. These results show that sustained low water levels have led to an increasingly homogeneous habitat and an overall net loss of fish habitat. A comparison of the fish communities sampled between 2003 and 2005 with those sampled in 2009 revealed that there was a significant decline in species richness. The remaining fish communities were also more homogeneous. We suggest that the observed changes in the wetland plant community due to prolonged low water-levels may have resulted in significant changes in the fish communities of coastal wetlands in eastern Georgian Bay.

Keywords: change detection, coastal wetlands, Great Lakes, remote sensing, water-level regulation

Introduction

Global climate change is expected to greatly alter the hydrological cycle on a world-wide basis, resulting in drought, extreme precipitation events, and increases in sea level (Karl & Trenberth 2003; Trenberth *et al.* 2003). Predictions for large inland lakes, such as the Laurentian Great Lakes, have been highly variable, but majority point to an overall decline in lake levels for all five lakes, with much greater extremes than those experienced over the past century (Mortsch & Quinn 1996; Magnuson *et al*. 1997; Angel & Kunkel 2010). These will be the result of predicted warmer winters, seasonal changes in precipitation, increased evaporation and water-surface temperatures, decreased ice cover, and earlier spring snowmelt (Lenters 2001; Quinn 2002; Sellinger *et al.* 2008; Hanrahan *et al*. 2010). These modifications in hydrology will have far-reaching effects on the structure and function of coastal ecosystems, including a change in habitat ranges that may negatively impact artisanal, commercial and recreational fisheries, and allow for the introduction of invasive species (Ross *et al.* 2001; Ficke *et al.* 2007).

Unlike smaller inland lakes, water levels in the Great Lakes fluctuate naturally both seasonally and annually, and in multi-year cycles (Lyon *et al.* 1986;

Lenters 2001; Quinn 2002; Sellinger *et al.* 2008; Hanrahan *et al*. 2010). Such fluctuations govern the type of aquatic plant communities in coastal marshes that occur along the margins of these large lakes (Keddy & Reznicki 1986; Quinlan & Mulamoottil 1987; Grosshans *et al*. 2004; Hudon 2004; Gathman *et al.* 2005; Wei & Chow-Fraser 2005). Plants in these wetlands have a range of tolerance to depth and duration of inundation that allow them to dominate under different water-level scenarios (Gathman *et al*. 2005). During periods of high lake levels, submerged vegetation typically dominate, whereas at low water levels, meadow species dominate (Burton 1985; Hudon 1997; Chow-Fraser *et al*. 1998; Mortsch *et al*. 2008; Wilcox & Nichols 2008). This relationship is, however, complicated by the observed time lag between water level and vegetation type such that the distribution observed at any given time is determined by water levels experienced two to five years earlier (Quinlan & Mulamoottill 1987).

Water levels in the Laurentian Great Lakes have a long history of human-induced regulation, which has disturbed the natural cycles of high and low water levels (Quinn, 2002). It is known that poor habitat conditions exist at extremely high (Gathman *et al*. 2005) or low water levels (Quinlan & Mulamoottil 1987), but the exact effects of a disruption in natural water cycles on coastal systems is not well studied. It is clear, however, that fluctuations are essential for maintaining healthy and functional coastal marshes because they prevent dominance by one type of vegetation community (Wilcox & Meeker 1991; Wilcox 2004; Gathman *et al*. 2005).

Of the five Great Lakes, Lake Michigan–Huron is expected to undergo the greatest change in water levels, decreasing by as much as 2.5 m below base case (Mortsch & Quinn 1996; Magnuson *et al*. 1997). A drop of such a magnitude should have profound impacts on the plant communities of coastal marshes, but it is the loss of periodicity in the cycle of highs and lows that may be of a greater concern to ecologists. Early evidence of such a loss was documented by Sellinger *et al.* (2008), who showed that water levels have remained near record low levels since 1999, which has resulted in a period of continuous drawdown for almost 10 years, compared with a maximum period of continuous low levels of five years during the past century. Such a period of sustained low water levels may drastically alter the distribution of aquatic plants and lead to a more structurally homogeneous plant community.

Coastal wetlands of eastern Georgian Bay, Lake Huron, represent some of the most pristine systems in the Great Lakes (Chow-Fraser 2006; Cvetkovic & Chow-Fraser 2011). Because human-induced disturbance (e.g. agricultural and urban development) is minimal compared to other areas of the Great Lakes, the major threat to these wetlands is prolonged exposure to low water levels such as that experienced over the past decade. These coastal marshes form in small, shallow bays and are naturally oligotrophic due to low nutrient input from the surrounding granite bedrock and their connection to Georgian Bay (DeCatanzaro & Chow-Fraser 2011). Majority of these are still in pristine condition and they support a diverse community of aquatic macrophytes, typically with diverse vertical and horizontal structure (Croft & Chow-Fraser 2007). This is important for the many wetland-dependent fish that use these areas for spawning and nursery habitat (Jude & Pappas 1992; Randall *et al.* 1996; Wei *et al.* 2004; Jude *et al.* 2005).

The ideal fish habitat must necessarily be optimized for both food availability and protection from predators (Savino & Stein 1982; Eadie & Keast 1984; Werner *et al*. 1983; Killgore *et al*. 1989). Many studies have shown a trade-off between dense aquatic vegetation, where fish are protected from predators but where fewer invertebrate prey exist, and the open water, where there is abundant food but where fish are much more vulnerable to predators (e.g. Eadie & Keast 1984; Werner *et al*. 1983; McIvor & Odum 1988). This trade-off results in many species preferentially using areas along the edge of dense vegetation and open water, or areas with intermediate vegetation densities (Höök *et al*. 2001; Jacobus & Webb 2006). A complex landscape with numerous patches of vegetation is therefore ideal as it allows fish to move amongst patches in relative safety.

Structural complexity can be expressed in various ways, from a comparison of stem density and percent coverage of species among sites (Trebitz *et al*. 2009), to determination of patch size within wetlands (Jacobus & Webb 2006), to a statistical measure of habitat variability across a region (Trebitz *et al*. 2009). Jacobus & Webb (2006) found that when average patch size was reduced to <128 m^2, species richness of the fish community fell, rare species began to disappear, and overall, the fish assemblage became less diverse. We predict that the prolonged period of low water levels experienced over the past decade in Lake Huron has reduced the structural complexity of the plant communities in Georgian Bay wetlands, by allowing terrestrial meadow species to displace the emergent and submersed aquatic vegetation (Wei & Chow-Fraser 2005; Leahy *et al.* 2005). We also hypothesize that the alteration in structure and composition of the habitat would lead to a significant reduction in the species richness of the fish communities because high habitat complexity is essential for maintaining high fish diversity (reviewed in Smokorowski & Pratt 2007; Cvetkovic *et al.* 2010).

The large distribution of coastal wetlands in eastern Georgian Bay, coupled with difficulties in accessing many of them, prevents majority of wetlands from being surveyed *in situ*. Satellite imagery provides an alternate survey method because spectral information can be used to identify different plant groups that occur over a very large area (Bartlett & Klemas 1980; Silva *et al.* 2008; Midwood & Chow-Fraser 2010). This approach has been used successfully to monitor changes in land-use (Dewan & Yamaguchi 2009), and to map terrestrial wetlands (Houloulis & Michener 2000) and coastal wetlands (Leahy *et al.* 2005; Baker *et* *al.* 2007). We will examine changes in the habitat complexity by conducting a change-detection analysis of two IKONOS satellite images acquired in 2002 and 2008 for two regions of eastern Georgian Bay. The 6-year difference between acquisitions ensures that the 5-year lag time suggested by Quinlan & Mulamoottil (1987) is taken into consideration. We will determine significant changes in above-surface aquatic wetland vegetation (floating and emergent) and quantify changes in average patch size within wetlands. Our overall goal is to quantify changes in vegetation coverage and structure that have occurred during a period of sustained low water levels and determine how these changes in habitat have influenced the fish community. Understanding wetland-vegetation dynamics is essential for making recommendations on future water-level regulation plans and understanding the potential response of the fish community to forecasted water levels.

Materials & Methods

Study Location

Georgian Bay is a large bay in northeastern Lake Huron. The shoreline of Georgian Bay is one of the longest and most complex in the world, allowing for the formation of thousands of coastal wetlands. On average, these wetlands are 1.4 (± 12.0) ha in size (P. Chow-Fraser, unpublished data). Low levels of human development and watershed alteration have allowed these wetlands to remain in a relatively pristine state with high fish and plant species' richness (Seilheimer & Chow-Fraser 2006, 2007; Croft & Chow-Fraser 2007; Cvetkovic & Chow-Fraser 2011).

Water Levels

Water level data were acquired from the Canadian Hydrographic Services, a Department of Fisheries and Oceans Canada. In order to account for the documented lag time in macrophyte communities, we compared water levels for the five years preceding the acquisition of our images, using only data from the growing seasons (April to September). Therefore, for 2002 imagery, we used mean water levels for the years 1997–2001, and for 2008 imagery, we used mean water levels for the years 2003–2007.

Process Tree Classification Development and Assessment

Midwood and Chow-Fraser (2010) developed a classification scheme for eastern Georgian Bay, called the process tree classification (PTC), that used 2002 IKONOS satellite imagery to map four distinct vegetation classes in wetlands: high-density floating (HD; covering greater than 50% of the surface), low-density floating (LD; covering less than 50% of the surface), emergent (E), and meadow (M) as well as water (W) and rock (R). In this study, we chose two of the 2002 IKONOS satellite images covering the regions of North Bay and Tadenac Bay (collected on July 1st, 2002 at 11:30 am; Figure 1). Images covering these same regions were acquired again on July 16th, 2008 at 11:22 am. For all images, bands were available in the visible (red, green and blue) as well as near-infrared spectra. All images were pre-processed by GeoEye (Dulles, VA, USA) using a proprietary procedure.

PTC2002 was designed specifically for use with 2002 IKONOS images and could not be applied to the 2008 satellite imagery (see methods in Midwood & Chow-Fraser 2010). Instead, the procedure used to create and validate PTC2002 was repeated for the 2008 imagery, and ground truth samples collected concurrently with image acquisition were used to create the classification. This allowed us to quickly create PTC2008 using the structure of PTC2002. As was the case for PTC2002 (Midwood & Chow-Fraser 2010), the minimum overall accuracy considered acceptable for PTC2008 was 85%. To verify the accuracy of PTC2008, ground truth samples for the six ground cover classes were collected in 10 wetlands (5 wetlands in each Tadenac Bay and North Bay) during the summer of 2008. These 10 wetlands were selected because they were included in both the 2002 and 2008 IKONOS imageries (see below) and they had already been ground truthed and classified in the 2002 images with the process tree classification (PTC2002; Midwood & Chow-Fraser 2010). Creation, validation and application of both PTC2002 and PTC2008 were conducted in Definiens Developer 7.0 (Definiens®AG, Munchen, Germany).

Change Detection

For this study we opted to use a post-classification analysis, which involves mapping vegetation in two images separately and then comparing the resulting maps (Coppin *et al*. 2004; Lu *et al*. 2004). The major disadvantage with this method is that the final accuracy of the change detection is the product of the initial classification accuracies and is therefore always lower (Coppin *et al*. 2004; Lu *et al*. 2004). Thus, the overall accuracy (comprising all 6 ground cover classes) of our change detection was calculated as the product between the overall accuracy in 2002 and the overall accuracy in 2008. Individual change-detection accuracies were also calculated for the 6 classes as the product of their individual accuracies in 2002 and 2008

While not always ideal for change detection, post-classification analysis is more easily applied when reference maps are available, and it does not require radiometric calibration of the independent images (Coppin *et al*. 2004; van Oort 2007). This method has been used successfully to assess change in terrestrial environments (Mas 1999), urban areas (Zhou *et al.* 2008; Dewan & Yamaguchi 2009), and wetland cover (MacLeod & Congalton 1998; Zhou *et al*. 2010).

MacLeod & Congalton (1998) identified four steps that are necessary for change detection analysis. First and most broadly, it must be determined if a change has in fact occurred during the dates of image acquisition. Next, the nature of the change should be determined so that specific classes can be identified and monitored during the analysis. Following class identification, changes in areal coverage should be identified. Finally, changes to spatial patterns of surface features should be determined. The assessment of areal change in these Georgian Bay wetlands will provide important information on how much vegetation is changing, and provide insight into the fish community.

To quantify changes that have occurred between 2002 and 2008, we selected 84 wetlands from both the Tadenac Bay and North Bay regions (Figure 1). The McMaster Coastal Wetland Inventory (P. Chow-Fraser, unpublished data) was used to identify potential wetlands in the IKONOS images. We only selected wetlands with minimum area of 0.25 ha and in which at least one class of aquatic vegetation was visible. Coastal marshes in both of these regions share a similar plant zonation that is dependent on water depth. Along the shoreline (shallowest water), there tends to be a small band of meadow vegetation. As depth increases, emergent vegetation becomes increasingly dominant until it begins to blend with floating vegetation out to a depth of approximately 1.5 m. Beyond this depth, submerged aquatic vegetation is dominant out to a depth of between 4 and 6 m depending on water clarity (J. Midwood, personal observations.).

We compared changes in patch size and areal extent of vegetation cover over the two time periods because these are known to influence fish communities in wetlands (Tonn & Magnuson 1982; Dibble *et al*. 1997; Jacobus & Ivan 2005;

Jacobus & Webb 2006). In addition to analyzing classes individually, we combined the categories of E and LD into a single class of low-density-emergent (LDE) to minimize error due to misclassification (Midwood & Chow-Fraser 2010). Changes in areal vegetation coverage were calculated in ArcMap 9.2. (ESRI Inc., Redlands CA, USA, 2006) for all 84 wetlands in both Tadenac Bay and North Bay. This was accomplished by first using PTC2002 to classify the 2002 IKONOS images and then using PTC2008 to classify the 2008 imagery. Areal coverage (m^2) of each class (W, R, HD, LD, E and M) was then calculated for individual wetlands in each year. We also calculated the "visible fish habitat" category (Midwood & Chow-Fraser 2010), which is a combination of E vegetation with both LD and HD vegetation. To determine if patch size had changed from 2002 to 2008, we calculated mean patch size for the three classes that represent fish habitat (E, HD, and LD). We also calculated the maximum polygon size for the three classes because mean patch size may obscure the presence of a single large patch.

Fish Sampling

Fish sampling protocols followed those described in Seilheimer & Chow-Fraser (2006, 2007). In each wetland, three sets of paired fyke nets were used to sample the fish community. Nets were set parallel to the shoreline in beds of aquatic vegetation. Two pairs of large nets (4.25 m long, 1.0 m × 1.25 m front opening with 13 mm and 4 mm bar mesh) were set in approximately 1 m of water, and one pair of small nets (2.1 m long, 0.5 m x 1.0 m front opening with 4 mm bar mesh) were set in approximately 0.5 m of water. After 24 hours, the nets were removed and all fish were measured, counted and identified to species as per Scott and Crossman (1998). All fish were returned unharmed after processing.

Fish sampling sites in this study were chosen opportunistically based on availability of historical data (Table 1). Five of the fifteen sites were not located in the same region as our change detection analysis (Figure 1), but habitat changes should be transferable to other regions of Georgian Bay because sustained low water levels are a regional problem. Five sites had been sampled in 2003 (Green Island, Matchedash Bay, Musky Bay, Oak Bay and Quarry Island), five sites in 2004 (Green Island, Matchedash Bay, Moreau Bay, Oak Bay and Robert's Bay) and eight in 2005 (Ganyon Bay, Hermann's Bay, Lily Pond, North Bay, Ojibway Bay, Tadenac Bay 1, Tadenac Bay 2 and Treasure Bay). In 2009, all 15 wetlands were sampled once; surveys were conducted as close as possible to the date when the sites had been sampled between 2003 and 2005. The average time between sampling events was 8.3±8.0 days earlier. In some instances, sampling in 2009 was conducted considerably earlier in the season (Lily Pond 81 days, Green Island 58 days, Moreau Bay 48 days, Musky Bay 34 days) or later (Ganyon Bay 48 days).

Statistical Analysis and Calculation of Diversity

All analyses were performed in SAS JMP IN 5.1 (SAS Institute, Cary, North Carolina, U.S.A.). An ANOVA was used to assess water level changes between 2002 and 2008. A Wilcoxon post-hoc test was used to compare the mean water level because of unequal variance in the 5 years preceding 2002 and 2008. We used paired t-tests to compare changes in the same wetland between 2002 and 2008, with respect to vegetation areal coverage and structure, and among years for changes in fish species richness. Paired t-test was also used to compare proportional changes of individual fish species, but to increase sample size, data from 2003 to 2005 were combined into a single category that we have designated as “Earlier” and these were compared with data collected in 2009, which we have designated as “Later”. By using a paired analysis, we were able to control for confounding variables such as latitude, climate, exposure, and anthropogenic development, which can influence the fish community (Brazner 1997; Jude *et al.* 2005; Seilheimer & Chow-Fraser 2006; Latta *et al.* 2008; Webb 2008). In order to include rare species that could not be analyzed individually, we created a Cyprinidae category that included all members of that family. Alpha-Beta-Gamma Diversity scores were calculated according to Whittaker (1956; reviewed in Veech *et al.* 2002) for the 15 wetlands included in this study. Alpha-Diversity quantifies the diversity of the local community (within wetlands), Beta-Diversity quantifies diversity among local communities (among wetlands) and Gamma-Diversity quantifies diversity within a specific region (south-eastern Georgian Bay). Alpha and Gamma Diversity can be inferred from direct field sampling but Beta-Diversity must be calculated (Beta = Gamma–Alpha).

Results

Water Levels

Between 2002 and 2008, there was a net decline in mean water level of 0.13 m during the growing season (Figure 2). Mean water level (April to September inclusive) for the 5 years preceding 2002 was significantly higher than that corresponding to the 5 years preceding 2008 (Wilcoxon test; mean = 176.46±0.45 m, 176.10±0.13 m respectively, prob>ChiSq = 0.003, DF=1). The 5-year period preceding 2002 encompassed a rapid drop of 1.11 m, from a high of 177.10 m in 1997 to a low of 175.99 m in 2001; by comparison, water levels during the 5-year period preceding 2008 were uniformly low, varying by only 0.27 m from 176.23 m to 175.96 m.

Change detection - Accuracy

The overall accuracy of the change detection was 80.1% (product of 2002 overall accuracy = 87.4% and 2008 overall accuracy = 91.7%; Table 2). The classes with the lowest accuracy in both 2002 and 2008 were LD (74.6% and 59.0% respectively, 44.0% for the change detection; Table 2) and E (77.9% and

74.5% respectively, 58.1% for the change detection; Table 2). When these classes were combined into LDE (2002 accuracy = 86.9% and 2008 accuracy = 85.0%, 73.9% for the change detection; Table 2) the overall accuracy of the change detection increased to 85.9%. The most accurately classified feature was W (98.5% and 97.6%, 96.1% change detection; Table 2) followed by M (95.6%, 97.2%, 91.3% change detection; Table 2). Rock was the next most accurate variable (92.4%, 92.3%, 86.8% change detection; Table 2), followed by HD (88.4%, 83.8%, 74.0% change detection; Table 2).

