

**FORECASTING THE RESPONSE OF COASTAL WETLANDS TO DECLINING
WATER LEVELS AND ENVIRONMENTAL DISTURBANCES
IN THE GREAT LAKES**

BY

ANHUA WEI, B.Sc., M.Sc.

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AUTHOR: Anhua Wei, B.Sc., M.Sc.

SUPERVISOR: Dr. Patricia Chow-Fraser

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ABSTRACT

The overall purpose of this study was to examine the impact of increasing human-induced and natural stresses on the distribution of aquatic vegetation and fish along the Great Lakes shoreline.

The first part of the dissertation proposes a predictive model to examine the effect of long-term water level fluctuations on the emergent plants in Lake Ontario coastal wetlands while taking into account other factors such as urbanization and invasive species. Starting with Cootes Paradise Marsh, a highly urbanized freshwater marsh at the west end of Lake Ontario, a new approach based on a digital elevation model is proposed and compared with the traditional water-level model. The results indicate that the new approach is better than the traditional water level model. To verify the effectiveness of the new approach, ten coastal wetlands from eastern Lake Ontario are tested. The results confirm that the new approach provides an effective way to determine the effect of the hydrological disturbance on emergent plants even when the target wetlands have the same water level.

The second part of the dissertation addresses response of fish and explores the use of high resolution satellite imagery to quantify the supply of fish habitat and wave disturbance in coastal wetlands. The results show that (1) the Great Lakes fish community utilizes wetlands disproportionately to their availability and the distribution of wetland-associated taxa is influenced by wetland type; (2) the high resolution IKONOS

imagery can be used effectively to monitor the change in aquatic vegetation and thus track alterations in fish habitat in Great Lakes coastal marshes; and (3) aquatic vegetation in coastal wetlands is affected by exposure disturbance that can be attributed to the lake and any disturbance originating from inside the wetland and therefore, the effect of exposure and geomorphology should be partitioned out when examining the effects of anthropogenic disturbance on submersed aquatic vegetation.

PREFACE

The following Ph.D. thesis consists of manuscripts that are published or submitted for publication in peer-review journals. Chapters 1, 2, 4, and 5 have been published. The remaining chapters are being reviewed for publication in peer review scientific journals.

The following are the proper citations for these papers, including co-authorship. I wrote all the manuscripts. My supervisor, Dr. Patricia Chow-Fraser was co-author on all the manuscripts. Chapter 4 was also co-authored by Dr. Dennis Albert.

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

Wei, A. and Chow-Fraser, P. 2007. Development of an index to account for degree of exposure and wetland geomorphology on submerged aquatic vegetation of Georgian Bay coastal wetlands.

Wei, A. and Chow-Fraser, P. 2007. A regional model of emergent vegetation in response to water level fluctuations in Lake Ontario.

Wei, A. and Chow-Fraser, P. 2006. Synergistic impact of water-level fluctuation and invasion of *Glyceria* on *Typha* in a freshwater marsh of Lake Ontario. *Aquatic Botany* 84: 63-69.

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

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永远感激父母的呵护养育恩、远行忆念恩、深加体恤恩和究竟怜愍恩。愿令一切学子亦常念父母恩。

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GENERAL INTRODUCTION

Great Lakes coastal wetlands

Coastal wetlands are a vital feature of Great Lakes ecosystems because they are comprised of primary producers that provide food and shelter for diverse communities of waterfowl, fish, and invertebrates (Maynard and Wilcox 1997). Many species of waterfowl, reptiles, amphibians, mammals and invertebrates rely on these special wetlands for feeding, resting and rearing their young. More than two-thirds of fish species that inhabit the Great Lakes depend on coastal wetlands for at least part of their life cycle. Wetlands have been defined in the Canada as: “land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic (e.g. water-loving) vegetation, and various kinds of biological activity adapted to wet environments.” (Maynard and Wilcox 1997). In this study, Great Lakes coastal wetlands refer to those wetlands that are hydrologically connected to the lakes (Figure i-1).

Stressors

Coastal wetlands are by nature stress-controlled systems as a result of the large-lake processes of the Great Lakes. In wetland ecology, a stressor can be defined as anything that changes the functioning of an ecosystem (Maynard and Wilcox 1997).