Change detection – Areal Coverage/Patch Size

We used PTC to classify 84 wetlands included in both the 2002 and 2008 IKONOS images; these were located in both the Tadenac Bay and North Bay regions (Figure 1). The change detection confirmed that significant changes in areal cover of the main vegetation categories had occurred between 2002 and 2008 (Table 3; Figure 3). During this period, we saw a significant increase in the areal cover of M and HD, with an average increase of 2020.9 m^2 and 2312.6 m^2, respectively in each wetland (paired t-test, $p < 0.0001$, DF=83). There was a concomitant and significant decrease in cumulative areal cover of LD vegetation, with an average loss of 2995.4 m^2 (paired t-test, $p < 0.0001$, DF=83). There was also a trend towards a decrease in cover of E vegetation, with an average loss of 498 m^2 (paired t-test, $p = 0.0825$, DF=83), although this was not statistically significant. We combined the LD, HD and E to form the functional category “fish habitat” and found a significant decrease in this feature between 2002 and 2008, with an average loss of 1181.5 m^2 in each wetland (paired t-test, $p < 0.0001$, DF=83). When only LD and E were combined, we still found a significant decrease in cumulative area with a mean loss of 3494.1 m^2 in each wetland (paired t-test, $p < 0.0001$, DF=83).

The change in areal cover of vegetation classes over the six years was also accompanied by a significant increase in the number of patches of E, HD and LD (Table 4). While the number of patches of E and LD increased in 2008 relative to that in 2002, the average patch size was significantly smaller in 2008 (Table 4). Although the average patch size of HD did not change significantly, they tended to be larger (Table 4). To ensure that mean patch size had not obscured larger changes associated with a few patches, we compared maximum patch size for these vegetation classes between years. There was a significant increase in the maximum patch size for HD (an average increase of 908.9±322.5 m^2 (paired t-test, $p = 0.006$, DF=83: Table 4) and a significant decrease in maximum patch size for E (an average loss of 390.5±146.4m^2; paired t-test, $p = 0.0092$, DF=83) and LD (average loss of 1945.0±366.0 m^2; paired t-test, $p < 0.0001$, DF=83).

Fish Community

The 15 wetlands we sampled for this portion of the study ranged from 1.5 ha (Tadenac Bay 1) to 347.8 ha (Matchedash Bay), with a mean size of 37.2 ha, but 75% of the wetlands were smaller than 24 ha (Table 1). Majority of the wetlands were located in the Severn Sound region of southeastern Georgian Bay. Exceptions include Hermann's Bay (within Twelve Mile Bay), Moreau Bay (within Go Home Bay) and Tadenac Bay 1 and 2 (within Tadenac Bay; Figure 1).

A total of 40 fish taxa were identified in all surveys conducted between 2003 and 2009. Species richness corresponding to the Earlier survey (2003–2005) ranged from 5 to 20 species per wetland, compared with 4 to 10 in the Later (2009) survey (Table 1). The mean richness declined significantly from 13.2 in the initial survey to 7.2 in the more recent survey (paired t-test, $p < 0.0001$). We examined changes in the proportion of catch represented by some of the most common species sampled in eastern Georgian Bay (Table 5). Pumpkinseeds (*Lepomis gibbosus*) and bowfin (*Amia calva*) increased significantly as a proportion of our catch (paired t-test, $p = 0.0008$, and $p = 0.0009$, respectively) while tadpole madtoms (*Noturus gyrinus*), blackchin shiners (*Notropis heterodon*), black crappie (*Pomoxis nigromaculatus*), and the Cyprinidae family all decreased significantly as a proportion of our catch (paired t-test, $p < 0.05$). No significant changes in the proportion of catch were observed for brown bullhead (*Ameiurus nebulosus*), rock bass (*Ambloplites rupestris*), largemouth bass

(*Micropterus salmoides*), yellow perch (*Perca flavescens*), longear sunfish (*Lepomis megalotis*), mimic shiner (*Notropis volucellus*), and bluntnose minnow (*Pimephales notatus*), although there were trends towards increasing proportions of brown bullheads and rock bass and decreasing proportions of largemouth bass, longear sunfish, and bluntnose minnows (Table 5; Figure 4).

We also observed declines in Alpha, Beta and Gamma Diversity between the Earlier and Later surveys, indicating an overall decline in species richness over the two time periods. The mean Alpha-Diversity (within wetlands) decreased from 13.2 in 2003–2005 to 7.2 in 2009. Gamma-Diversity (within a region) also decreased from 37 (Time 1) to 24 (Time 2), and, Beta-Diversity (among wetlands) decreased from 23.8 to 16.8 over time.

All wetlands were surveyed once in a calendar year and at different times during the season (Table 1). To account for the possible confounding effects of time of sampling between the initial (2003-2005) and latter (2009) surveys, we re-analyzed the data by including only wetlands that varied by less than 2 weeks within the calendar year (n=10). We still found significant differences for species richness between survey periods (paired t-test; $p < 0.0001$).

Discussion

This is one of the first studies to utilize remote sensing to analyze change over a large geographic area of the Laurentian Great Lakes, identify significant changes in wetland vegetation in response to a loss of hydrological variability, and link changes in the fish community to these habitat changes. Our results demonstrate that sustained low water levels have resulted in encroachment of meadow vegetation into previously aquatic habitat. This has led to a net loss of aquatic vegetation, which provides critical habitat for many fish species. The remaining aquatic habitat has become increasingly homogeneous due to increased patch sizes of dense floating vegetation. During a similar time period, we have also documented a decline in fish species richness in coastal wetlands that have been impacted by sustained low water levels.

Although there has been a net decline in water levels from 2002 to 2008, we do not believe that the observed change in the fish and plant communities can be attributed to a drop of 13 cm over this period. Instead, we attribute our observations to a change in periodicity of water-level fluctuation. The rapid decline in water levels of over 1 m between 1999 and 2002 would have resulted in wetlands in a state of disequilibrium. Without episodes of high water level in the intervening years, vegetation that colonized in 2002 would have persisted and become more dense. Consistent with previous studies, we observed a significant increase in meadow vegetation in response to lower, less variable water levels (Hudon 1997; Hudon 2004; Wei & Chow-Fraser 2008; Wilcox & Nichols 2008). Thus, encroachment of meadow vegetation into areas of the marsh previously dominated by aquatic taxa has directly contributed to an overall loss of fish habitat in coastal wetlands of eastern Georgian Bay.

Of the aquatic classes, floating vegetation benefitted most from the sustained low water levels, covering more than 50% of the surface area of wetlands in dense patches by 2008, and this is consistent with findings of Quinlan & Mulamoottil (1987). Given that floating species such as *Nuphar variegata* and *Nymphaea odorata* tend to be limited to a depth of 170 cm in the coastal marshes of eastern Georgian Bay (J. Midwood, unpublished data), a drop of 13 cm would have little effect on their overall distribution. The favourable conditions, however, would have led to a transformation from primarily LD floating to HD floating over the 6 years of sustained low water levels.

In general, floating vegetation is not considered ideal fish habitat compared with emergent or submerged aquatic vegetation (SAV) because it is less structurally diverse and supports fewer epiphytes (Höök *et al*. 2001; Smokorowski & Pratt 2007), and this is especially true when it occurs in dense patches. In addition, it is undesirable because it covers the water surface, and prevents SAV from becoming established (Parr & Mason 2002), further reducing habitat structure. By comparison, suitable habitat structure is comprised of sparse patches of emergent and floating vegetation mixed with a diverse array of SAV. Therefore, conversion of LD vegetation into HD vegetation results in a net loss of desirable fish habitat.

The greatest change in coverage of HD vegetation occurred in the largest patch size, almost doubling from 2002 to 2008. The plant community changed from a heterogeneous patchwork, comprised of clusters of different vegetation, to one dominated by extensive areas containing homogeneous HD vegetation cover. This is similar to observations of Wilcox and Meeker (1991) who found that stabilization of water levels in a lentic system reduced vegetation diversity and structural complexity.

In accordance with the species-area relationship described by Arrhenius (1921), the observed decrease in the amount of available fish habitat from 2002 to 2008 resulted in lower fish species richness in coastal wetlands. Species richness not only changed at the scale of the wetland, we also observed decreases in species richness at the regional (Gamma Diversity) level, suggesting that declines in species richness may not be isolated to the 15 wetlands we sampled. While changes in the amount of habitat can explain the observed decline in species richness, the influence of concurrent changes in habitat structure on diversity must also be addressed.

Complex aquatic habitat contains numerous patches of vegetation that allow small fishes to move amongst them for foraging and protection from predators (Werner *et al*. 1983; Killgore *et al*. 1989). Large patches of contiguous dense vegetation can limit the amount of space in which prey fish can forage and force them to frequent edges of vegetation patches, where they are more vulnerable to predatory fishes, such as northern pike (*Esox lucius*), yellow perch (*Perca flavescens*) and largemouth bass (*Micropterus salmoides*), that hunt along the edge (Savino & Stein 1989, Killgore *et al.* 1989).

In a northern Lake Michigan–Huron coastal wetland, Jacobus and Webb (2006) predicted that a loss of vegetation patches with percent coverage ranging from 15–25% would have the greatest impact on fish species diversity. They also found that species richness plateaued when patches reached 128 m^2. Consistent with this prediction, we found a decline in areal coverage of LD and E (<50% coverage) as well as a significant decline in their average patch size. This has important implications because significantly fewer tadpole madtom, black crappie, blackchin shiner, and Cyprinidae were associated with these small patches of LD. Because they are key diet items of muskellunge, northern pike and largemouth bass, loss of habitat for these small fish could negatively impact these large piscivores. By contrast, some species actually prefer dense vegetation (Jacobus & Ivan 2005). For instance, we found a greater number of pumpkinseeds (*Lepomis gibbosus*) and bowfins (*Amia calva*) in the Later surveys, and this is consistent with the literature that pumpkinseeds prefer dense vegetation (Killgore *et al*. 1989) and that bowfins utilize shallow water areas with dense vegetation (Scott & Crossman 1998; Mundahl *et al.* 1998).

Due to a net loss of desirable habitat for species other than pumpkinseeds and bowfin, we observed a decline in Beta Diversity. This indicates that wetland fish communities have become less heterogeneous in composition in recent years. Because water-levels showed a net decline of only 13 cm during our study, we attribute the changes in diversity to the loss of interannual variability in water level rather than to the magnitude of water-level decline. From a management perspective, within a regulated system like the Laurentian Great Lakes it is critical to maintain as much of the natural variability in water levels regardless of the mean water levels.

The vegetation classes used in this study were formed at the level of resolution afforded by our satellite imagery. As such, we could not distinguish vegetation at the level of detail commonly used in published wetland work (i.e. species assemblages), but instead used a more simple functional taxonomy based on unique spectral signatures (Midwood and Chow-Fraser 2010). Although this limits our ability to compare directly with findings in previous literature, this approach allowed us to conduct a regional study (84 wetlands across 194 km^2) that would otherwise have been impossible given the level of difficulty in sampling Georgian Bay wetlands. We are confident that as technology improves and more investigators choose satellite platforms to produce vegetation classes, we would eventually be able to match the taxonomic resolution of conventional studies.

Few published studies have examined the influence of water-level reduction on changes in the fish community in coastal wetlands of Lake Huron. Webb

(2008) sampled five embayments in the Les Cheneaux Islands (Michigan) and found that a change of 1.2 m over a 9-year period (1996-2004) did not significantly affect the fish assemblages in the "inner marsh" where hardstem bulrush (*Schoenplectus acutus*) dominated. We attribute this apparent discrepancy in conclusions between studies to the heterogeneous nature of Webb's study sites and to geomorphological differences between wetlands in the Les Cheneaux Islands and those in southeastern Georgian Bay.

The five sites in Webb's study were heterogeneous, and varied with respect to degree of exposure and human development along the shoreline, whereas the 15 sites in this study are much more homogeneous, and are primarily protected wetlands with minimal human impact (Cvetkovic & Chow-Fraser 2011). Any effect of reduced water levels may have been masked by differences in exposure and human-induced disturbance. In addition, we argue that the cause of changes in the fish community in our study is the change in type and availability of wetland habitat resulting from the water-level decline and not merely the drop in water level itself. Hence, if the plant community in the Les Cheneaux wetlands had not changed significantly as water levels fluctuated, we should not expect a corresponding change in the fish community.

The type of aquatic vegetation in coastal marshes of the Great Lakes will depend on various factors including wetland geomorphology, bathymetry, exposure and substrate type (Keough *et al.* 1999; Riis & Hawes 2003; Albert *et al.*

2005; Capers & Les 2005). Webb (2008) sampled in a zone referred to as the "inner marsh" that occurs closest to the shoreline where there are fringing stands of hardstem bulrush (*Schoenplectus acutus*), interspersed with patches of floating taxa (primarily yellow water lily (*Nuphar variegata*)) and pondweeds (*Potamogeton spp*) and a "well-developed understory of floating or submerged swaying bulrush (*S. subterminalis*)" (Webb 2008). By comparison, the coastal wetlands of southeastern Georgian Bay have a relatively expansive and diverse emergent plant community that includes spikerush (*Eleocharis smallii.*)*,* Giant burreed (*Sparganium eurycarpum*)*,* arrowheads (*Sagittaria cuneata and S. latifolia*), pickerelweed (*Pontederia cordata*) as well as different species of bulrush (*S. acutus, S. validus and S. americanus*). This does not tend to be a well-delineated zone such as the fringing bulrush zone but is often interspersed with pockets of floating taxa such as fragrant water lily (*Nymphaea odorata*), yellow water lily (*Nuphar variegata*), floating hearts (*Nymphoides cordata*), watershield *(Brasenia schreberi*), floating burreed (*Sparganium fluctuans*) and wild rice (*Zizania palustris*). In water depths > 50 cm, submergent taxa (too many to name here) are abundant and sometimes grow luxuriantly (see Croft and Chow-Fraser 2007 for a complete list of aquatic plants). It is possible that changes in water level within this inner marsh zone did not lead to a similar change in the emergent-floating vegetation in the Les Cheneaux wetlands as they did in the Georgian Bay wetlands. Therefore, we suggest that low water levels may have differential impacts on wetlands depending on differences in geomorphology and dominant vegetation type.

To fully capture fish species richness in a wetland, investigators have suggested that a combination of different gear be used (Conrow *et al.* 1990, Weaver *et al.* 1993; Jackson & Harvey 1997; Chow-Fraser *et al.* 2006) and/or multiple sampling dates within a season be included (Pope & Willis 1996; Brazner 1997; Scott & Crossman 1998). Because our initial data were limited to single-event sampling with fyke nets, it was necessary to be consistent with our effort (Breen & Ruetz 2005) when comparing fish community assemblages between our “early” and “later” surveys. While fyke nets are known to preferentially capture small-bodied fishes (e.g. Cyprinidae; Ruetz *et al.* 2007) and cause such schooling species to exhibit an all-or-none capture rate (Uzarski *et al.* 2005), investigators have successfully utilized single-day fyke net sampling to create indices (Uzarski *et al.* 2005; Seilheimer & Chow-Fraser 2006, 2007; Bhagat *et al.* 2007) and to assess the fish community (Chow-Fraser *et al.* 2006; Uzarski *et al.* 2009). Brady *et al.* (2007) concluded that, for synoptic studies, it is better to sample more wetlands than increase effort per wetland. Therefore, despite the caveats we have mentioned here, we are confident that the changes presented in this paper are representative of the overall change in eastern Georgian Bay wetlands.

Cvetkovic *et al*. (2010) demonstrated that fish community composition in coastal wetlands is directly linked to aquatic macrophytes. To further elucidate this relationship, they recommended that studies be conducted to map habitat at a regional scale. In this study, we have demonstrated that changes in the fish community may be linked to habitat changes, identified through mapping and a change detection analysis. Our work suggests that use of remote sensing can be an effective strategy to track alteration in fish communities based on broad-scale changes in habitat structure and quantity in response to declining and/or increasingly stable water levels. The work presented in this study emphasizes the importance of maintaining water level variability, even over the short-term. Stasis in water levels allowed vegetation to increase in density and the results were an overall loss of fish habitat and a reduction in coastal wetland fish diversity

Acknowledgements

Funding for this project came from Georgian Bay Forever and the Ontario Ministry of Natural Resources through the Canada Ontario Agreement. J. Midwood received funding from the Ontario Graduate Scholarship. We are indebted to numerous summer students and other graduate students in the Chow-Fraser lab who helped to collect field samples, especially Sarah Thomasen. We also greatly appreciate the work of Maja Cvetkovic, Jeremy Lucyk and two anonymous reviewers who helped to edit earlier drafts of this manuscript.

References

Albert DA, Wilcox DA, Ingram JW, Thompson TA (2005) Hydrogeomorphic classification for Great Lakes coastal wetlands. *Journal of Great Lakes Research*, **31**(Supplement 1), 129–146.

Angel JR, Kunkel KE (2010) The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. *Journal of Great Lake Research*, **36**(S2), 51-58.

Arrhenius O (1921) Species and area. *The Journal of Ecology,* **9**(1), 95-99.

Baker C, Lawrence R, Montagne C, Patten, D (2007) Change detection of wetland ecosystems using Landsat imagery and change vector analysis. *Wetlands*, **27**(3), 610-619.

Bartlett D, Klemas V (1980) Quantitative assessment of tidal wetlands using remote sensing. *Environmental Management*, **4**(4), 337-345.

Bhagat, Y, Ciborowski JJH, Johnson LB, Uzarski DG, Burton TM, Timmermans STA, Cooper MJ (2007) Testing a fish index of biotic integrity for responses to different stressors in Great Lakes coastal wetlands. *Journal of Great Lakes Research*, **33**(Special Issue 3), 224-235.

Brady VJ, Ciborowski JJH, Johnson LB, Danz NP, Holland JD, Breneman DH, Gathman JP (2007) Optimizing fishing time: one vs. two-night fyke net sets in Great Lakes coastal systems. *Journal of Great Lakes Research*, **33**(Special Issue 3), 236-244.

Brazner JC (1997) Regional, habitat, and human development influences on coastal wetland and beach fish assemblages in Green Bay, Lake Michigan. *Journal of Great Lakes Research*, **23**(1), 36-51.

Breen MJ, Ruetz CR (2005) Gear bias in fyke netting: evaluating soak time, fish density, and predators. *North American Journal of Fisheries Management*, **26**, 32-41.

Burton TM (1985) The effects of water level fluctuations on Great Lakes coastal marshes. In: *Coastal Wetlands. Proceedings of the first Great Lakes Coastal Wetlands Colloquium* (eds Prince, HH, D'Itri FM), pp 3-13. Lewis Publishers Inc., Chelsea, MI.

Capers RS, Les DH (2005) Plant community structure in a freshwater tidal wetland. *Rhodora*, **107**(932), 386-407.

Chow-Fraser P (2006) Development of the Water Quality Index (WQI) to Assess Effects of Basin-wide Land-use Alteration on Coastal Marshes of the Laurentian Great Lakes. In: *Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators* (eds Simon TP, Stewart PM), pp 137-166. Authorhouse, Bloomington.

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

Chow-Fraser P, Kostuk K, Seilheimer T, Weimer M, MacDougall T, Theÿsmeÿer T (2006) Effect of wetland quality on sampling bias associated with two fish survey method for coastal wetlands of the lower Great Lakes. In: *Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators* (eds Simon TP, Stewart PM), pp 137-166. Authorhouse, Bloomington.

Conrow R, Zale AV, Gregory RW (1990) Distributions and abundances of early life stages of fishes in a Florida lake dominated by aquatic macrophytes. *Transactions of the American Fisheries Society*, **119**, 521–528.

Coppin P, Jonckheere I, Nackaerts K, Muys B, Lambin E (2004) Digital change detection methods in ecosystem monitoring: a review. *International Journal of Remote Sensing*, **25**(9), 1565-1596.

Croft M, Chow-Fraser P (2007) Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of Great Lakes coastal marshes. *Journal of Great Lakes Research*, **33**(S3), 172-197.

Cvetkovic M, Wei A, Chow-Fraser P (2010) Relative importance of macrophyte community versus water quality variables for predicting fish assemblages in coastal wetlands of the Laurentian Great Lakes. *Journal of Great Lakes Research*, **36**(1), 64-73.

Cvetkovic M, Chow-Fraser P (2011) Use of ecological indicators to assess the quality of Great Lakes coastal wetlands. *Ecological Indicators*, **36**, 64-73.

deCatanzaro R, Chow-Fraser P (2011) Effects of landscape variables and season on reference water chemistry of coastal marshes in eastern Georgian Bay. *Canadian Journal of Fisheries and Aquatic Sciences*, **68**, 1009-1023.

Dewan AM, Yamaguchi, Y (2009) Using remote sensing and GIS to detect and monitor land use and land cover change in Dhaka Metropolitan of Bangladesh during 1960–2005. *Environmental Monitoring and Assessment*, **150**, 237-249.

Dibble ED, Killgore KJ, Harrel SL (1997) *Assessment of fish-plant interactions*. Miscellaneous Paper A-97-6, U.S. Army Corps of Engineers, Washington, D.C.

Eadie J, Keast A (1984) Resource heterogeneity and fish species diversity in lakes. *Canadian Journal of Zoology*, **62**(9), 1689-1695.

Ficke A, Myrick C, Hansen L (2007) Potential impacts of global climate change on freshwater fisheries. *Reviews in Fish Biology and Fisheries*, **17**, 581-613.

Gathman J, Albert D, Burton T (2005) Rapid plant community response to a water level peak in northern Lake Huron coastal wetlands. *Journal of Great Lakes Research*, **31**, 160-170.

Grosshans R, Wrubleski D, Goldsborough LG (2004) Changes in the Emergent Plant Community of Netley-Libau Marsh Between 1979 and 2001. Delta Marsh Field Station (University of Manitoba) Occasional Publication No. 4, Winnipeg, Canada. 55 pp.

Hanrahan JL, Kravtsov SV, Roebber PJ (2010) Connecting past and present climate variability to the water levels of Lakes Michigan and Huron. *Geophysical Research Letters*, **37**(1), L01701.

Höök TO, Eagan NM, Webb PW (2001) Habitat and human influences on larval fish assemblages in northern Lake Huron coastal marsh bays. *Wetlands*, **21**(2), 281-291.

Houhoulis PF, Michener WK (2000) Detecting wetland change: a rule-based approach using NWI and SPOT-XS data. *Photogrammetric Engineering & Remote Sensing*, **66**(2) 205-211.

Hudon C (1997) Impact of water level fluctuations on St. Lawrence River aquatic vegetation. *Canadian Journal of Fisheries and Aquatic Sciences*, **54**(12), 2853-2865.

Hudon C (2004) Shift in wetland plant composition and biomass following low-level episodes in the St. Lawrence River: looking into the future. *Canadian Journal of Fisheries and Aquatic Sciences*, **61**(4), 603-617.

Jackson DA, Harvey HH (1997) Qualitative and quantitative sampling of lake fish communities. *Canadian Journal of Fisheries and Aquatic Sciences*, **54**, 2807-2813.

Jacobus J, Ivan LN (2005) Evaluating the Effects of Habitat Patchiness on Small Fish Assemblages in a Great Lakes Coastal Marsh. *Journal of Great Lakes Research*, **31**(4), 466-481.

Jacobus J, Webb PW (2006) Using Fish Distributions and Behavior in Patchy Habitats to Evaluate Potential Effects of Fragmentation on Small Marsh Fishes: A Case Study. *Journal of Great Lakes Research*, **31**(S1), 197-211.

Jude D, Albert D, Uzarski DG, Brazner J (2005) Lake Michigan's coastal wetlands: Distribution, biological components with emphasis on fish, and threats In: *State of Lake Michigan: Ecology, Health and Management* (eds Edsall T, Munawar M), 439-477. Ecovision World Monograph Series. Backhuys Publisher, Kerkwerve, The Netherlands.

Jude DJ, Pappas J (1992) Fish utilization of Great Lakes coastal wetlands. *Journal of Great Lakes Research*, **18**(4), 651-672.

Karl TR, Trenberth KE (2003) Modern global climate change. *Science*, **302**, 1719-1723.

Keddy P, Reznicek A (1986) Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. *Journal of Great Lakes Research*, **12**(1), 25-36.

Keough JR, Thompson TA, Guntenspergen GR, Wilcox DA (1999) Hydrogeomorphic factors and ecosystem responses in coastal wetlands of the Great Lakes. *Wetlands*, **19**(4), 821-834.

Killgore KJ, Morgan RP, Rybicki NB (1989) Distribution and abundance of fishes associated with submersed aquatic plants in the Potomac River. *North American Journal of Fisheries Management*, **9**(1), 101-111.

Latta WC, Breck JE, Marshall Duchon ER (2008) Species-area and latitudinal patterns for Michigan fishes. *American Midland Naturalist*, **159**, 349-363.

Leahy MG, Jollineau MY, Howarth PJ, Gillespie AR (2005) The use of Landsat data for investigating the long-term trends in wetland change at Long Point, Ontario. *Canadian Journal of Remote Sensing*, **31**, 240-254.

Lenters J (2001) Long-term trends in the seasonal cycle of Great Lakes water levels. *Journal of Great Lakes Research*, **27**(3), 342-353.

Lu D, Mausel P, Brondizio E, Moran E (2004). Change detection techniques. *International Journal of Remote Sensing*, **25**(12), 2365-2401.

Lyon JG, Drobney RD, Olson CE (1986) Effects of Lake Michigan water levels on wetland soil chemistry and distribution of plants in the Straits of Mackinac. *Journal of Great Lakes Research*, **12**, 175-183.

Macleod RD, Congalton RG (1998) A quantitative comparison of change-detection algorithms for monitoring eelgrass from remotely sensed data. *Photogrammetric Engineering & Remote Sensing*, **64**(3), 207-216.

Magnuson J, Webster K, Assel RA, *et al.* (1997) Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian shield region. *Hydrological Processes*, **11**(8), 825-871.

Mas J (1999) Monitoring land-cover changes: a comparison of change detection techniques. *International Journal of Remote Sensing,* **20**(1), 139-152.

McIvor C, Odum W (1988) Food, predation risk, and microhabitat selection in a marsh fish assemblage. *Ecology*, **69**(5), 1341-1351.

Midwood JD, Chow-Fraser P (2010) Mapping Floating and Emergent Aquatic Vegetation in Coastal Wetlands of Eastern Georgian Bay, Lake Huron, Canada. *Wetlands*, **30**(6), 1-12.

Mortsch L, Quinn F (1996) Climate change scenarios for Great Lakes Basin ecosystem studies. *Limnology and Oceanography*, **41**(5), 903-911.

Mortsch L, Sabila G, Deadman P (2008) Response of vegetation communities in three Lake Huron fens to historical water level fluctuations. *Aquatic Ecosystem Health & Management,* **11**(2), 1-16.

Mundahl ND, Melnytschuk C, Spielman DK, Harkins JP, Funk K, Bilicki AM (1998) Effectiveness of bowfin as a predator on bluegill in a vegetated lake. *North American Journal of Fisheries Management*, **18**, 286-294.

Parr L, Mason C (2004) Causes of low oxygen in a lowland, regulated eutrophic river in Eastern England. *Science of the Total Environment*, **321**(1-3), 273-286.

Pope KL, Willis DW (1996) Seasonal influences on freshwater fisheries sampling data. *Reviews in Fisheries Science*, **4**(1), 57-73.

Quinlan C, Mulamoottill G (1987) The effects of water level fluctuations on three Lake Ontario shoreline marshes. *Canadian Water Resource Journal*, **12**(1), 64-77.

Quinn F (2002) Secular changes in Great Lakes water level seasonal cycles. *Journal of Great Lakes Research*, **28**(3), 451-465.

Randall RG, Minns CK, Cairns VW, Moore JE (1996) The relationship between an index of fish production and submerged macrophytes and other habitat features at three littoral areas in the Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*, **53**(S1), 35-44.

Riis TA, Hawes I (2003) Effect of wave exposure on vegetation abundance, richness and depth distribution of shallow water plants in a New Zealand lake. *Freshwater Biology*, **48**, 75-87.

Ross RM, Lellis WA, Bennett RM, Johnson CS (2001) Landscape determinants of nonindigenous fish invasions. *Biological Invasions*, **3**, 347–361.

Ruetz, CR, Uzarski DG, Krueger DM, Rutherford ES (2007) Sampling a littoral fish assemblage: comparison of small-mesh fyke netting and boat electrofishing. *North American Journal of Fisheries Management*, **27**, 825-831.

Savino J, Stein R (1982) Predator-prey interaction between largemouth bass and bluegills as influenced by simulated, submersed vegetation. *Transactions of the American Fisheries Society*, **111**, 255-266.

Savino J, Stein R (1989) Behavior of fish predators and their prey: habitat choice between open water and dense vegetation. *Environmental Biology of Fishes*, **24**(4), 287-293.

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

Seilheimer T, Chow-Fraser P (2006) Development and use of the Wetland Fish Index to assess the quality of coastal wetlands in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences,* **63**, 354-366.

Seilheimer T, Chow-Fraser P (2007) Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. *Journal of Great Lakes Research*, **33**(SP3), 154-171.

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

Silva TSF, Costa MPF, Melack JM, Novo EMLM (2008) Remote sensing of aquatic vegetation: theory and applications. *Environmental Monitoring and Assessment*, **140**(1), 131-145.

Smokorowski K, Pratt T (2007) Effect of a change in physical structure and cover on fish and fish habitat in freshwater ecosystems-a review and meta-analysis. *Environmental Reviews*, **15**, 15-41.

Tonn W, Magnuson J (1982) Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. *Ecology*, **63**(4), 1149-1166.

Trebitz AC, Brazner JC, Pearson MS, Peterson GS, Tanner DK, Taylor DL (2009) Patterns in habitat and fish assemblages within Great Lakes coastal wetlands and implications for sampling design. *Canadian Journal of Fisheries and Aquatic Sciences*, **66**(8), 1343-1354.

Trenberth KE, Dai A, Rasmussen RM, Parsons DB (2003) The Changing Character of Precipitation. *Bulletin of the American Meteorological Society*, **84**(9), 1205-1217.

Uzarski DG, Burton TM, Cooper MJ, Ingram JW, Timmermans STA (2005) Fish habitat use within and across wetland classes in coastal wetlands of the five Great Lakes: development of a fish-based index of biotic integrity. *Journal of Great Lakes Research*, **31**, 171–187.

Uzarski DG, Burton TM, Kolar RE, Cooper MJ (2009) The ecological impacts of fragmentation and vegetation removal in Lake Huron's coastal wetlands. *Aquatic Ecosystem Health and Management*, **12**(1), 45-62.

Van Oort PAJ (2007) Interpreting the change detection error matrix. *Remote Sensing of Environment*, **108**(1), 1-8.

Veech JA, Summerville KS, Crist TO, Gering JC (2002) The additive partitioning of species diversity: recent revival of an old idea. *Oikos*, **99**(1), 3-9.

Weaver MJ, Magnuson JJ, Clayton MK (1993) Analyses for differentiating littoral fish assemblages with catch data from multiple sampling gears. *Transactions of the American Fisheries Society*, **122**, 1111-1119.

Webb PW (2008) The impact of changes in water level and human development on forage fish assemblages in Great Lakes coastal marshes. *Journal of Great Lakes Research*, **34**, 615-630.

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

Wei A, Chow-Fraser P (2005) Untangling the confounding effects of urbanization and high water level on the cover of emergent vegetation in Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario. *Hydrobiologia*, **544**(1), 1-9.

Wei A, Chow-Fraser P (2008) Testing the transferability of a marsh-inundation model across two landscapes. *Hydrobiologia*, **600**(1), 41-47.

Werner EE, Gilliam JF, Hall DJ, Mittelbach GG (1983) An experimental test of the effects of predation risk on habitat use in fish. *Ecology*, **64**(6), 1540-1548.

Wilcox DA, Meeker JE (1991) Disturbance effects on aquatic vegetation in regulated lakes in northern Minnesota. *Canadian Journal of Botany*, **69**(7), 1542-1551.

Wilcox DA (2004) Implications of hydrologic variability on the succession of plants in Great Lakes wetlands. *Aquatic Ecosystem Health and Management*, **7**(2), 223-231.

Wilcox DA, Nichols S (2008) The effects of water-level fluctuations on vegetation in a Lake Huron wetland. *Wetlands*, **28**(2), 487-501.

Zhou H, Jiang H, Zhou G, *et al.* (2010) Monitoring the change of urban wetland using high spatial resolution remote sensing data. *International Journal of Remote Sensing*, **31**(7), 1717-1731.

Zhou W, Troy A, Grove M (2008) Object-based land cover classification and change analysis in the Baltimore metropolitan area using multitemporal high resolution remote sensing data. *Sensors*, **8**(8), 1613-1636.

Table 1. Location (decimal degrees) and size of wetlands where the fish community was surveyed. Species richness and sampling dates (in brackets) for each year are also shown.

Wetland Name	Wetland Code	Latitude	Longitude	Wetland Size (ha)	2003 Species Richness	2004 Species Richness	2005 Species Richness	2009 Species Richness
Ganyon Bay	GY	44.91995	-79.81976	1.90	—	—	8 (Aug 5)	6 (Jun 18)
Green Island	GI	44.78574	-79.74797	4.90	18 (Jul 9)	14 (Jun 3)	—	7 (Aug 18)
Hermann's Bay	HRM	45.08662	-79.99669	2.90	—	—	6 (Aug 31)	5 (Aug 18)
Lily Pond	LY1	44.87076	-79.81547	3.20	—	—	8 (Sep 1)	10 (Jun 12)
Matchedash Bay	MB	44.75885	-79.69687	347.80	17 (Jul 8)	15 (May 27)	—	12 (May 27)
Moreau Bay	MO	45.01460	-79.94510	23.60	—	17 (Jun 17)	—	6 (Aug 5)
Musky Bay	MS	44.81197	-79.77945	19.40	18 (Jul 9)	—	—	9 (Aug 12)
North Bay	NB	44.89717	-79.79465	10.30	—	—	13 (Jun 15)	6 (Jun 16)
Oak Bay	OB	44.79466	-79.73221	50.20	11 (Jul 8)	14 (Jun 9)	—	9 (Jun 9)
Ojibway Bay	OJ	44.88786	-79.85587	1.70	—	—	10 (Jun 15)	7 (Jun 24)
Quarry Island	QI	44.83510	-79.80897	21.20	20 (Jul 10)	—	—	7 (Jun 25)
Robert's Bay	RB	44.85583	-79.83063	6.00	—	15 (Jun 2)	—	4 (Jun 17)
Tadenac Bay 1	TD1	45.03583	-79.99325	1.50	—	—	10 (Jul 19)	5 (Jul 16)
Tadenac Bay 2	TD2	45.03977	-79.98508	2.70	—	—	5 (Jul 20)	7 (Jul 15)
Treasure Bay	TB	44.87190	-79.86013	60.20	—	—	15 (Jun 14)	8 (Jun 23)

Table 2. Combined accuracy for the change detection based on class. LD Floating and Emergent vegetation classes were combined during classification to form the LDE category.

Class	2002 Accuracy (%)	2008 Accuracy (%)	Change Detection Accuracy
Meadow	95.6	97.2	91.3
HD Floating	88.4	83.8	74.0
LD Floating	74.6	59.0	44.0
Emergent	77.9	74.5	58.1
Rock	92.4	92.3	86.8
Water	98.5	97.6	96.1
Overall Accuracy	87.4	91.7	80.1
LDE	86.9	85.0	73.9
Overall Accuracy w LDE	94.1	91.3	85.9

Table 3. Areal change in vegetation coverage for 84 wetlands based on 2002 and 2008 IKONOS imagery. LD Floating and Emergent vegetation classes were combined during classification to form the LDE category.

	Meadow	HD Floating	LD Floating	Emergent	LDE	Total Area Fish Habitat
% Sites Increasing	88.0	89.0	4.0	35.7	10.7	27.0
% Sites Decreasing	12.0	11.0	96.0	64.3	89.3	68.0
Mean Change	*2020.9 m^2	*2312.6 m^2	*2995.4 m^2	498.7 m^2	*3494.1	*1181.5 m^2

* prob. >|t| = <0.0001; N = 84

Table 4. Structural changes in wetland vegetation based on changes observed in 2002 and 2008 IKONOS images. The M, R, and W class are not included because they are not considered components of fish habitat. LD Floating and Emergent vegetation classes were combined during classification to form the LDE category

	Δ # Patches	**Δ Mean Patch Size (m^2)**	**Δ Max Patch Size (m^2)**
Emergent	+22 ± 3*	-50.9 ± 7.7*	-390.5 ± 146.3**
HD Floating	+39 ± 6*	+7.3 ± 3.9	+908.9 ± 322.5**
LD Floating	+76 ± 14*	-92.5 ± 9.2*	-1945.0 ± 366.0*
LDE	+85 ± 14*	-165.9 ± 21.2*	-3584.9 ± 834.5*

* prob. >|t| = <0.0001; N = 84
** prob. >|t| < 0.05; N = 84

Table 5. Comparison of the proportion of the 13 most common fish species or groups between "earlier" and "later" sampling period (2003–2004–2005 and 2009, respectively). P-values in bold indicate significant differences between survey periods.