Coastal wetland ecosystems are characterized by simultaneous variations in many

environmental attributes, including water-level fluctuations and eutrophication (Chow-Fraser et al. 1998). Since the response of ecosystems to one factor may be influenced by other factors (Nilsen and Orcutt 1996), it is necessary to study the simultaneous effects of multiple factors on Great Lakes coastal wetlands. The major natural disturbance affecting coastal wetlands is water level fluctuations (Keddy and Reznicek 1986; Maynard and Wilcox 1997; Chow-Fraser 2005). The Great Lakes, particularly Lake Huron and Michigan, have experienced declining water levels in recent years, and these have affected important coastal wetlands that provide diverse habitats for plants, fish, and wildlife. These changes are caused by a combination of short-term, seasonal, and longer-term weather conditions, as well as climate changes. Climate models predict that air temperature changes will likely lead to climatic conditions that will significantly reduce water levels from 0.23 to 2.48 m among all the lakes (Table i-1).

Human population growth in the Great Lakes basin has imposed further stresses on coastal wetlands since European settlement. Figure i-2 shows a long-term pattern of human population growth in the Great Lakes basin. The majority of human population has been concentrated in Lake Michigan, Lake Erie and Lake Ontario (Figure i-3). Human-induced stressors add new stresses to which coastal wetlands are not naturally exposed. On a regional basis, Grosslink and Baumann (1980, cited in Mitsch and Grosslink, 2000) also found that loss rate of coastal wetlands was closely related to population density in the USA from 1954 to 1974. Figure i-4 shows the spatial pattern of land use in the Great Lakes basin. The urbanized areas and agricultural areas are

distributed mainly in the lower lakes which include Lakes Ontario, Erie, lower portion of Lakes Michigan and Huron. Studies have shown that urbanization increases runoff due to increased impermeable surfaces such as rooftops, roadways, and sidewalks. Increased runoff often results in increased loadings of nutrients, and toxic substances such as heavy metals, pesticides, oils, road salts, and detergents into waterways (Want et al. 2000), which may have detrimental effects on aquatic vegetation (Owen 1999).

Response of coastal wetlands to water levels

The cyclical expansion and contraction of emergent marsh in response to water-level fluctuations are well documented, and many studies have confirmed the negative relationship between percentage cover of emergent vegetation and increased lake elevation (e.g. Lyon et al., 1986; Keddy & Reznicek, 1986; Williams, 1995; Williams & Lyon, 1997; Hudon, 1997; Chow-Fraser et al., 1998; Gottgens et al., 1998). In the past, authors have tended to relate changes in areal cover of emergent vegetation to historical water levels to demonstrate the effect of water-level fluctuations on the vegetation dynamics of a particular site. While this approach has been useful for examining effects of water level disturbance at a single site over multiple years, it cannot be applied to multiple sites from a region experiencing the same water level at a given time, because in the latter case, water level is a constant, and cannot be used as an explanatory variable for emergent vegetation. Currently, there is a great need to develop predictive models to examine the effects of different water level regimes on emergent vegetation at a

basin-wide scale (Wescoat et al. 2006). The recent drop in lake levels in Lakes Huron and Michigan to near-record-low levels has also increased the need to develop basin-wide forecasts of how wetlands in these middle lakes will respond to further declines in lake levels with anticipated climate change scenarios (Magnuson et al. 1997). However, predictive models based on water level would be inappropriate for such basin-wide forecasts. A regional model taking into spatial variation at different sites needs be sought.

Many stressors affecting Great Lakes coastal wetlands are operating at the scale of landscape (Maynard and Wilcox, 1997). Therefore, it is important to monitor changes in supply of this critical fish habitat at a regional scale. Use of traditional field-based methods to detect and record the change in aquatic vegetation in Great Lakes wetlands is a daunting task because of their wide distribution throughout the Great Lakes shoreline. The use of aerial photos, satellite imagery, and a Geographical Information System (GIS) makes such a large-scale assessment easier. In addition, the spatial context of the organisms has been often lost when much of the traditional ecology deals with understanding the factors affecting the distribution and abundance of organisms (Molles 2002). Because GIS and remote sensing systems preserve this geographical information they are valuable tools for exploring large-scale wetland responses to natural and human-induced disturbances.

Use of GIS in wetland research

Explicit consideration of spatial information in ecological studies has become

increasingly more important in our attempt to better understand and manage ecological processes, and spatial analysis has emerged as an important tool in this rapidly growing sub-discipline of ecology (Fortin 2002). Levin (1992) pointed out that there was no single natural scale at which ecological phenomena should be studied. In some cases, patterns emerge from the collective behaviors of large ensembles of smaller-scale units; in other cases, patterns are imposed by larger-scale constraints. An example of this can be found in the fisheries literature, where geographic scale has recently become a key consideration in developing plans to conserve and restore fish habitat (Sly, P. G. et al. 1992; Crowder et al. 1996; Imhof et al. 1996; Kelso et al. 1996; Lewis et al. 1996; Minns et al. 1996; O'Driscoll et al. 1996; Armstrong et al. 1998; Folt et al. 1998; Mather et al. 1998).

Although fisheries and wetland ecologists have realized that cross-scale studies are critical to complement more traditional studies carried out on narrowly-defined spatial and temporal scales, large-scale analyses of fish distribution in systems as large as the Laurentian Great Lakes have not been attempted until recent years (e.g. Wei 2002). This is probably directly related to two main factors: 1) lack of appropriate computational tools and 2) lack of comprehensive and basin-wide geo-referenced data. Recent advances in the integration of GIS and readily available spatial statistics, however, has moved us closer to achieving the goal of analyzing fish distribution at the scale of all five Great Lakes.

Even without the benefit of spatial analysis, GIS has been useful for tracking organisms and habitat features, which are important first steps in development of effective environmental management plans. By superimposing results of biotic surveys over

geographic features, managers have been able to target critical habitats for protection and conservation (Miller 1994). Maps also provide spatial information that can be easily interpreted by a wide variety of experts, as well as by the general public. Therefore, maps are used routinely to display distribution of natural resources (e.g. fish habitats) in environmental impact assessment studies and environmental management plans alike. They help integrate interdisciplinary information and identify information gaps that are important in habitat and species conservation (Miller and Allen 1994).

Although there is a long tradition of publishing species distribution maps in ecology (Butterfield et al. 1994), most of these maps only show the locations of species with their associated habitat features, and rarely indicate intensity of use by the species. Any association between species distribution and particular habitat features is gleaned by eye without the benefit of spatial analysis. In a GIS environment, however, distribution maps can assess information regarding intensity of use and relative densities of habitat features, thus making them more ecologically relevant.

GIS in a nutshell

A GIS can be defined as “a computer system for collecting, checking, integrating and analyzing information related to the surface of the earth” (Rhind 1988). A full GIS requires hardware (computers and peripherals), software, data, people, and sound analysis methods for interpreting the results generated by the GIS.

GIS data are the core of a GIS. GIS data store the geometric location of geographic

features, along with associated attribute information. The locational information is stored in a vector or raster data structure (Fig. i-5), and corresponding attribute data (e.g., species information, water depth, substrate type, water quality) is stored in a set of tables related to the geographic features they describe (Environmental Systems Research Institute, Inc. 1999). Figure i-5 is a coastal wetland showing both a vector model and a raster model. Vector data records spatial information as geographical coordinates in a planar coordinate system. Point features are recorded as single x,y locations. Line features, including the outlines of polygons, are recorded as an ordered series of x, y coordinates. In ecology, vector data are very well suited to recording the location of discrete geographic features like wetland boundaries, streams, and locations of fishing nets. Raster data records spatial information in a regular grid (Fig. i-5). Each cell within this grid contains a number representing a particular geographic feature, such as water depth, etc. Raster data is usually used to store information about geographic features that vary continuously over a surface such as elevation, water depths, etc. Remote sensing data is a form of raster data in which each cell or pixel stores a value recorded by an optical or electronic device mounted on an airplane or a satellite (Fig. i-5).

The most common GIS analysis tasks have been identified by Mitchell (1999) as follows: 1) mapping where things are, 2) mapping the most and least, 3) mapping density, 4) finding what's inside, finding what's nearby, and 5) mapping change. Figures i-5, 6, 7, 8, 9, 10, and 11 illustrate the most common GIS operations being implemented in this study, showing how these operations alter the geometric properties of the features in a

dataset and create new data. With spatial analysis operations, we can find point features that are nearest to the selected point or line features, point features according to the polygon features they fall inside, line features according to the polygon they intersect, polygon features according to the polygon features they intersect, features that are adjacent to the selected features. We can create new features by merging adjacent features into one feature and select features by specifying a buffer distance. Advanced spatial analysis operations can do much more (See ESRI, 1999 for more detail).