Common Name	Scientific Name	P value	Mean "Earlier" Proportion of Catch	Mean "Later" Proportion of Catch
Pumpkinseeds	*Lepomis gibbosus*	**0.0008**	0.37	0.69
Bowfin	*Amia calva*	**0.0009**	0.01	0.06
Tadpole Madtom	*Noturus gyrinus*	**0.0219**	0.02	0.00
Blackchin Shiner	*Notropis heterodon*	**0.0475**	0.02	0.00
Black Crappie	*Pomoxis nigromaculatus*	**0.0217**	0.03	0.00
Brown Bullhead	*Ameiurus nebulosus*	0.1080	0.13	0.06
Rock Bass	*Ambloplites rupestris*	0.7080	0.03	0.04
Largemouth Bass	*Micropterus salmoides*	0.1580	0.14	0.05
Yellow Perch	*Perca flavescens*	0.7423	0.03	0.03
Longear Sunfish	*Lepomis megalotis*	0.2242	0.27	0.01
Mimic Shiner	*Notropis volucellus*	0.0894	0.02	0.00
Bluntnose Minnow	*Pimephales notatus*	0.0810	0.05	0.01
Carps & Minnows	Cyprinidae	**0.0299**	0.15	0.02

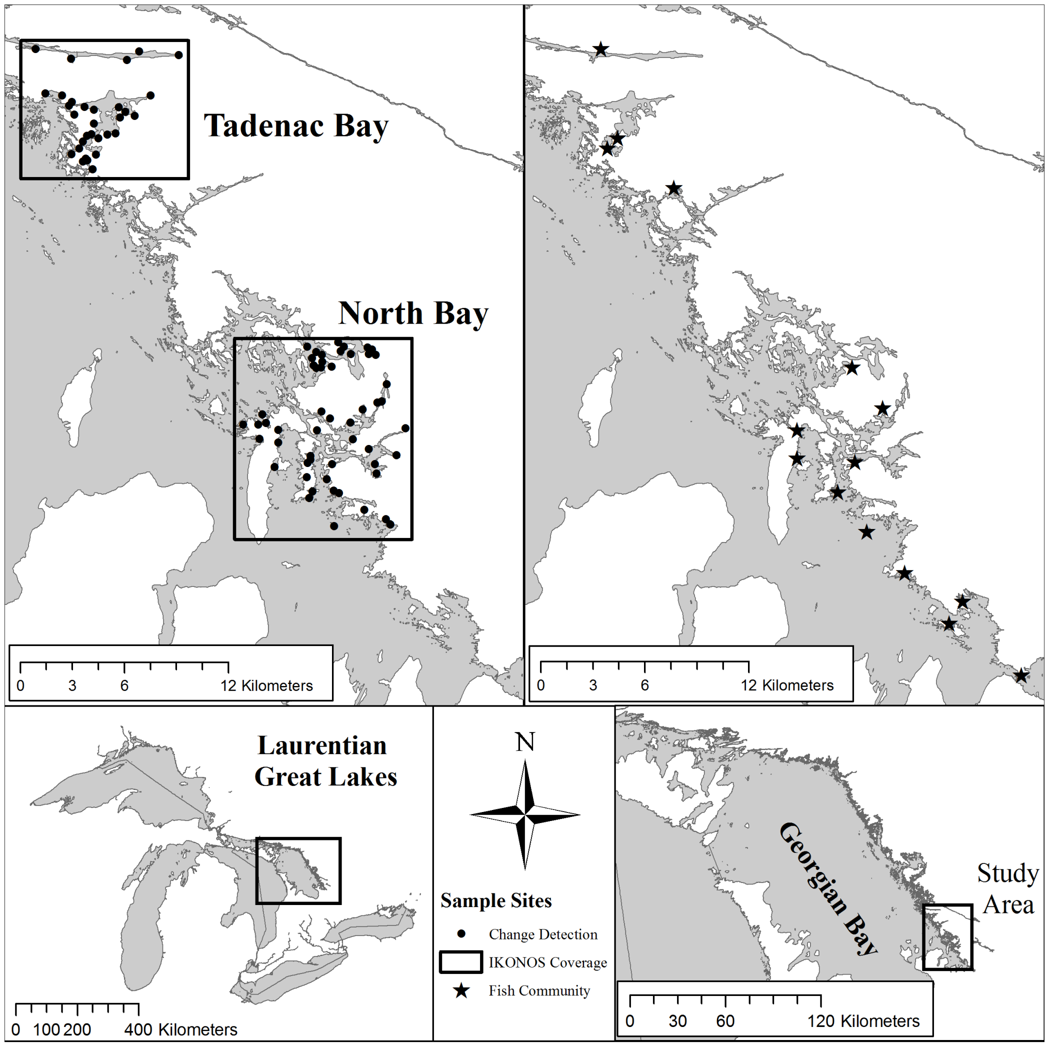


Figure 1. Location of 84 wetlands (round dots) used in the analysis of change detection. Wetlands were located in two regions; Tadenac Bay is a relative pristine area with minimal human development. By comparison, North Bay is more densely populated and has greater boat traffic. IKONOS satellite images covering both regions were acquired in July 2002 and again in July 2008. Wetlands where fish data were collected (stars) partially overlap with wetlands used in the change detection analysis.

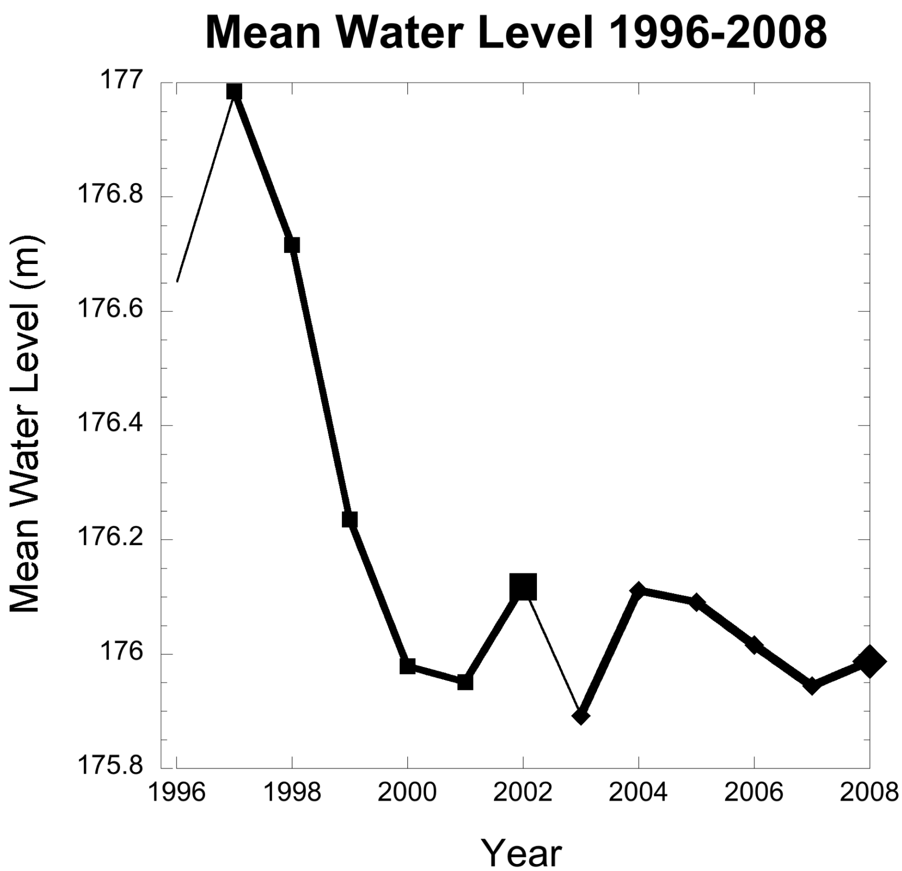


Figure 2. Change in water levels of Lake Huron from 1996 to 2008 (Data from Canadian Hydrographic Services, Department of Fisheries and Oceans). The large square and diamond represent the years IKONOS imagery was acquired (2002 and 2008, respectively). Thicker lines show the water levels in the five years preceding imagery acquisition

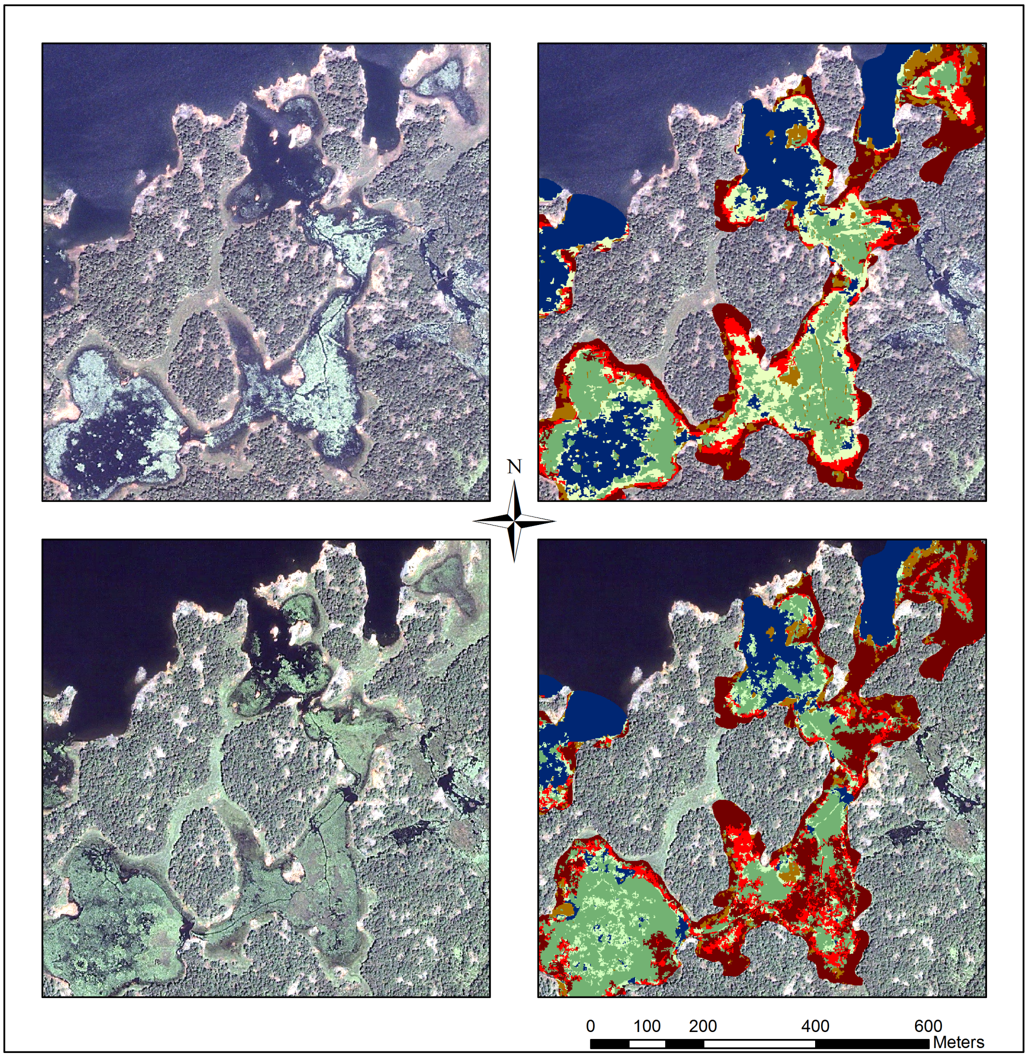


Figure 3. Comparison of two original IKONOS images (A,B) with images that have been classified (C,D). Red=emergent vegetation, dark green = dense floating vegetation, light green=sparse floating vegetation, maroon=meadow vegetation, blue=water and brown=rock. All images show Black Rock Bay in the Tadenac Bay region of eastern Georgian Bay. The top images were acquired July 1st, 2002 and the bottom images were acquired July 16th, 2008. Comparing image C to image D it is clear that meadow vegetation (maroon) has colonized previously aquatic habitats.

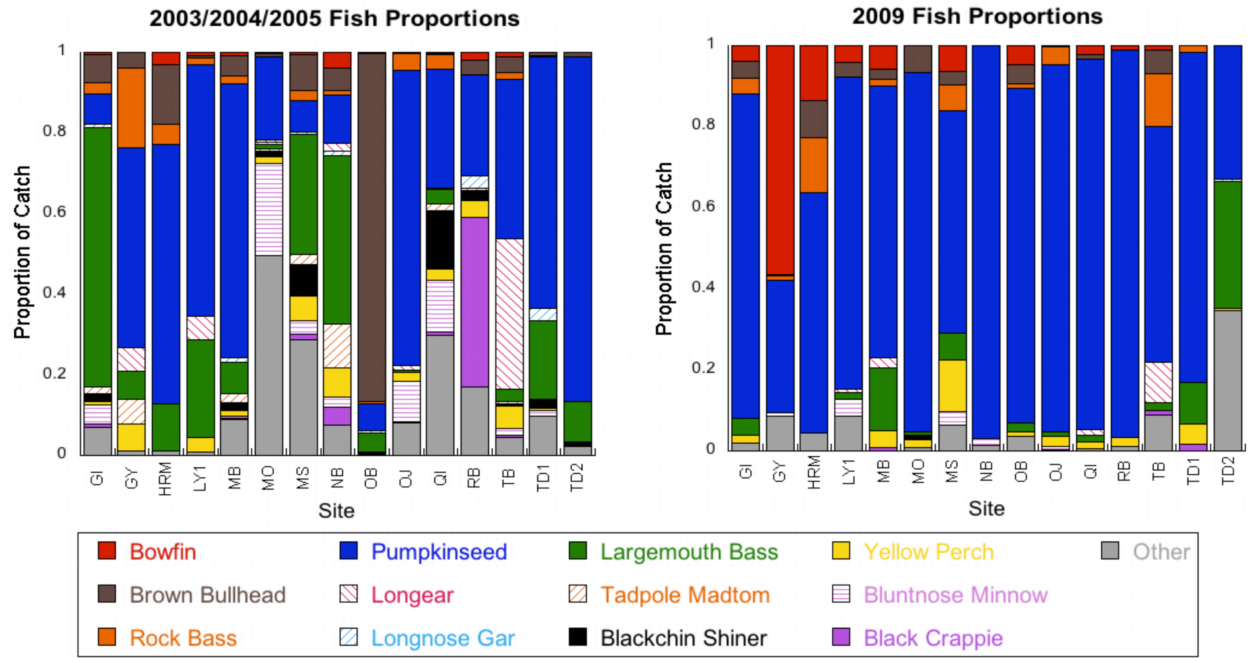


Figure 4. Proportion of catch represented by each species, in each wetland for the "Early" (20032005) and "Later" (2009) sampling periods. There was a significant decline in species richness from the Early to Later time periods.

Chapter 5:

Complexing coastal marshes of eastern Georgian Bay using movements of resident and migratory fishes.

Abstract

Coastal wetlands provide critical spawning and foraging habitat for fishes. In the Laurentian Great Lakes, diurnal migration of fish into and from wetlands is well documented, but movement among coastal wetlands is more poorly understood despite the important conservation implications. The Ontario Wetland Evaluation System (OWES) affords protection to large wetlands (>2 ha), but smaller wetlands can be grouped into complexes if they are closer than 750 m. In the numerous and predominantly small (<2 ha) coastal wetlands of eastern Georgian Bay, Lake Huron, sustained low water levels have altered fish habitat. In many instances these wetlands are spread out beyond the current OWES complexing limit of 750 m. Therefore in order to protect Georgian Bay wetlands and maintain the fish community within them, it is essential to understand how both resident and migratory fishes utilize small, locally situated wetlands. In the summer of 2010 we assessed fish movement in two regions, Tadenac Bay and Moon Island. In each region, four-five wetlands located in close proximity were sampled eight times. Fish caught in each wetland were tagged with a wetland-specific colour. Majority of fishes were wetland residents, with pumpkinseeds (*Lepomis gibbosus*) accounting for 70% of the total catch. In total, 5537 fish were tagged and 146 of these were recaptured (2.6 %). Of these, 9 (6.2%) were recaptured in a wetland different from where they had been tagged. In 2011, we resampled wetlands in Tadenac Bay to determine over-wintering movements. Of

the 3359 fish caught, 23 were tagged (0.7%), and of these only 1 (4.3%) traveled beyond the wetland where it had been originally tagged. For both within-season and annual movements, the majority of fishes recaptured did not travel beyond their wetland of origin. Furthermore, in 2011 we implanted radio tags in 12 northern pike (*Esox lucius*), a migratory species, to track their movements among coastal wetlands. Northern pike that frequented wetland areas tended to be young (2-5 years) and small (<600 mm). On average, these smaller northern pike moved among wetlands that were 1.4 km apart, although some moved as far as 3.9 km. Our results suggest that while the vast majority of fishes remain in a single wetland throughout the year, northern pike use multiple wetlands over relatively large areas during the active season. This suggests that while the current distance used by OWES for delineating wetland complexes (750 m) likely protects most resident fish species, it does not cover the observed movement patterns of a top predator, the northern pike. A modification to this OWES rule for coastal wetlands would help to more accurately delineate complexes and protect critical fish habitat in the Great Lakes.

Introduction

Habitat is inherently defined at a species-specific scale (Franklin et al. 2002), yet conservation measures are typically implemented at regional scales for protection of multiple species and ecosystems. Therefore, conservation efforts must incorporate the diverse spatial requirements of all species in order to protect and maintain biodiversity (Noss 1992; Sale 1998). The spatial requirements of organisms that move diurnally and seasonally require greater effort to quantify, but such information is critical for fish species that form metacommunities (Sale 1998; Gotelli and Taylor 1999; Mouillot 2007). It is widely accepted that many fish exhibit diurnal movements between the nearshore and offshore in freshwater ecosystems; however, few studies exist that document movements among discrete environments within a region, even though such movements within a metapopulation help to maintain genetic diversity at both a population and community level (Jackson et al. 2001).

Thousands of coastal wetlands occur along eastern Georgian Bay (Ontario, Canada) providing spawning and foraging habitat for majority of fish species in Lake Huron (Jude and Pappas 1992; Randall et al. 1997; Wei et al. 2004; Cvetkovic et al. 2010). The aquatic portion of these wetlands (referred to as *low marsh*) is the only portion that can be used by the fish community, and on average it covers an area of 1.4 ha (Midwood et al. 2012). Although these wetlands are currently pristine (Chow-Fraser 2006; Cvetkovic and Chow-Fraser 2011), the negative impact of recreational and urban development has risen in recent decades and is expected to increase. A more insidious threat is the loss of critical habitat due to a decade of sustained low water levels that has been associated with a decline in fish species richness (Midwood and Chow-Fraser 2012). Climate change models forecast even lower water levels that will likely diminish overall fish habitat quality and quantity, and will thus continue to negatively impact the coastal fish community (Mortsch and Quinn 1996; Sellinger et al. 2008; Angel and Kunkel 2010). Protection of these pristine wetlands is therefore essential to prevent human disturbance from compounding the observed impact of declining water levels.

Since Georgian Bay falls entirely within the province of Ontario, protection of its coastal marshes falls under the jurisdiction of the Ontario Ministry of Natural Resources (OMNR). Wetlands must undergo an evaluation based on the Ontario Wetland Evaluation System (OWES; OMNR 1993) in order to be deemed provincially significant, and thus provided some measure of provincial protection. To qualify for evaluations, wetlands must be at least 2 ha in size. Alternatively, OWES allows small wetlands, such as the ones in eastern Georgian Bay, to be grouped into complexes if they are within 750 m of each other and/or there is biological evidence to support grouping them (OMNR 1993). A recent inventory of eastern Georgian Bay wetlands found that 89 % of the 3771 aquatic marshes are less than 2 ha in size (Midwood et al. 2012). This excludes the majority of Georgian Bay wetlands from protection unless they are within 750 m of each other or there is documented evidence of fish movement amongst them. To date, no studies have been conducted to quantify the distances moved by fishes in and among coastal wetlands of Georgian Bay.

Jude and Pappas (1992) identified two main groups of wetland fishes, resident and migratory species. Resident fishes are typically small-bodied fishes that are wetland obligates, spending the majority of their life in wetlands. Migratory fishes fall into three groups: *spawning non-resident* are those that visit wetlands only during their spawning season, *nursery* species that only remain in wetlands until they reach maturity, and *wanderers* that occasionally pass through wetlands, but are uncommon. At least one species should be selected from both resident and migratory wetland groups to fully evaluate fish movement among wetlands.

Three very common and abundant *resident* taxa in Georgian Bay wetlands include the pumpkinseed (*Lepomis gibbosus*), yellow perch (*Perca flavescens*) and largemouth bass (*Micropterus salmoides*) (Cvetkovic et al. 2010). Fish and Savitz (1983) determined home ranges for these species and found a home range of 0.23-1.12 ha for pumpkinseeds, 0.54-2.20 ha for yellow perch, and 0.18-2.07 ha for largemouth bass. Assuming these home ranges can be applied to Georgian Bay, it is conceivable that at least some individuals may use multiple wetlands (with mean size of 1.4 ha) throughout their lives, while others may not move at all. By comparison, northern pike (*Esox lucius*), which is a much larger piscivorous species and often the top predator in coastal systems (Scott and Crossman 1998), has been identified as a *migratory* species that uses wetlands for both spawning and nursery habitat and can move daily up to 8000 m (Diana and Mackay 1977; Cook and Bergersen 1988; Jude and Pappas 1992; Koed et al. 2006; Kobler et al. 2008). It is therefore likely that northern pike in Georgian Bay would move freely among several adjacent wetlands.