Thesis organization and objectives

The overall purpose of this study was to examine the response of coastal wetlands to declining water levels and environmental disturbances in the Great Lakes. I have organized my thesis into six chapters. Chapters 1 to 3 will address response of emergent vegetation to the long-term water level fluctuations in Lakes Ontario coastal wetlands while taking into account other factors such as urbanization and invasive species. Chapters 4, 5, and 6 will address the response of fish and explore the use of high resolution satellite imagery to quantify the supply of fish habitat and wave disturbance in coastal wetlands. Because all these chapters have been previously published or prepared as journal articles, there was some overlap in the chapters, particularly the list of literature cited in each.

Chapter 1: Untangling the confounding effects of urbanization and high water level on the cover of emergent vegetation in Cootes Paradise Marsh (Published in Hydrobiologia

(2005) 544: 1-9).

This chapter proposes a new approach to investigate the effect of long-term water level fluctuations on coastal marshes while taking into account the effect of site bathymetry and urbanization within the watershed. The goals of this paper were to: 1) calculate the inundated area, a derived measurement that reflects the bathymetry of the marsh and the mean water level for a particular year and compare the predictive powers of the inundated area and water level, 2) determine the contribution of urbanization to the changes in emergent plants independently of hydrological disturbance.

Chapter 2: Synergistic impact of water-level fluctuation and invasion of Glyceria on Typha in a freshwater marsh of Lake Ontario (Published in Aquatic Botany (2006) 84: 63-69.

Previous work (Chapter 1) documented the effects of high water levels and increased urbanization in the Hamilton region on emergent community as a whole in Cootes Paradise Marsh. Chapter 2 examines these simultaneous impacts at the species level focusing on the effects of multiple stressors on the native *Typha* marsh community in Cootes Paradise Marsh. The goals of this study were to: 1) examine the effects of multiple stressors on major emergent plants in Cootes Paradise Marsh, 2) detect the spatial displacement of *Typha* by *Glyceria* and, 3) determine the correlations between the exposed habitats and response of the major emergent plants.

Chapter 3: A regional model of emergent vegetation in response to water level fluctuations in Lake Ontario.

This chapter describes the development of a regional model that can be used to assess the aggregate response of emergent plants to hydrological disturbance over a large geographical scale. A conventional approach to study the effect of water-level fluctuations on the cover of emergent plants in wetlands is to relate historical water levels to areal changes in emergent vegetation. However, application of this method could be problematic if multiple sites from a region experiencing the same water level are compared because in this instance, water level would be a constant and cannot be used as an explanatory variable for emergent cover. The goals of this study were to: 1) modify the approach described in Chapter 1 to account for the differences in wetland area at multiple sites, 2) confirm the published results of Wei and Chow-Fraser (2005) that predicted the inundated area can be used to assess the response of emergent cover for multiple sites that have the same water level and, 3) test the effect of urbanization on emergent vegetation at a regional scale.

Chapter 4: Influence of shoreline features on fish distribution in the Laurentian Great Lakes (Published in Canadian Journal of Fisheries and Aquatic Sciences (2004) 61: 1113-1123).

This chapter describes the association between certain species or species-complexes and particular shoreline features within the Great Lakes. The primary goals of this study

were: 1) test if the Great Lakes fish communities utilize certain shoreline features (especially wetlands) disproportionately to their availability and 2) test if the distribution of wetland-associated taxa is influenced by wetland type (i.e., protected embayment versus open-shoreline wetland) which reflects the difference in exposure disturbance.

Chapter 5: Use of IKONOS imagery to map coastal wetlands of Georgian Bay

(Published in Fisheries (2007) 32: 167-173).

This chapter describes the use of IKONOS satellite imagery to map aquatic habitats in coastal wetlands of Georgian Bay. Use of traditional field-based method to detect and record the change in aquatic vegetation in Great Lakes wetlands is not cost-effective because of the wide distribution of the wetlands throughout the Great Lakes shoreline. Mapping wetlands for such a large geographic area necessitates the use of remote sensing technology to obtain an accurate inventory of these ecosystems. IKONOS satellite Imageries for Fathom Five National Marine Park in Lake Huron and an area of eastern Georgian Bay were acquired in 2002 and eleven wetlands were chosen for habitat mapping with remote sensing software. The primary objective of this study was to explore the capabilities of using IKONOS satellite imagery to map different types of aquatic vegetation and habitat features in Great Lakes wetlands.

Chapter 6: Development of an index to account for degree of exposure and wetland geomorphology on submerged aquatic vegetation of Georgian Bay coastal wetlands.

This chapter describes the development and use of a Wetland Exposure Index (WEI) that can be used to quantify exposure conditions in coastal wetlands. This index was based on wetland geomorphology and the Relative Exposure Index (REI), which has been widely applied to lake and marine ecosystems. Ten wetlands from Fathom Five National Marine Park, located at the boundary zone of Lake Huron and Georgian Bay, were evaluated for the effectiveness of this new index. The goals of this study were: 1) develop an index that can be used to quantify the exposure condition in coastal wetlands, 2) compare the relative power of the new index and Relative Exposure Index (REI) in explaining the variation in submergent aquatic vegetation (SAV) while taking into account the effect of water quality on SAV.

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Table i-1. Projected changes in mean lake levels for 2050. Units are in meters.

Decline in water levels	Lake	Lake	Lake	Lake	Lake
	Superior	Michigan	Huron	Erie	Ontario
Croley 1990, 1993	-0.23	-1.62	-1.62	-1.36	-1.30
Mortsch and Quinn, 1996	-0.23 to -0.47	-0.99 to -2.48	-0.99 to -2.48	-1.30	-1.30

Figure i-1. A simplistic wetland model showing different plant zones. Beginning from the upland to open water area along the depth gradient are wet meadow (WM), emergent plants (EM), floating-leaved plants (FL), and submergent aquatic vegetation (SAV).

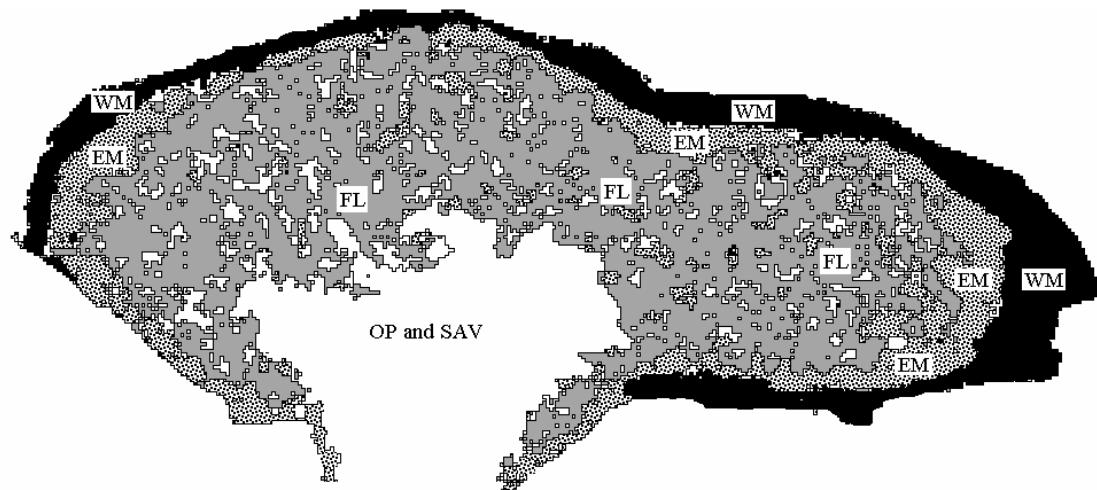


Figure i-2. Human population growth in the Great Lakes basin (Sources: Environment Canada and United States Environmental Protection Agency)