Our overall goal in this paper is to quantify the movement of common fish species in and among wetlands in order to evaluate the appropriateness of the current OWES complexing distance of 750 m. We predict that the small, *resident* fishes will move shorter distances compared with the large *migratory* northern pike. To determine average distances moved by *resident* fishes, we used a mark-recapture program in two minimally disturbed embayments of eastern Georgian Bay, Moon Island and Tadenac Bay. We then executed a radio-tracking study the following year to further track the distance moved by a *migratory* fish, the northern pike, among adjacent coastal wetlands within Tadenac Bay. By knowing how far each species travels away from its wetland of origin, we can develop appropriate guidelines to combine wetlands into wetland-complexes that reflect meaningful ecological relationships.

Methods

Study Sites

Coastal wetlands in eastern Georgian Bay typically form in protected embayments. The underlying substrate is granitic rock, and consequently the water is characteristically dystrophic with low nutrient levels (DeCatanzaro and Chow-Fraser 2011). To assess the movement of wetland resident fishes, we used the McMaster Coastal Wetland Inventory (MCWI; Midwood et al. 2012) to identify several clusters of small wetlands that included at least three wetlands within 750 m of each other and one wetland beyond this distance. Site selection was further refined to: 1) minimize the potentially confounding impacts of human disturbance on fish behaviour, and 2) ensure easy access to the study sites. Based on these search criteria, we identified two wetland clusters in eastern Georgian Bay that are accessible and minimally impacted: Moon Island and Tadenac Bay (Chow-Fraser 2006; Cvetkovic and Chow-Fraser 2011; Figure 1). While differences in dominant vegetation and substrate types were observed in these two regions (Data not shown), due to the absence of human impacts in both locations, we combined fish movement data from both regions to expand the applicability of our findings to more Georgian Bay wetlands.

The Moon Island cluster is located in Massasauga Provincial Park where "back-country" camping is the primary source of disturbance. Five coastal wetlands (MA, MB, MC, MD, and ME) ranging in size from 0.43 ha to 1.71 ha (mean = 1.20 ha) were sampled in Moon Island (Figure 1; Table 1). On average, the distance between wetland centroids was 810±540 m. Initially four of the five wetlands were selected for sampling due to limited availability of sampling gear. Wetland ME was dropped mid-sampling as it became hydrologically disconnected. In its place, wetland MC was added.

The second wetland cluster is located in Tadenac Bay, a privately owned fishing camp with minimal development. Recreational fishing is the main activity in this Bay, but catch-and-release angling is typically practiced. Four coastal wetlands were sampled in Tadenac Bay (TA, TC, TD, and TE; Figure 1; Table 1). These wetlands ranged in size from 1.45 ha to 2.36 ha (mean = 1.54 ha; Table 1), with an average distance among them of 950±670 m.

Due to the large home range of northern pike, mark-recapture methods were not a viable option, and we opted to instead track northern pike across the > 400 ha embayment of Tadenac Bay (Figure 2). Tadenac Bay was selected over Moon Island for this portion of the study because it has only one access point to Georgian Bay, which would potentially allow us to determine if northern pike had left our study area. In addition, Tadenac Bay is relatively large with 39 wetlands containing 63.8 ha of potential fish habitat.

<u>*Mark-Recapture*</u>

Fish Sampling

In 2010, fyke nets were set over 8 weeks from May to September (4 weeks in Moon Island and 4 weeks in Tadenac Bay). In order to limit biases associated with selection of sampling location, we divided up the shoreline of each wetland into 15 m-wide segments, and a random number table was used to select the segments where the nets were to be set. Each week, three sets of paired fyke nets (two large nets, 4.25 m long, 1.0-m × 1.25-m front opening with 13- and 4-mm bar mesh and one small net, 2.1 m long, 0.5-m x 1.0-m front opening with 4-mm bar mesh; see Seilheimer and Chow-Fraser 2007) were set twice in each wetland on alternating days. Fyke nets were left in each wetland for ~20 hr in order to capture the diurnal movement of fishes. In Moon Island, wetland ME was sampled during weeks 1 and 2 and wetland MC was sampled during weeks 3 and 4.

To assess the potential for over-winter movement, in summer 2011 we resampled wetlands in Tadenac Bay over a 2-week period following the same protocol. Each wetland was sampled four times, twice in late May and twice in July. Each fish was inspected for tags from the previous summer; no new fish were tagged.

Fish Tagging

Fish tagging only occurred during summer 2010. Visible Implant Elastomer (VIE) tags (Northwest Marine Technology Inc., Shaw Is., Washington, USA) were selected for this project because they are easily applied, have a negligible impact on the fishes, are low cost, and are viable for the duration of the study (Malone et al. 1999; McCairn and Fox 2004; Hoey et al. 2006; Jacobus and Webb 2006). Each wetland was assigned a unique colour (Table 1) and all fish captured, with the exception of those less than 50 mm in length, were identified to species, measured, and tagged in one of four body-locations depending on the week they were captured. Due to handling difficulties, no brown bullheads (*Ameiurus nebulosus*) were tagged. Prior to tagging, fish were anaesthetized in a solution of 0.4% clove oil until they could no longer right themselves (typically 3-5 min). In weeks 1 and 2, fish were tagged on the right and left cheek, respectively. For weeks 3 and 4, fish were tagged on the right and left side of the body, respectively, anterior to the caudal fin. By adjusting the tag location, it allowed us to determine in which week a fish had been tagged. Since large fish (>250 mm) were only caught infrequently, we did not vary their tagging location. Instead, each fish was tagged multiple times on the caudal fin. We also found that fish in the family Cyprinidae could not be tagged in any of our four body-locations; instead these fishes were tagged on the right (weeks 1 and 3) or left (weeks 2 and 4) side of the dorsal fin.

<u>*Radio-Tracking*</u>

Fish Sampling

Northern pike were captured in trap nets (2 m x 3 m) that were set overnight, perpendicular from shore, and in a minimum water depth of 2 m. Nets were set in four locations spaced throughout Tadenac Bay (Figure 2). Site 1 was situated in the same embayment as our mark-recapture study. Site 2 was in a location where northern pike had been found during the OMNR End-Of-Spring-Trap-Net surveys (E. McIntyre, pers. comm.). The final two locations (sites 3 & 4) were situated in areas where anglers of Tadenac Club tended to catch northern pike (M. Trudeau, pers. comm.).

Fish Tagging

Twelve pike that weighed greater than 1.0 kg were selected for tagging. This ensured that the weight of the radio-tag (16 g) represented less than 2% of their body weight (Rogers and White 2007). Captured northerrn pike were kept in the net to await surgery. Pike were then moved into a 60-L container filled with 20 L of 60 ppm clove oil. Once northern pike ceased to respond to external stimuli, their length and weight were measured and the sex was determined as per Casselman (1974). Based on their length and weight, age of each northern pike was estimated according to Wainio (1966, in Scott and Crossman 1998). Northern pike were then introduced ventral-side up in a U-shaped foam surgical table. A maintenance anesthetic dose of 30 ppm of clove oil was pumped over their gills to maintain their anesthetized state. First, a 2-3 cm incision was made mid-ventral and anterior to the pelvic girdle. Then a small hole on the left side of the body was made with a 16-gauge needle. The transmitter antenna was run through this hole with help from the needle and the transmitter was inserted into the body cavity. Fish were implanted with a Lotek (Newmarket, ON) MCFT2-3A radio-transmitter (16-mm diameter x 46-mm length). Incisions were closed with two interrupted 3-0 monofilament sutures and at least two throws of a surgeons knot. Following surgery, fish were placed on top of the trap net and immersed in water from their natural environment so that their recovery could be monitored. Similar procedures have been employed in other studies with minimal impacts to the fish (see Cooke et al. 2003; Koed et al. 2006).

Northern pike tracking began 2 weeks after surgery, which is the recommended time to ensure that they had recovered and returned to their natural movement patterns (Rogers and White 2007; Kobler et al. 2008). We conducted intensive morning, afternoon, and evening surveys once a month throughout the summer, for a total of 4 weeks, starting May 24 and ending on August 24. Between these 4weeks, four single-day surveys (at roughly weekly intervals) were conducted opportunistically for a grand total of 52 surveys. A survey consisted of driving a set route by boat through our study area. During this drive, we used a

Lotek F150-3FB radio antenna and a Lotek SRX_400A/WX5G manual tracking radio receiver to identify the location of the northern pike. During the afternoon survey, we located the northern pike using a standard triangulation method; for the morning and evening surveys, northern pike locations were not triangulated due to time constraints, and a single GPS point was transcribed onto a map to represent the northern pikes' location. To determine if a northern pike had left our study area, we established a base-station at the entrance of Tadenac Bay that consisted of a receiver and antenna powered by a marine battery. This station monitored the entrance/exit to Tadenac Bay 24-hours a day and recorded all northern pike passing by. Unfortunately, early in the study we discovered that the base-station had blind spots that allowed northern pike to move past without detection, and in early August, the base-station failed completely. Although all observations of northern pike from the base-station are included, we will not discuss these data independently.

GIS Analysis & Statistics

Mark-recapture data were brought into a GIS (ArcMap 9.2 ESRI Inc., Redlands, California, U.S.A., 2006) and the minimum distance between the initial tagging and recapture locations were measured for all recaptured fishes. Distances were measured as the shortest straight-line distance passing through the water between the initial tag location and the capture point. If the exact tagging location of an individual could not be determined (as was often the case for pumpkinseeds), an average distance was calculated from all possible tagging locations to the recapture site. These movement distance measurements likely represent a conservative estimate of actual movement.

For the northern pike, all sample locations (from both triangulation and mapping) were entered into a GIS for further analysis. By overlaying their positions on a file containing all wetlands, we were able to determine the number of wetlands with which each pike was associated. We considered a northern pike to be "associated" with a wetland if it was found within that wetland or it was within 35 m of the wetland. This distance was deemed to be a conservative estimate that would include the submerged aquatic vegetation adjacent to the wetland (J. Midwood pers. obs.). Once these wetlands were identified for each northern pike, the average, minimum, and maximum distances among them were measured.

Results

<u>Mark-Recapture</u>

Summary of Fish Caught

In Moon Island, in total 2742 fish were tagged in the five wetlands (Figure 3; Table 1). Pumpkinseeds (*Lepomis gibbosus*) were by far the most commonly captured species, accounting for 69.9% of the total catch (Table 2). Other common species included largemouth bass (*Micropterus salmoides*) and yellow perch (*Perca flavescens*)(12.3% and 11.3%, respectively). Of the remaining 11 species, only two accounted for more than 1%: rock bass (*Ambloplites rupestris*) and bluntnose minnows (*Pimephales notatus*)(2.04% and 1.82%, respectively).

In the four wetlands in Tadenac Bay, a total of 3498 fish were tagged (Table 1). Pumpkinseeds were once again the most common species, accounting for 68.5% of the total catch (Table 2). Other common species were largemouth bass, bluntnose minnow, yellow perch, and longear sunfish (*Lepomis megalotis*)(8.95%, 6.17%, 5.89%, and 5.32%, respectively). Of the remaining 11 species, only two accounted for more than 1%: rock bass and black crappie (*Pomoxis nigromaculatus*)(2.34 % and 1.06%, respectively).

Overall Movements

In the summer of 2010, in total 146 tagged fish were recaptured. With the exception of bowfin (*Amia calva*), the six most commonly tagged fishes were also the only species that were recaptured. Based on the estimated distance between tagging and recapture location, bowfin and largemouth bass travelled the farthest (480 m ± 206 m and 135 m ± 214 m, respectively; Table 3). Longear sunfish were next at 82 m ± 35 m, but they were only found in Tadenac Bay. Pumpkinseeds were recaptured most frequently (116) and on average travelled 78 m ± 51 m. Rock bass and bluntnose minnows were found to move the shortest distances of

40 m ± 29 m and 27 m ± 17 m, respectively. Due to the low number of recaptures (2), average distance travelled could not be calculated with standard error for yellow perch, which were only recaptured in Moon Island. Movement distance between 2010 tagging and recapture in 2011 were not estimated for individuals unless they were observed to have changed wetlands.

"Movers"

Of the 146 fish recaptured in 2010, 9 (6.2%) were "movers" since they were observed to have moved beyond their wetland of origin. Moon Island had five movers comprised of three bowfin and two largemouth bass; none of the 57 recaptured pumpkinseeds moved beyond their initial wetland (Table 4). In Tadenac Bay, all four movers were pumpkinseeds (Table 4).

Radio-Tracking

Twelve northern pike were successfully tagged in early May 2011. For simplicity, we will refer to each northern pike in a coded fashion, where northern pike 12 will be referred to as P12. Northern pike length ranged from 563 mm to 962 mm (mean = 750 ± 152 mm) and their weight ranged from 1.1 kg to 6.4 kg (mean = 3.4 ± 2.2 kg; Table 5). With the exception of P19, we were able to determine the sex for all pike; six were determined to be male and the remaining five were female. Age estimates drawn from Wainio (1966, in Scott and

Crossman 1998) suggested that half of the pike were relatively young (2-5 years) and the other half were between 3 and 8 years of age (Table 5).

Following the 2-week recovery period, two northern pike (P17 & P22) were no longer found within the study area; these pike were subsequently found to be alive and living outside Tadenac Bay (Figure 2). Four other northern pike did not spend sufficient time in our study area (P14, P16, P18 & P21; Table 5). The remaining six northern pike were associated with one to five wetlands (Table 6). For each northern pike, with the exception of P15, which was associated with just one wetland, we estimated the minimum, maximum, and average distance between each wetland in which they were observed (Table 6). For all six northern pike, the average distance traveled between adjacent wetlands was 1440 ± 740 m. The maximum observed distance traveled between wetlands for one northern pike was 3900 m (P11) (Figure 4), although it must be acknowledged that P13 moved beyond our study area for several weeks and it is possible that she utilized wetlands that were a greater distance apart (Figure 2).

Discussion

There was a clear distinction between the movement distances of resident and migratory fishes. The majority of the recaptured *resident* fish did not stray beyond their wetland of origin. In contrast, radio-tracking of northern pike demonstrated that the majority of these *migratory* fish moved among multiple wetlands that were on average 1.4 km apart. By studying the distances moved by these two groups of wetland obligate fishes we can begin to develop species-specific conservation strategies.

Resident fishes: movement patterns

During the summer of 2010, only a small percentage (6.2%) of the recaptured resident fishes was observed to leave the wetland where they were initially tagged. Based on our observations during summer and resulting return of most individuals to the same wetland following the winter, we can conclude that majority of fishes did not move among wetlands and therefore, wetland origin is important for many species. This is consistent with previous studies where high site fidelity has been observed for our most common resident, the pumpkinseed (98%; McCairns and Fox 2004).

We documented an average movement distance for pumpkinseeds of 78 m (±51 m). Assuming this to be the greatest distance a pumpkinseed travels, if these straight line distances are squared, they provide a rough estimate for pumpkinseed home range of 0.61ha (±0.26 ha). This coarse estimate is consistent with Fish and Savitz (1983), who found pumpkinseed home ranges to vary from 0.23-1.12 ha (47-105 m straight line distance).

McCairn and Fox (2004) suggested that because of low levels of dispersal between populations, movement of a single individual between two areas could provide important gene flow. We observed a minimal level of dispersal between wetlands by pumpkinseeds, which may provide sufficient gene flow for this species in this region. Concurrent with our fish tagging in summer 2010, we used a 6-m seine net to sample an additional wetland in our Tadenac Bay study region, Tadenac B (TB; data not shown). We did not include the results of this sampling in this study due to unequal effort and different sampling gear, but all fish captured using the seine net were tagged with a unique green VIE tag. In the summer of 2011, we captured one pumpkinseed (length = 115 mm) that migrated over the winter from wetland TB to wetland TE, covering a distance of 1660 m. While we are unable to estimate what proportion of the population would make such a migration, a single individual moving this far may suggest that, for pumpkinseeds, gene flow may exist at a larger spatial scale than would be anticipated.

Despite some observed movement, majority of pumpkinseeds stayed within their original wetland throughout the summer and winter. Thus, wetland conservation that is focused on preserving pumpkinseed habitat should focus at the scale of a single wetland or several closely situated sites (below the 750-m OWES complexing rule; OWES 1993). However, if protection is focused at the scale of the pumpkinseed metacommunity, our observation of movement among wetlands as far as 1660 m would suggest that regional protection of wetlands beyond the 750-m OWES complexing rule is critical for maintaining genetic diversity.

Results of this study reconfirm the fish species groupings outlined by Jude and Pappas (1992). For three of the six commonly occurring fishes categorized as resident wetland species (bluntnose, longear, and rock bass), we did not observe individuals to move beyond the wetland where they were initially tagged. Of the remaining species, the majority of recaptured pumpkinseeds and largemouth bass did not move (96.6% and 83.3%, respectively), confirming that they are, for the most part, wetland residents.

In contrast, all three recaptured bowfin moved to a new wetland. These observations may suggest that while they are wetland residents, bowfins are not as faithful to their wetland of origin as other species. For a large species such as the bowfin, our findings suggest that habitat protection cannot occur solely at the local site level. The three recaptured bowfins moved an average distance of 480 m suggesting that the current 750-m OWES complexing rule may be sufficient. It should be noted however that we tagged a total of 24 bowfin and only recaptured 3. The locations of the remaining bowfin are unknown and it is possible that they moved beyond our study area. While not often a species of interest for recreational fisheries, bowfin could play an important role in energy transfer among wetlands. Radio-tracking of bowfin could provide a more accurate estimate of their home range and movement patterns.

Migratory fish: northern pike tracking

Based on their observed movement among wetlands, northern pike appear to be wetland migratory fish, which is in accordance with Jude and Pappas (1992). While our study did not start until after the northern pike had spawned, most of the northern pike that remained in our study area moved among multiple wetlands. Some of the smaller northern pike, whose ages were estimated to be between 2-5 years (Wainio 1966 in Scott and Crossman 1998), were found predominantly within wetlands (i.e., P11, P12, P15, P20), while larger northern pike (e.g., P13) tended to be found in deeper waters that are adjacent to a wetland (average depths small pike = 2.28 m ± 0.72 m and large pike = 2.82 m ± 0.29 m, Data not shown). Age of sexual maturity for pike has been estimated at between 3-4 years for females and 2-3 years for males (Scott and Crossman 1998). Therefore, a possible explanation for the differences observed between small and large northern pike is that the smaller northern pike are still using wetlands as nursery habitat. If this is the case, it is possible that P19 was a transitional individual, because, while it did spend time near wetlands, it eventually moved from Tadenac Bay into Georgian Bay in a similar fashion as the older northern pike.

Based on our observations of the six northern pike that stayed within our study area, it appears that the majority of northern pike are dependent on more than one wetland to fulfill their life history requirements. Therefore, we believe that the 750 m complexing rule will not adequately protect wetland habitat for a large, mobile predator like the northern pike. To protect younger northern pike (2-5 years), wetlands should instead be grouped at a minimum of 1500 m apart. Based on the precautionary principle and our observation of a maximum movement of 3900 m, a superior complexing distance would be closer to 4000 m.

All pike larger than 700 mm left our study area for at least some period of time and some of them were discovered in open water (Figure 2). Although we did not include this aspect in our study, we believe that the larger northern pike play a pivotal role in linking wetland habitats along the shore, as well as linking nearshore and pelagic environments. A similar linkage has been observed in benthic and pelagic environments and is known to help cycle nutrients between these two distinct environments (Schindler and Scheuerell 2002). Future work should attempt to document the movement of larger northern pike in order to determine if they play a role in this nearshore-offshore coupling.

Summary and Conclusions

Currently, there is an opportunity in Georgian Bay to be proactive rather than reactive towards wetland conservation and by implication, fish habitat protection. Wetlands in the Bay are still relatively pristine, making restoration or remediation unnecessary. Instead, protection alone could help maintain a healthy fishery. Currently, OWES provides the best method for identifying and protecting ecologically important wetlands in Ontario. In order to more accurately delineate coastal wetland complexes, we recommend that the current wetland complexing distance of 750 m (OWES 1993, Wetland Complexes Section Rule 2, pg 19) be increased to between 1500 m and 4000 m. This modification should only be applied to coastal wetlands that are directly connected to the Great Lakes.

Due to the observed global decline in fisheries production, Suski and Cooke (2007) have suggested that current approaches to fisheries management are deficient. A more regional approach to fisheries management and protection can help to maintain important source populations (Hedges et al. 2010). The research presented here has clearly demonstrated the importance of regional species-specific research. Two wetland groups, migratory and resident, show vastly different habitat utilization. Despite these differences, wetland protection tailored towards a migratory top-predator like the northern pike would also provide protection for resident species like pumpkinseeds and rock bass, since it would incorporate their maximum movement distances. This is not to suggest that individual small-wetlands should be dismissed. They play an important role as unique habitat for the resident species that rely on them.

In order to inform conservation decisions, it is critical to understand the home range and movement patterns of individual species. This will ensure adequate protection for all species, since the size of a protected area should be dependent on the scale of the metacommunity (Mouillot 2007). Despite this necessity, regional dispersal processes are not as thoroughly studied as local processes like competition and predation (Cottenie and De Meester 2004; Bouvier et al. 2009). This study is one of the first to document movement patterns for both resident and migratory wetland fish species. Results of this study can be used to update wetland complexing criteria in OWES and can also help to inform conservation strategies in regions with similar fish communities.

Acknowledgements

Funding for this project came from Georgian Bay Forever, Ontario Ministry of Natural Resources through the Canada Ontario Agreement, and the Ontario Chapter of Sierra Club Canada. J. Midwood received funding from the Ontario Graduate Scholarship. We are indebted to numerous summer students and other graduate students in the Chow-Fraser lab who helped to collect field samples, especially Maja Cvetkovic and Chris Biberhofer. Special thank you also to Mary Muter, Denny Hare, and Mark Trudeau; this project would not have happened without their support.

References

Angel JR, Kunkel KE (2010) The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. Journal of Great Lakes Research 36: 51–58.

Bouvier L, Cottenie K, Doka S (2009) Aquatic connectivity and fish metacommunities in wetlands of the lower Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 66: 933–948.

Casselman JM (1974) External sex determination of northern pike, *Esox lucius* Linnaeus. Transactions of the American Fisheries Society 103:343-347.

Chow-Fraser P (2006) Development of the Water Quality Index (WQI) to Assess Effects of Basin-wide Land-use Alteration on Coastal Marshes of the Laurentian Great Lakes. In: Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Authorhouse, Bloomington, pgs. 137–166.

Cook M, Bergersen E (1988) Movements, habitat selection, and activity periods of northern pike in Eleven Mile Reservoir, Colorado. Transactions of the American Fisheries Society 117: 495-502.

Cooke S, Graeb B, Suski C, Ostrand K (2003) Effects of suture material on incision healing, growth and survival of juvenile largemouth bass implanted with miniature radio transmitters: case study of a novice and experienced fish surgeon. Journal of Fish Biology 62: 1366-1380.

Cottenie K , De Meester L (2004) Metacommunity structure: synergy of biotic interactions as selective agents and dispersal as fuel. Ecology 85: 114-119.

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

Cvetkovic M, Wei A, Chow-Fraser P (2010) Relative importance of macrophyte community versus water quality variables for predicting fish assemblages in coastal wetlands of the Laurentian Great Lakes. Journal of Great Lakes Research 36: 64-73.

DeCatanzaro R, Chow-Fraser P, Smith R (2011) Effects of landscape variables and season on reference water chemistry of coastal marshes in eastern Georgian Bay. Canadian Journal of Fisheries and Aquatic Sciences 68: 1009–1023.

Diana J, Mackay W (1977) Movements and habitat preference of northern pike (*Esox lucius*) in Lac Ste. Anne, Alberta. Transactions of the American Fisheries Society 106: 560-565.

Fish PA, Savitz J (1983) Variations in home ranges of largemouth bass, yellow perch, bluegills, and pumpkinseeds in an Illinois lake. Transactions of the American Fisheries Society 112: 147-155.

Franklin AB, Noon BR, George TL (2002) What is habitat fragmentation? Studies in Avian Biology 25: 20–29.

Gotelli N, Taylor CM (1999) Testing metapopulation models with stream-fish assemblages. Evolutionary Ecology Research 1:835-845.

Hedges KJ, Koops MA, Mandrak NE, Johannsson OE (2010) Use of aquatic protected areas in the management of large lakes. Aquatic Ecosystem Health & Management 13: 135–142.

Hoey A, McCormick M (2006) Effects of subcutaneous fluorescent tags on the growth and survival of a newly settled coral reef fish, Pomacentrus amboinensis (Pomacentridae). Proceedings of the 10th International Coral Reef Symposium, 420-424.

Jackson D, Peres-Neto P, Olden J (2001) What controls who is where in freshwater fish communities–the roles of biotic, abiotic, and spatial factors. Canadian Journal Of Fisheries And Aquatic Sciences 58: 157–170.

Jacobus J, Webb PW (2006) Using fish distributions and behavior in patchy habitats to evaluate potential effects of fragmentation on small marsh fishes: a case study. Journal of Great Lakes Research 31: 197–211.

Jude D, Pappas J (1992) Fish utilization of Great Lakes coastal wetlands. Journal of Great Lakes Research 18: 651–672.

Kobler A, Klefoth T, Wolter C, Fredrich F, Arlinghaus R (2008) Contrasting pike (*Esox lucius L.*) movement and habitat choice between summer and winter in a small lake. Hydrobiologia 601: 17–27.

Koed A, Balleby K, Mejlhede P, Aarestrup K (2006b) Annual movement of adult pike (Esox lucius L.) in a lowland river. Ecology of Freshwater Fish 15: 191–199.

Malone J, Forrester G, Steele M (1999) Effects of subcutaneous microtags on the growth, survival, and vulnerability to predation of small reef fishes. Journal of Experimental Marine Biology and Ecology 237: 243–253.

McCairns R, Fox M (2004) Habitat and home range fidelity in a trophically dimorphic pumpkinseed sunfish (*Lepomis gibbosus*) population. Oecologia 140: 271–279.

Midwood JD, Chow-Fraser P (2012) Changes in aquatic vegetation and fish communities following 5 years of sustained low water levels in coastal marshes of eastern Georgian Bay, Lake Huron. Global Change Biology 18: 93–105.

Midwood, J.D., Rokitnicki-Wojcik, D. & Chow-Fraser, P. 2012. Development of an inventory of coastal wetlands for eastern Georgian Bay, Lake Huron. ISRN Ecology vol. 2012, Article ID 950173, 13 pages. doi:10.5402/2012/950173.

Mortsch L, Quinn F (1996) Climate change scenarios for Great Lakes Basin ecosystem studies. Limnology and Oceanography 41: 903–911.

Mouillot D (2007) Niche-assembly vs. dispersal-assembly rules in coastal fish metacommunities: implications for management of biodiversity in brackish lagoons. Journal of Applied Ecology 44:760-767.

Noss R (1992) Issues of scale in conservation biology. In: Conservation Biology: The Theory and Practice of Nature Conservation, Preservation and Management. Chapman & Hall, New York, pgs 240-241.

OMNR 1993. Ontario Wetland Evaluation System. Northern Manual. Omtario Ministry of Natural Resources (OMNR), No.50254.

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

Rogers K, White GC (2007) Analysis of movement and habitat use from telemetry data. Analysis of movement and habitat use from telemetry data. In: Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland, pgs 625-676.

Sale PF (1998) Appropriate spatial scales for studies of reef-fish ecology. Australian Ecology 23: 202–208.

Schindler D, Scheuerell M (2002) Habitat coupling in lake ecosystems. Oikos 98: 177–189.

Scott WB, Crossman EJ (1998) Freshwater Fishes of Canada. Galt House Publications, Oakville, ON.

Seilheimer T, Chow-Fraser P (2007) Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. Journal of Great Lakes Research 33: 154–171.

Sellinger C, Stow C, Lamon E, Qian S (2008) Recent water level declines in the Lake Michigan−Huron System. Environ. Sci. Technol 42: 367–373.

Suski CD, Cooke SJ (2007) Conservation of aquatic resources through the use of freshwater protected areas: opportunities and challenges. Biodiversity Conservation 16:2015-2029.

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

Table 1: Location of each of the wetlands used in this study and the number of fish tagged and recaptured in each site. The wetland specific colour used to tag each fish indicated in brackets next to the wetland code. Wetland MC was sampled during the last 2 weeks of sampling and wetland ME was sampled during the first 2 weeks of sampling. Wetland MF and TB were sampled using a seine during the last 2 weeks of the sampling season.

Region	Year	Wetland Code (Colour)	Latitude	Longitude	Area (ha)	Number Captured Fish	Number Tagged Fish	Number Fish Recaptured	Number of Movers
Moon Island	2010	MA (Red)	45.13784	-80.02494	1.71	508	442	5	1
Moon Island	2010	MB (Green)	45.13549	-80.02578	1.23	619	580	17	0
Moon Island	2010	MC (Orange)	45.13769	-80.02950	0.43	382	354	12	2
Moon Island	2010	MD (Blue)	45.13687	-80.03282	1.29	908	861	37	2
Moon Island	2010	ME (Orange)	45.13998	-80.04012	1.33	204	204	0	0
Tadenac Bay	2010	TA (Red)	45.05228	-79.95976	1.45	702	681	15	0
Tadenac Bay	2010	TC (Yellow)	45.05211	-79.96417	1.96	1087	1045	30	2
Tadenac Bay	2010	TD (Blue)	45.05403	-79.96390	1.95	778	749	19	2
Tadenac Bay	2010	TE (Orange)	45.05587	-79.96938	2.38	638	621	11	0
					Total	5844	5537	146	9
Tadenac Bay	2011	TA	45.05228	-79.95976	1.45	1344	1310†	6	1
Tadenac Bay	2011	TC	45.05211	-79.96417	1.96	404	395†	9	0
Tadenac Bay	2011	TD	45.05403	-79.96390	1.95	381	376†	6	0
Tadenac Bay	2011	TE	45.05587	-79.96938	2.38	1290	1278†	4	1*
					Total	3419	3359†	25	2

† These values do not represent the number of fish tagged in 2011 since no tagging was conducted during these surveys. Instead, these numbers represent the number of captured fish that might have been tagged in 2010 (i.e., excludes brown bullheads and fish smaller than 50 mm).

* This fish was not tagged as part of this study, instead, in a companion study we captured fish in an additional wetland using seining and tagged them with a unique colour. This individual was observed to have travelled 1660 m over the winter. Please see note in the discussion for more information.

Table 2: Summary by fish species of tagged (T) and recaptured (R) individuals for both Tadenac Bay and Moon Island in summer 2010. The proportion of the total catch represented by each individual (P_{Catch}) and the percentage of tagged individuals that was recaptured (P_{Recap}) are also presented.

Species	Common Name	Tadenac Bay				Moon Island			
		T	P_{Catch}	R	P_{Recap}	T	P_{Catch}	R	P_{Recap}
Fundulus diaphanus	Banded killifish	2	0.07	—	—	1	0.04	—	—
Pomoxis nigromaculatus	Black crappie	37	1.20	—	—	3	0.12	—	—
Notropis heterondon	Blackchin shiner	12	0.39	—	—	5	0.20	—	—
Notropis heterolepis	Blacknose shiner	4	0.13	—	—	—	—	—	—
Pimephales notatus	Bluntnose minnow	205	6.62	3	1.46	50	2.05	1	2.00
Amia calva	Bowfin	3	0.10	—	—	24	0.98	3	12.50
Umbra limi	Central mudminnow	1	0.03	—	—	2	0.08	—	—
Semotilus atromaculatus	Creek chub	—	—	—	—	1	0.04	—	—
Notemigonus crysoleucas	Golden shiner	5	0.16	—	—	1	0.04	—	—
Micropterus salmoides	Largemouth bass	296	9.56	7	2.36	302	12.4	5	1.66
Lepomis megalotis	Longear sunfish	172	5.56	5	2.90	—	—	—	—
Lepisosteus osseus	Longnose gar	12	0.39	—	—	2	0.08	—	—
Esox lucius	Northern pike	13	0.42	—	—	14	0.57	—	—
Lepomis gibbosus	Pumpkinseed	2125	68.64	58	2.73	1750	71.69	57	3.26
Ambloplites rupestris	Rock bass	80	2.58	2	2.50	56	2.29	3	5.36
Micropterus dolomieu	Smallmouth bass	1	0.03	—	—	19	0.78	—	—
Perca flavescens	Yellow perch	128	4.13	—	—	211	8.64	2	0.95
	Total	3096		75		2441		71	

Table 3: Estimated average distance moved by fishes in 2010. Fish were tagged with Visible Implant Elastomer Tags. Values were calculated as the shortest distance between the site where a fish was tagged and where it was recaptured. When the initial tagging location was unknown, we took the average distance from all potential tagging locations. No northern pike were recaptured and sufficient yellow perch were not captured to calculate standard deviation.

Species	Total Number Tagged	Total Number Returners/ Movers	Average Distance Traveled (m)	Percentage Moving Wetlands
Bluntnose	255	4	27±17	0.0
Bowfin	27	3	480±206	100.0
Largemouth	598	12	135±214	16.7
Longear	172	5	82±35	0.0
Pumpkinseed	3875	116	78±51	3.4
Rock Bass	136	4	40±29	0.0
Yellow Perch	339	2	29± —	0.0
Northern Pike	27	0	—	—
Other Fishes	108	—	—	—

Table 4: Distances traveled for "movers" from the wetland where they were initial tagged to the wetland where they were recaptured. Distance was estimated as the shortest straight-line distance between these two points.

Wetland Group	Species	Length (mm)	Initial Wetland	Recapture Wetland	Distance (m)
Moon Island	Bowfin	265	MA	MC	378
Moon Island	Bowfin	510	MD	MA	718
Moon Island	Bowfin	345	MD	MC	345
Moon Island	Largemouth bass	61	MC	MD	341
Moon Island	Largemouth bass	62	MA	MD	755
Tadenac Bay	Pumpkinseed	83	TC	TD	247
Tadenac Bay	Pumpkinseed	73	TC	TD	283
Tadenac Bay	Pumpkinseed	78	TD	TC	206
Tadenac Bay	Pumpkinseed	92	TD	TC	269

Table 5: Length, weight, sex, and estimated age (from Scott & Crossman 1998) for the 12 northern pike tracked in this study. The location of the tagging sites as well as the study area can be found in Figure 3. The total numbers of observations for each northern pike are listed. Tracking window refers to the time, in days, between the first and last observation. Northern pike that remained in the study area are identified. It is unknown if northern pike number 12 is still in our study area because it has not be observed since July 21, 2011.

Pike Tag Code	Site Tagged	Length (mm)	Weight (kg)	Sex	Estimated Age	Tracking Window (days)	Total Observations	In Study Area (Y/N)
11	1	632	1.5	M	2-5	93	52	Y
12	1	583	1.2	M	2-5	60	39	unknown
13	1	962	6.0	F	6-8	93	47	Y
14	1	773	3.2	M	3-8	38	29	N
15	3	563	1.2	M	2-5	93	52	Y
16	1	912	6.4	F	5-8	6	7	N
17	2	817	4.1	F	4-8	0	2	N
18	4	913	5.2	F	4-8	44	20	N
19	1	574	1.1	unknown	2-5	87	48	N
20	1	620	1.5	M	2-5	90	39	Y
21	1	916	6.4	F	4-8	18	10	N
22	2	729	2.6	M	2-7	0	2	N
	Average	749.5±152.4	3.4±2.2					

Table 6: Number of wetlands associated with each northern pike. Also included is a summary of the distances among wetlands for each pike. With the exception of northern pike number 15 that used only one wetland, all northern pike used wetlands that were greater than 750 m apart.

Pike Tag Code	Length (mm)	Number Wetlands	Avg. Dist. Btw. Wetlands (km)	Min. Dist. Btw. Wetlands (km)	Max Dist. Btw. Wetlands (km)
11	632	5	2.37	0.14	3.90
12	583	3	0.75	0.23	1.03
13	962	5	2.12	0.33	3.77
15	563	1	N/A	N/A	N/A
19	574	4	1.05	0.41	1.90
20	620	4	0.93	0.19	1.20
		Average	1.44±0.74	0.26±0.11	2.36±1.39

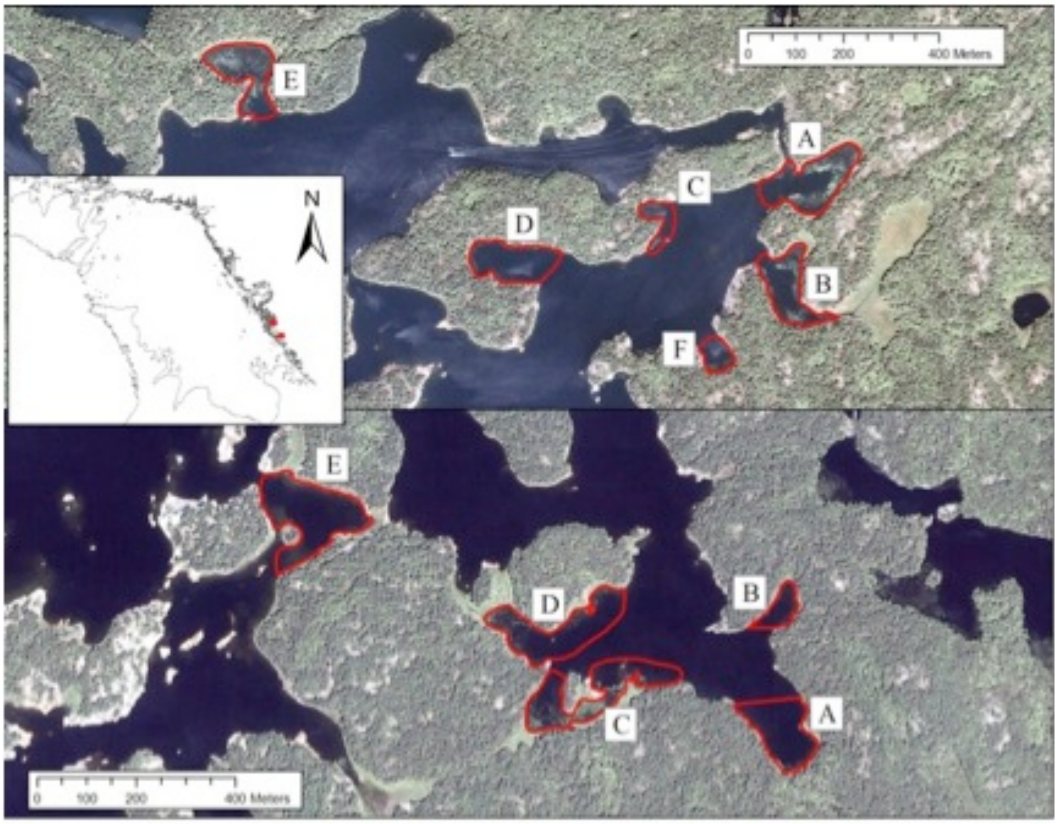


Figure 1: Location of study sites in Moon Island (top photo) and Tadenac Bay (bottom photo). In Moon Island, wetlands A, B, and D were sampled throughout summer. Wetland E was sampled during weeks 1 and 2 and wetland C was sampled during weeks 3 and 4. Wetland F was surveyed using a seine during weeks 3 and 4 but data collected there were not discussed fully in this study. In Tadenac Bay, wetlands, A, C, D, and E were sampled throughout the summer and wetland B was sampled using a seine during weeks 3 and 4 but data collected there were not discussed fully in this study, with the exception of the one pumpkinseed that moved from wetland B to wetland E between the summers of 2010 and 2011.