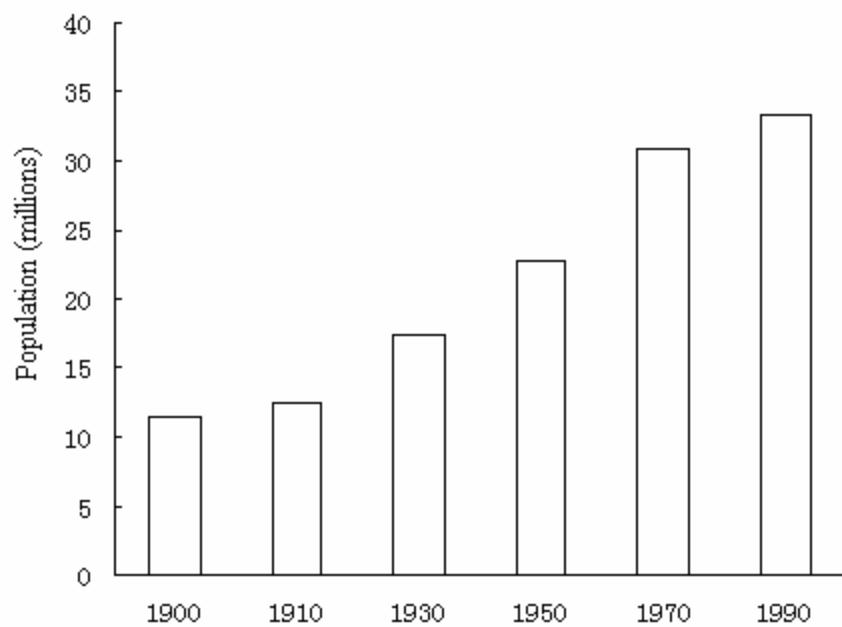


Figure i-3. Spatial distribution of human population in the Great Lakes for 1991 (Sources: Environment Canada and United States Environmental Protection Agency)

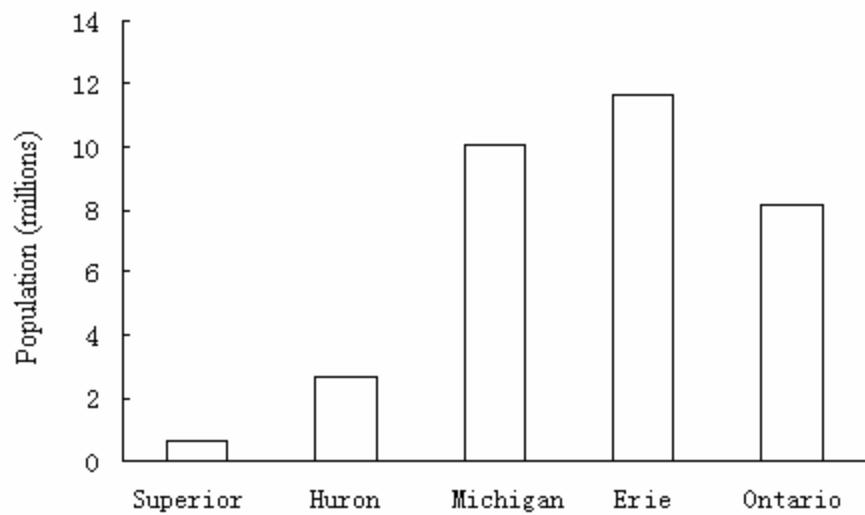


Figure i-4. Land use in the Great Lakes basin (Sources: Environment Canada and United States Environmental Protection Agency). Agr: agriculture, Res: residential.

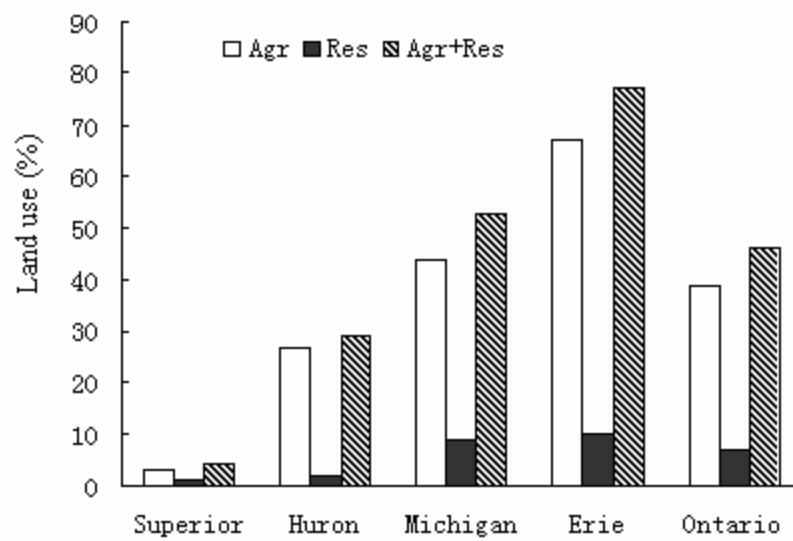
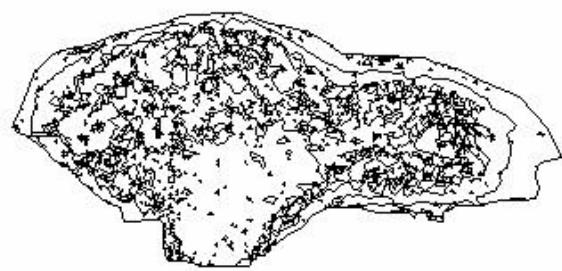
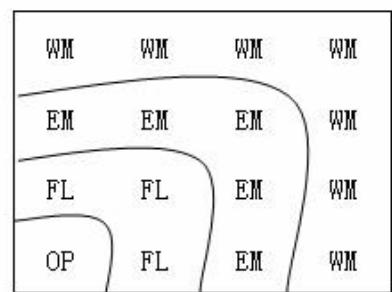


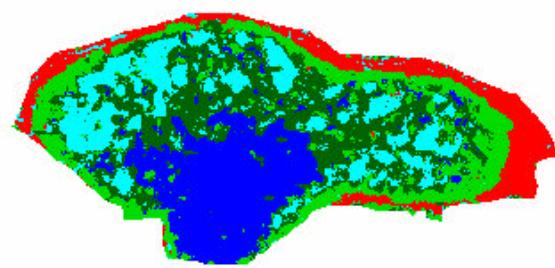
Figure i-5. GIS models of a coastal wetland



Vector data in GIS



Vector data model



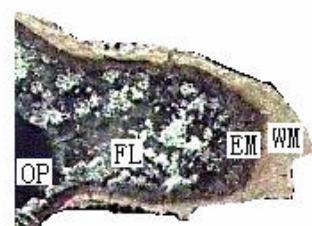
Raster data in GIS

WM	WM	WM	WM
EM	EM	EM	WM
FL	FL	EM	WM
OP	FL	EM	WM

Raster data model



Satellite image



Wetland

Figure i-6. Clip one theme based on another. This process creates a new theme by using a polygon theme as a cookie cutter on a point, line, or polygon theme.

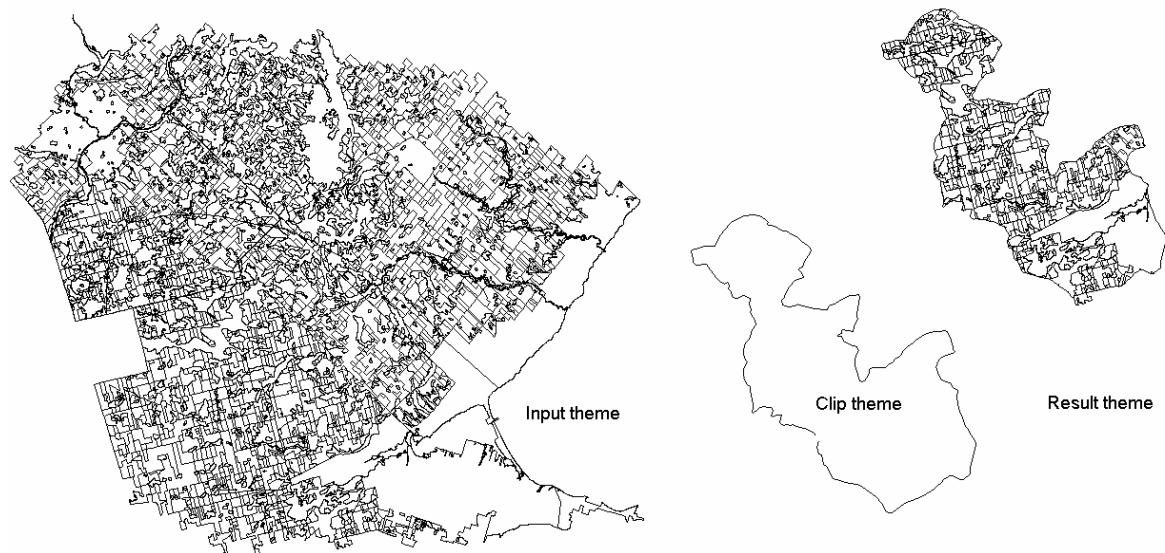


Figure i-7. Intersect two themes. This process is similar to clipping a theme, except it preserves only those features falling within the spatial extent common to both themes. The attribute data from both themes are included in the new theme's attribute table.