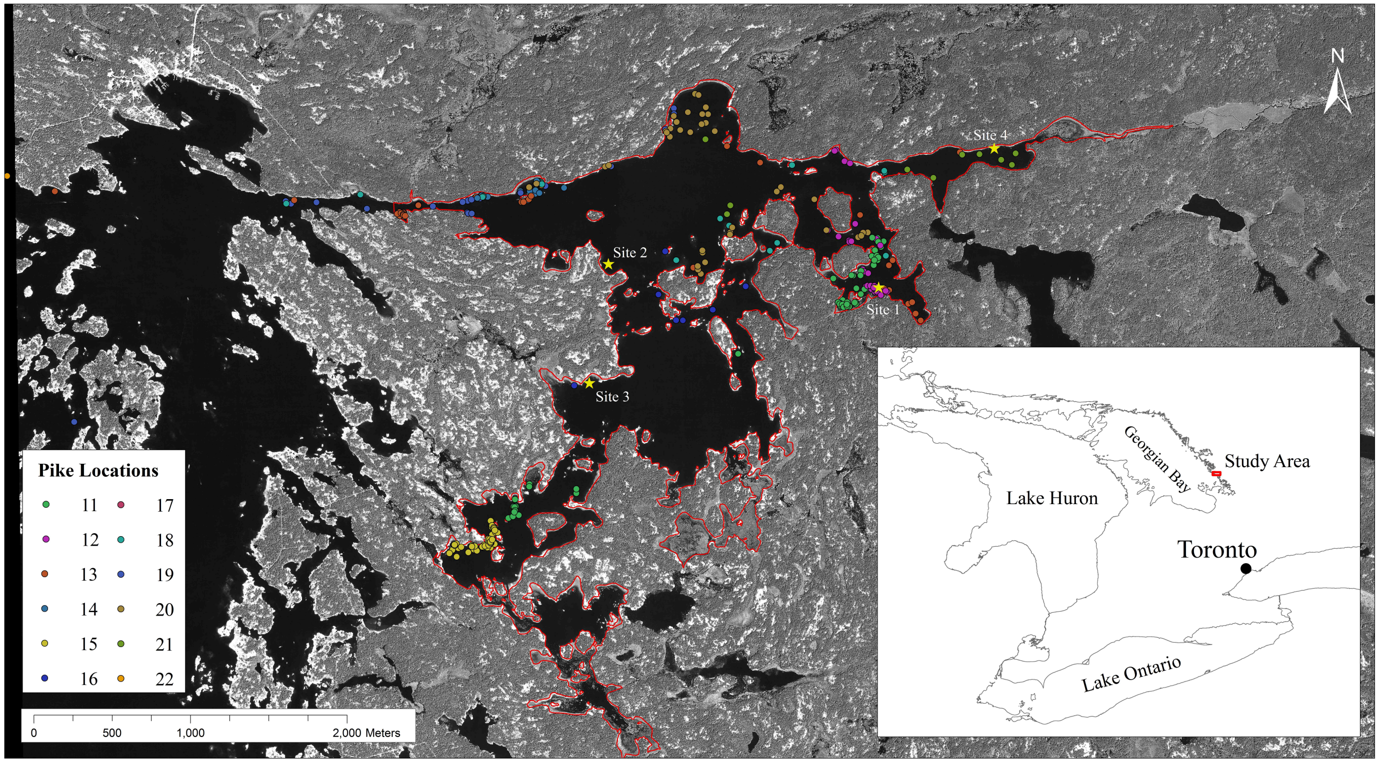


Figure 2: Location of pike throughout Tadenac Bay. The red line represents the boundaries of our study area. A yellow star represents the four locations where pike were initially captured. We opportunistically sampled outside of our study area and the position where pike were found beyond our study area can be seen to the west of Tadenac Bay.

Figure 3: Example of tagged pumpkinseed (*Lepomis gibbosus*). This fish was first caught in the yellow wetland and then recaptured in the blue wetland. Both captures occurred during the same week since the tags are found on the same cheek.

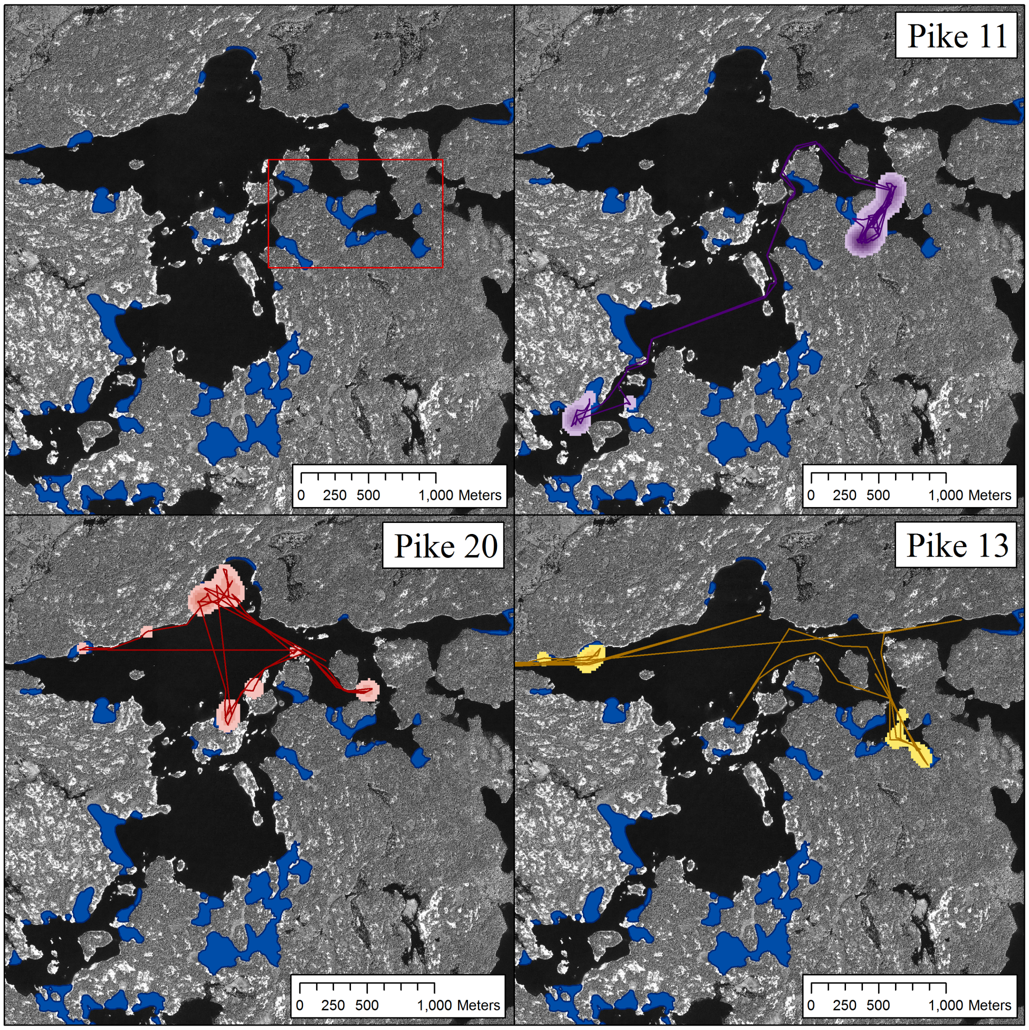


Figure 4: Example of observed northern pike movements from May 24th, 2011 until August 24th, 2011. The red bounding box represent the location of the tagging study conducted in the summer of 2010. Movement patterns for P13 clearly show it exited Tadenac Bay on multiple occasions. The movements of this northern pike beyond our study area are unknown.

GENERAL DISCUSSION

Thesis Summary

The primary objectives of this thesis were to provide a better understanding of the dynamics of fish habitat in response to changing water levels and to determine the scale of habitat selection by fishes. To this end, there are three primary focuses: ecosystem monitoring, ecosystem modeling, and broadly, evaluating ecological questions (Figure 1). In Chapters 1 and 2, we develop tools that can be used to monitor coastal wetland habitat. Next, in Chapter 3, we model the response of submerged vegetation to declining water levels. In Chapter 4, we explain how water levels impact aquatic vegetation and how these impacts may potentially impact the fish community. Finally, in Chapter 5, I discuss how one of our monitoring tools, the McMaster Coastal Wetland Inventory (MCWI), and changes in fish communities led us to study the movement of fishes among coastal wetlands and the implications of our findings.

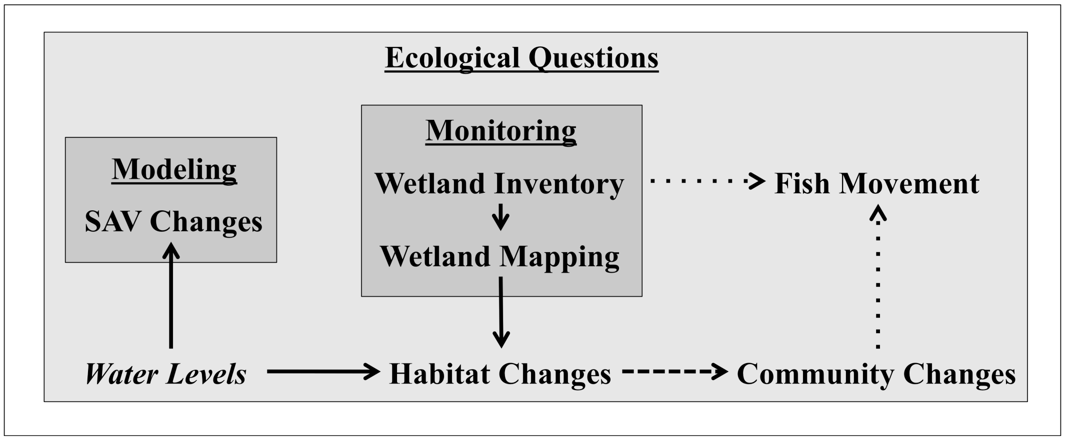


Figure 1: Conceptual diagram linking the three primary focuses of the research presented in this thesis. Ecological questions are a component of all of the research. The solid lines imply a direct influence, i.e. Water Levels impact SAV and Habitat; a Wetland Inventory is necessary for Mapping, which in turn is a crucial component of assessing Habitat Change. The dashed line between Habitat and Community Changes refers to the implied connection between the two. Finally, the dotted lines show how other research raised questions that informed our study of Fish Movements.

Specifically, in Chapter 1 we create the most comprehensive inventory of Georgian Bay coastal wetlands to date (McMaster Coastal Wetland Inventory; MCWI). Researchers can use this inventory to randomly select a statistically valid subset of wetlands, allowing their findings to be applied across the entire region. Based on this inventory, the majority of coastal wetlands in Georgian Bay are small (<2 ha, 89.0%). Even though they are small, it is still important to protect these wetlands because they provide critical habitat for a diverse array of species.

In Chapter 2, we develop an object-based approach to map aquatic vegetation in coastal wetlands. This is one of the first applications of this technique in a wetland setting and allows us to develop an accurate and broadly applicable classification method. By applying the classification to 50 wetlands in southeastern Georgian Bay, we were able to establish a baseline of habitat composition for 50 wetlands. The same techniques in this chapter can be applied to a wide variety of wetland ground cover types, provided adequate ground-truth samples are acquired.

In Chapter 3, we first evaluate a published equation that predicts the maximum depth of plant colonization (Z_{max}), concluding that it consistently underestimates Z_{max} values for Georgian Bay wetlands. We then develop a rule-based model to map potential submerged aquatic vegetation (SAV) habitat that can also be used to predict how this habitat will respond to changing water levels. When applied to six wetlands there is a variable response that is largely dependent on the geomorphology of the wetland. Generally, as water levels decline, there is a decrease in the amount of potential SAV habitat.

In Chapter 4, we conduct a post-classification change detection analysis to determine the impact of low-water levels on coastal wetland vegetation. Since wetlands are naturally in a state of perpetual succession, the loss of the external stress from fluctuating water levels, allows terrestrial vegetation to thrive. There is also a shift from sparse to dense floating vegetation. Sparse floating vegetation provides better habitat for fishes because it grows jointly with SAV, thereby increasing the overall structure of the habitat. In general, sustained low-water levels lead to a net loss of fish habitat. This loss of habitat likely explains both the decline in fish species richness and increase in homogeneity of the fish community that occurred during the same time period.

In Chapter 5, we document the movement of both resident fishes and a migratory fish (northern pike) among coastal wetlands. While majority of resident fishes appear to be dependent on a single wetland, there is some of evidence of movement among sites by a small subset of the population (~6%). While small, this subset of the population may represent an important genetic link among these wetland fish communities. In contrast, northern pike use wetlands that are as far as 3.9 km away, traveling on average 1.4 km to use marsh habitat. Movement among coastal wetlands is therefore a common occurrence in migratory fish species such as northern pike.

In summary, the results of this thesis clearly show that the numerous coastal wetlands in eastern and northern Georgian Bay, while currently pristine, are under threat from declining water levels. Adoption of the recommendations outlined herein and an exploration of some the future research that is suggested below will help to preserve this habitat and maintain the natural fish communities that dwell within.

Recommendations

Based on some of the major findings documented in this thesis, below I propose some potential recommendations that will help in the conservation of coastal wetlands.

1. Under the Ontario Wetlands Evaluation System (OWES), wetlands < 2 ha in size cannot be evaluated unless they are part of a larger complex of wetlands located within 750 m of each other (OMNR 1993). Based on the observed movement of northern pike among wetlands that were on average 1.4 km apart, I recommend that the OWES complexing rules for coastal wetlands be increased to at least this distance, if not the maximum distance documented at 3.9 km. To demonstrate the application of this rule, wetlands delineated in the MCWI should be complexed based on these rules.

2. Wetland area is often used as an indicator of regional wetland health. Once inventories are created, changes in wetland area can represent either an improvement (increasing total area) or a cause for concern (decreasing total area). At the State of the Lakes Ecosystem Conference (SOLEC) in 1998, wetland area was selected as the most important indicator for wetland monitoring purposes. For Georgian Bay, our findings would suggest that area is not the best metric since most wetlands are relatively

small, yet numerous, when compared with wetlands in other parts of the Great Lakes. Using area as the main metric would severely undervalue the importance of coastal wetlands in Georgian Bay, despite their ecological integrity. I would therefore caution against the use of wetland area as a sole or main indicator of wetland health and instead, recommend that some of the other indicators recommended by the 1998 SOLEC report (e.g., Fish Community Health, Amphibian Diversity, Invertebrate Community Health, Habitat Adjacent to Coastal Wetlands, etc.) be used in conjunction with area to better reflect the relatively pristine nature of Georgian Bays small, coastal wetlands.

3. An important component of Chapter 2 was the application of our classification method. Far too often researchers are focused on the development of a tool, resulting in only limited application. In this study, we not only provided a tool for mapping vegetation, we also applied it to 50 wetlands. In the future, I would encourage the same sort of action whenever a novel classification method is developed. This ensures that it is not only used but that habitat information is also collected over a large area, thus capturing a snapshot of the habitat distribution. If it was important enough to classify, it is likely also important enough to map.

4. For the SAV rule-based model we developed, the main limitation for its regional application is the lack of topographic and bathymetric data for most of the Georgian Bay shoreline. Currently, the best way to fill in this gap is to acquire Light Detection and Ranging (LiDAR) data for the Georgian Bay coastline. This would not only allow wide-scale application of our model, it would also allow us to model wetland habitat at historic high-water levels. Since acquisition of this imagery can be quite expensive, I would recommend a pilot study to explore its application in the context of Georgian Bay .

5. Based on the observed loss of fish habitat following sustained low water levels and the commensurate change in the fish community, I strongly recommend that Great Lakes water level management decisions focus on restoring natural water level cycles throughout the basin, especially in Lake Michigan-Huron. Without a return to a more natural water cycle, the changes presented in this study will only continue, since it is disturbance that is able to maintain naturally high levels of biodiversity.

6. Unfortunately, in this thesis we could not concretely demonstrate a causal relationship between the changes to both fish habitat and the fish community following sustained low water levels. While we were fortunate

to have access to historical data that overlapped with our satellite imagery that allowed us to imply a relationship, the ideal situation would have been to have fish habitat and community data collected concurrently. This highlights a long recognized shortfall in many studies that occur in natural systems, namely the absence of complete, long-term databases. It is therefore essential, given the forecasted changes in Georgian Bay water levels and likely increases in regional development, to establish long-term wetland monitoring sites. These sites should be visited on a regular basis, ensuring that sampling dates are as consistent as possible. A wide range of variables should be collected, including community data for fish, invertebrates, amphibians, and reptiles. Collecting environmental data (water/soil chemistry, macrophyte density and composition, etc.) and ground-control points, will ensure that the impact of future changes can be documented with confidence. The MCWI could be used to select an appropriate subsample of wetlands from areas that are currently pristine, developed, threatened by development, and threatened by low-water levels (i.e. potential for hydrologic disconnection).

Future Work

The suggestions below represent either an area that, given time, I would have liked to explore more, or novel questions that became apparent through this research.

1. We presented a series of conceptual responses of potential SAV habitat to declining water levels. These were largely driven by the geomorphology of the wetland. Based on the four different scenarios, wetlands delineated in the MCWI could be categorized as one of the four different types. This would provide a rough estimate of how many wetlands in Georgian Bay will lose or gain SAV habitat as a result of changing water levels.

2. While the IKONOS satellite imagery used in these studies is relatively high-resolution, we were only able to map a few wetland cover types. Recent advances in satellite and aerial image acquisition have increased pixel resolution beyond the 1.0-m^2 of the IKONOS imagery used in our study (ex. GeoEye-1 resolution - 0.5-m^2). This finer-scale resolution could allow future studies to map more detailed vegetation groupings, possibly allowing for the classification of species of interest. This has important implications for Georgian Bay, where low water levels may promote the growth of the invasive species *Phragmites australis*. Work should be undertaken to identify the unique spectral signature of this aggressive

species as early detection is still the best control mechanism, and in the numerous and widely distributed Georgian Bay wetlands remote sensing will provide the only feasible means of keeping it in check.

3. A question that is invariably asked when we link wetland habitat losses to lower water levels is: 'Will new wetlands not form elsewhere?". Factors that determine where new wetlands can form include both the availability of appropriate embayments (affording some level of protection from wind/wave action) and sufficient time for organic substrates to accumulate. While the rate of accruement and specific morphology of potential wetland embayments are not explicitly known, acquisition of LiDAR imagery could help to identify new areas that may support the development of a coastal wetland. Should water levels continue to decline, these "potential" wetlands could become conservation priorities.

4. In our analysis of fish community changes, we were only able to compare early and late community data. Therefore, I encourage future studies to attempt to corroborate our findings in other wetlands. While not discussed in this thesis, fish community data were collected in the summer of 2011 in northern Georgian Bay. When these data were compared with data collected from 5-6 years earlier, there were no significant changes found in

the communities. This may suggest that either our “earlier” time period was after community changes had occurred or that the wetlands in northern Georgian Bay respond differently than those in southern and eastern Georgian Bay. In our observations, dominant vegetation is different in the northern Georgian Bay wetlands (more bulrushes and wild rice) and many of the wetlands are also less prone to isolation (more exposed). It is therefore possible that while low-water levels have affected the more sheltered wetlands of southern Georgian Bay, habitat in the north has not been impacted to the same extent. This hypothesis should be tested in order to explain the observed differences between fish communities in southern and northern Georgian Bay.

5. From my work it is evident that wetlands, even ones seemingly disconnected, are in fact linked by a top-predator and, in rare instances, even by resident fishes. The bulk of the northern pike data discussed in Chapter 5 were based on movements by 6 of the 12 fish that were tagged. This begs the question of where the remaining 6 northern pike went. Through opportunistic sampling in the summer, we discovered some of these larger northern pike around offshore islands in a more pelagic environment. Furthermore, while the data are not presented in this thesis, we tracked northern pike in late November 2011 and found that some of

the larger northern pike that were offshore in the summer had returned to Tadenac Bay. This suggests that these northern pike may use the nearshore environment for spawning, move to a more pelagic environment to feed throughout the summer, and return in the winter to more protected nearshore areas. By continuing to track these 12 northern pike over the next two years, a better understanding of their movement patterns among seasons can be developed and any evidence for site fidelity can be documented.

6. For tracking the northern pike, increased effort afforded to each individual would generate a better idea of individual movement patterns over a short time period. This will provide a more detailed understanding of daily movement of fishes in and among coastal areas.

7. In a similar fashion, the availability of smaller radio rags will allow movement patterns of small-bodied resident fishes to be documented in more detail. These types of data can be used to corroborate the findings of our mark-recapture study.

8. One lingering question from our mark-recapture study that was well beyond the scope of this thesis was, “How many fish must move between

wetlands to maintain genetic diversity and prevent a founder effect?". Genetic studies have been conducted on muskellunge (*Esox masquinongy*) in Georgian Bay, and by isolating specific genetic markers (microsatellites), these studies documented regional populations that were quite distinct (Chris Wilson, unpub. data). A similar study could be employed at a smaller scale by focusing on a species such as the pumpkinseed. It would be of great interest to determine how related sunfish in wetlands are compared with sunfish captured in other regions of Georgian Bay.

APPENDICIES

Appendix 1: Summary of total ground-truth samples collected at 12 wetlands for (1) PTC Development and (2) PTC Validation. Ground-truth data collected correspond to 6 wetland classes: meadow (M), high-density floating (HD), low-density floating (LD), emergent (E), rock (R) and water (W). In Roseborough Bay, a map containing class-distribution information was created in the field. "N/A" means that no ground truth samples had been collected because of absence of that class at the site.

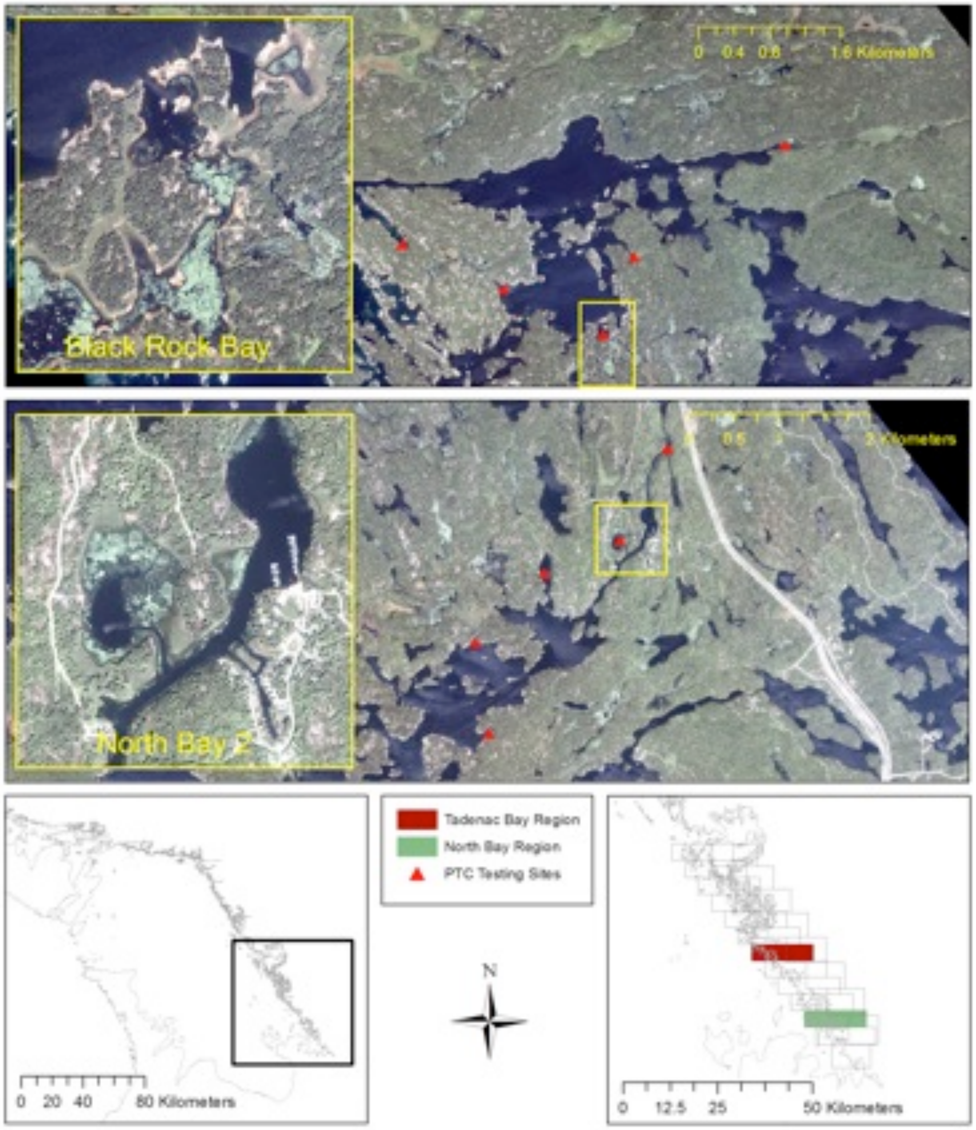
					Number of Ground Truth Samples						
Wetland	**Region**	**Latitude**	**Longitude**	**Purpose**	**M**	**HD**	**LD**	**E**	**R**	**W**	**Total**
Black Rock Bay †	Tadenac Bay	45.04237	-79.97216	1	8	12	8	14	5	1	48
Coffin Rock †	Tadenac Bay	45.04802	-79.98745	1	4	5	4	10	4	2	29
Garden Channel *	Sans Souci	45.18628	-80.12147	1	0	5	3	5	2	0	15
North Bay 1 †	North Bay	44.89638	-79.79377	1	7	5	6	9	2	1	30
Oak Bay *	Severn Sound	44.79630	-79.73158	1	7	11	9	18	2	1	48
			Sub-total	1	26	38	30	56	15	5	170
Alexander Bay †	Tadenac Bay	45.05309	-80.00310	2	0	N/A	1	11	0	0	12
Miners Creek †	Tadenac Bay	45.06040	-79.95571	2	1	8	1	6	N/A	3	19
North Bay 2 †	North Bay	44.54074	-79.47043	2	0	2	2	2	0	6	12
North Bay 3 †	North Bay	44.89118	-79.80360	2	5	4	2	8	0	0	19
North Bay 5 †	North Bay	44.88156	-79.80388	2	5	4	4	9	0	0	22
North Bay River †	North Bay	44.91627	-79.77777	2	6	16	8	16	1	0	47
Roseborough Bay *	Go Home Bay	44.99491	-79.92316	2	--	--	--	--	--	--	0
South Bay 2 †	North Bay	44.87332	-79.77232	2	0	7	6	2	0	0	15
Thunder Bay †	Tadenac Bay	45.05139	-79.96998	2	4	3	3	6	1	0	17
Treasure Bay †	Severn Sound	44.86854	-79.86049	2	8	5	3	16	1	3	36
West Black Rock Bay †	Tadenac Bay	45.04181	-79.97855	2	2	6	2	3	3	0	16
			Sub-total	2	31	55	32	79	6	12	215

* - sites where field map was manually delineated.
† - sites where field map was derived using a printed version of the IKONOS satellite image.

Appendix 2: Location of the 50 wetlands randomly selected from all 144 wetlands (size > 2ha) located in the southern half of Georgian Bay (Severn Sound to Parry Sound). Areal coverage of four vegetation classes (meadow (M), emergent (E), high-density floating (HD) and low-density floating (LD)) as well as rock (R) and water (W) are provided. A conservative estimate of total visible fish habitat (VFH) was calculated by summing the areal cover of E, HD and LD.

Wetland Region	Latitude	Longitude	M (m^2)	HD (m^2)	LD (m^2)	E (m^2)	R (m^2)	W (m^2)	VFH (m^2)	Total Area (m^2)
Beausoleil Is.	44.88281	-79.86917	1035.6	237.7	8241.8	7950.4	637.0	3119.2	16429.9	21221.7
Beausoleil Is.	44.89076	-79.88243	494.7	2914.3	6592.3	1210.2	196.9	9757.1	10716.8	21165.5
Beausoleil Is.	44.87211	-79.85334	135.0	66.0	2463.0	731.0	11.0	8828.0	3260.0	12234.0
Big Is.	45.00818	-79.94119	389.1	153.5	2800.3	823.8	2.0	31116.6	3777.6	35285.3
Bone Is.	44.93507	-79.87339	225.7	65.5	2547.0	526.9	257.5	17572.1	3139.4	21194.7
Clifton Bay	44.96043	-79.87796	1396.0	193.0	1656.9	4750.3	11821.2	7144.4	6600.2	26961.8
Cognashene	44.93507	-79.87339	1011.3	4238.0	10989.6	4331.7	62.9	27370.3	19559.3	48003.8
Cognashene	44.95361	-79.91339	189.4	2588.0	8492.3	941.9	814.6	38716.1	12022.2	51742.3
Garden Channel	45.19124	-80.12014	3368.9	1716.0	5934.9	7383.9	125.4	26397.6	15034.8	44926.7
Go Home Bay	45.00579	-79.91308	753.0	464.4	4680.0	784.5	967.9	66633.8	5928.9	74283.6
Go Home Bay	44.98486	-79.92137	332.2	1990.5	15388.7	3217.7	659.5	18793.4	20596.9	40382.0
Gooseneck Bay	45.20954	-80.11172	305.3	141.0	2455.3	527.9	72.6	21544.7	3124.2	25046.8
Moon Bay	45.11505	-80.01757	475.3	690.5	10995.4	1877.7	23.5	60496.0	13563.6	74558.4
Moon River	45.10709	-79.93030	384.0	2740.0	36488.8	865.5	72.8	52180.0	40094.3	92731.1
Musky Bay	44.81539	-79.78990	497.2	124.9	2878.2	581.1	507.5	16735.1	3584.2	21324.0
Musky Bay	44.81158	-79.78080	19926.7	12951.4	38072.3	10666.4	2209.3	95317.2	61690.1	179143.3
North Bay	44.85584	-79.81714	369.2	2093.7	5893.7	542.8	1121.8	61632.3	8530.2	71653.5
North Bay	44.89091	-79.80318	590.9	1042.9	4409.0	1179.4	382.0	29552.0	6631.3	37156.2
Oak Bay	44.79757	-79.73432	5349.9	10985.5	47372.4	4900.2	488.4	430333.3	63258.1	499429.7
Parry Is.	45.23759	-80.17095	422.5	3946.0	8400.4	2906.1	40.2	88219.7	15252.5	103934.9
Parry Is.	45.25317	-80.11600	632.2	25.3	1003.4	767.9	38.1	30249.4	1796.6	32716.3
Parry Is.	45.28117	-80.12285	376.7	12.0	956.4	1090.2	2.9	19422.6	2058.6	21860.8
Parry Is.	45.23093	-80.08945	223.6	5.0	1370.6	554.0	18.3	20242.0	1929.6	22413.5

Parry Is.	45.27255	-80.07470	8.0	171.0	1090.1	205.9	25.0	35613.7	1467.0	37113.7
Port Rawson	45.19398	-80.02767	1069.4	1011.9	6906.8	3257.4	42.3	25118.0	11176.1	37405.8
Port Rawson	45.19653	-80.02255	387.8	1040.0	5591.1	4504.5	2.0	30037.7	11135.6	41563.1
Quarry Is.	44.82994	-79.80867	2731.2	770.2	11270.6	3818.8	161.3	83316.2	15859.6	102068.3
Roberts Is.	44.86454	-79.82977	222.8	260.4	1561.2	760.8	238.0	19880.8	2582.4	22924.0
Roberts Is.	44.86260	-79.83793	4956.0	12138.0	12686.0	2358.8	281.0	41625.0	27182.8	74044.8
Sandy Is.	45.27732	-80.26603	419.4	395.5	1971.4	1149.1	526.9	28437.9	3516.0	32900.2
Sandy Is.	45.28024	-80.25684	1760.5	109.0	5133.1	3590.1	1980.6	10378.1	8832.2	22951.4
Sans Souci	45.17458	-80.05957	943.0	206.0	7965.5	2244.2	408.5	52633.4	10415.7	64400.6
Sans Souci	45.18575	-80.06551	4245.7	3437.5	13234.0	5106.4	57.2	45384.0	21777.9	71464.8
Sans Souci	45.16560	-80.06176	1692.4	4222.5	5803.9	3791.8	36.4	21153.9	13818.2	36700.9
Sans Souci	45.21712	-80.07511	83.9	82.5	2292.7	1159.4	4.6	17895.9	3534.6	21519.0
South Bay	44.86453	-79.78574	6551.6	11703.0	24324.0	4628.0	1512.0	70814.0	40655.0	119532.6
Tadenac Bay	45.04222	-79.99020	555.3	336.0	3988.8	1871.0	641.9	12892.9	6195.8	20285.9
Tadenac Bay	45.04121	-79.97393	619.9	2084.0	2752.7	2958.9	396.8	16925.3	7795.6	25737.6
Tadenac Bay	45.02999	-79.98538	3778.7	15350.1	53251.4	22852.9	2915.7	93129.0	91454.4	191277.8
Twelve Mile Bay	45.08832	-80.04240	972.1	846.0	6216.9	3822.1	469.6	49162.1	10885.0	61488.8
Twelve Mile Bay	45.08295	-79.92893	6630.0	3556.5	20423.0	7854.0	120.5	14357.6	31833.5	52941.6
Twelve Mile Bay	45.09095	-80.06656	967.9	2176.1	4981.5	4655.2	44.7	27633.8	11812.8	40459.2
Twelve Mile Bay	45.08737	-80.08322	86.4	174.0	3320.9	476.6	728.4	27369.8	3971.5	32156.1
Wilson's Bay	44.99545	-79.95274	128.9	0.0	2588.3	298.7	127.7	30304.9	2887.0	33448.5
Woods Bay	45.15175	-79.99787	547.4	71.5	2741.8	1377.7	9.7	18272.0	4191.0	23020.1
Woods Bay	45.13109	-79.99041	8611.4	16690.0	57718.9	9475.6	1962.1	158948.0	83884.5	253406.0
Woods Bay	45.12958	-79.98349	527.2	842.0	10964.8	4491.1	157.3	28236.0	16297.9	45218.4
Woods Bay	45.12723	-79.97611	84.0	1848.5	18659.9	154.6	205.2	51306.0	20663.0	72258.2
Woods Bay	45.10880	-79.95341	883.0	182.5	12055.9	1885.5	217.0	108825.0	14123.9	124048.9
Woods Bay	45.13398	-79.99590	92.9	10.5	1834.4	390.8	168.9	46739.6	2235.7	49237.1
Average			**1768.8**	**2582.0**	**10628.2**	**3165.0**	**679.5**	**46955.3**	**16375.3**	**65778.9**



Appendix 3: Example of two IKONOS images covering the regions of Tadenac Bay (dark red) and North Bay (green). The locations of wetlands in each image are marked by the red triangles. A marina complex can be seen in the North Bay 2 image (zoom).

Appendix 4: Number of sample objects (SOs) collected for each of the 6 wetlands classes (meadow (M), high-density floating (HD), low-density floating (LD), emergent (E), rock (R) and water (W)) in wetlands used in the creation and validation of the Process Tree Classification (PTC). Values in the brackets represent the average size of the sample objects (m^2). Sample objects were chosen based on ground truth samples and/or field-derived maps. "N/A" means that the class in question had not been sampled in that wetland.

		Samples Selected in Definiens						
Wetland	**Purpose**	**M**	**HD**	**LD**	**E**	**R**	**W**	**Total**
Black Rock Bay	PTC Development	51(118)	44(74)	31(126)	42(106)	35(23)	34(367)	237
Coffin Rock	PTC Development	40(123)	36(32)	25(48)	35(86)	28(23)	27(399)	191
Garden Channel	PTC Development	23(102)	20(46)	14(73)	19(63)	16(20)	15(323)	107
North Bay 1	PTC Development	35(123)	31(49)	22(96)	29(90)	24(24)	23(417)	164
Oak Bay	PTC Development	81(87)	71(36)	49(85)	67(84)	55(22)	54(393)	377
Black Rock Bay *	PTC Development	20(122)	92(39)	16(80)	37(80)	70(21)	31(478)	266
	Total Sample Objects	**250(111)**	**294(40)**	**157(76)**	**229(81)**	**228(22)**	**184(402)**	**1342**
Alexander Bay	PTC Validation	11(90)	N/A	2(71)	5(127)	6(24)	9(518)	33
Miners Creek	PTC Validation	16(120)	20(56)	11(86)	11(80)	N/A	9(434)	67
North Bay 2	PTC Validation	5(112)	15(57)	10(94)	4(98)	3(20)	12(420)	49
North Bay 3	PTC Validation	2(139)	4(57)	3(93)	3(99)	5(28)	6(422)	23
North Bay 5	PTC Validation	3(118)	7(32)	4(100)	4(74)	5(22)	4(410)	23
North Bay River	PTC Validation	3(131)	14(54)	7(96)	4(113)	N/A	12(422)	40
Roseborough Bay	PTC Validation	16(90)	16(47)	8(57)	9(67)	6(25)	21(262)	76
South Bay 2	PTC Validation	11(114)	8(45)	7(88)	6(58)	5(25)	7(383)	39
Thunder Bay	PTC Validation	11(165)	N/A	7(115)	5(117)	6(20)	8(425)	37
Treasure Bay	PTC Validation	15(53)	14(16)	15(84)	10(88)	4(18)	15(352)	73
West Black Rock Bay	PTC Validation	2(156)	14(7)	9(25)	6(45)	8(15)	4(679)	43
	Total Sample Objects	**95(117)**	**99(41)**	**91(83)**	**60(88)**	**64(22)**	**94(430)**	**503**

* - This second set of samples was collected in Black Rock Bay to refine the threshold values for the 4 vegetation classes.