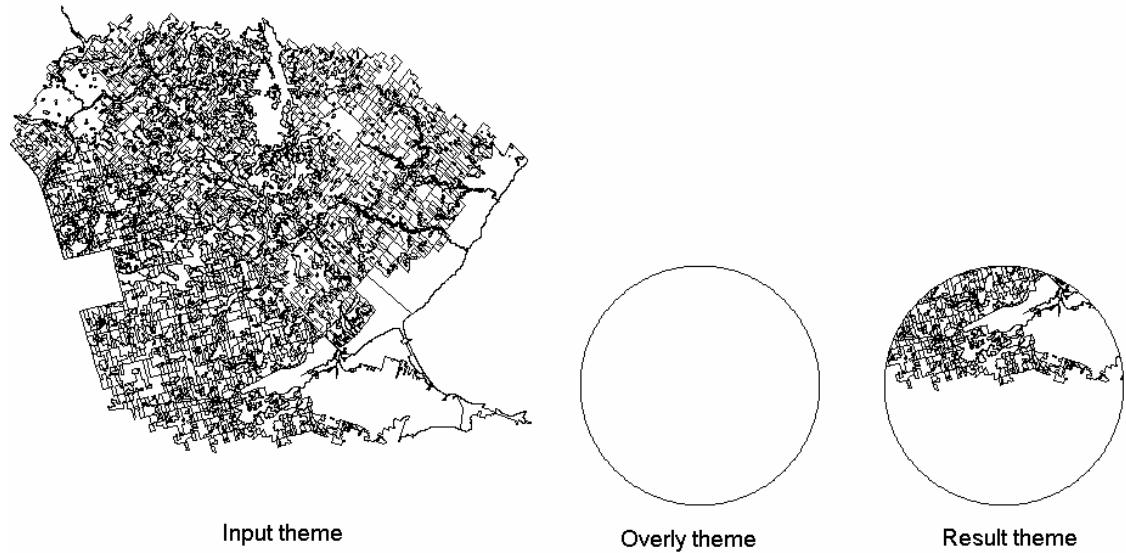


Figure i-8. Dissolve features based on an attribute. This process aggregates the features that have the same value for an attribute.

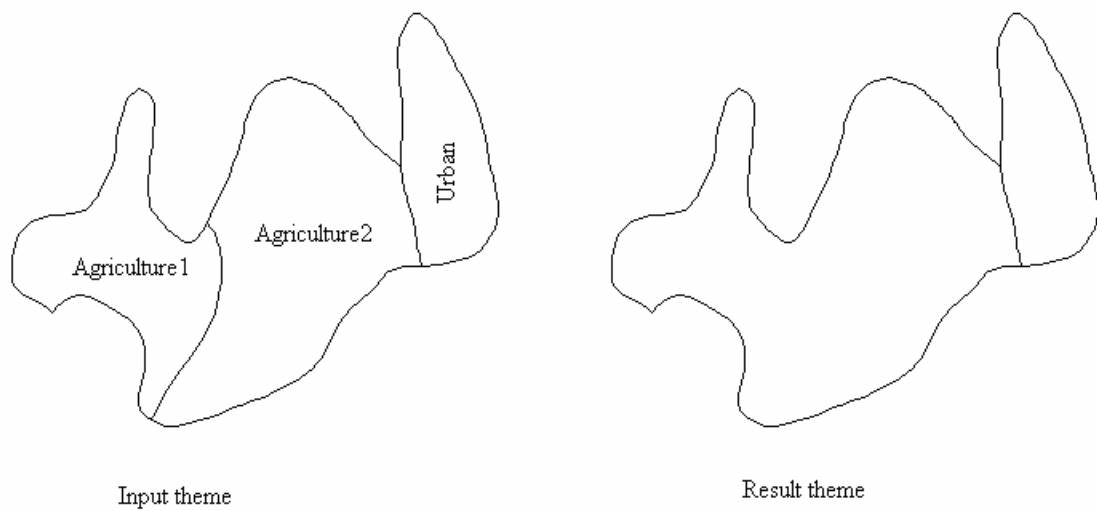


Figure i9. Union two themes. Union combines features of an input theme with an overlay theme to produce a result theme that contains the attributes and full extent of both themes.

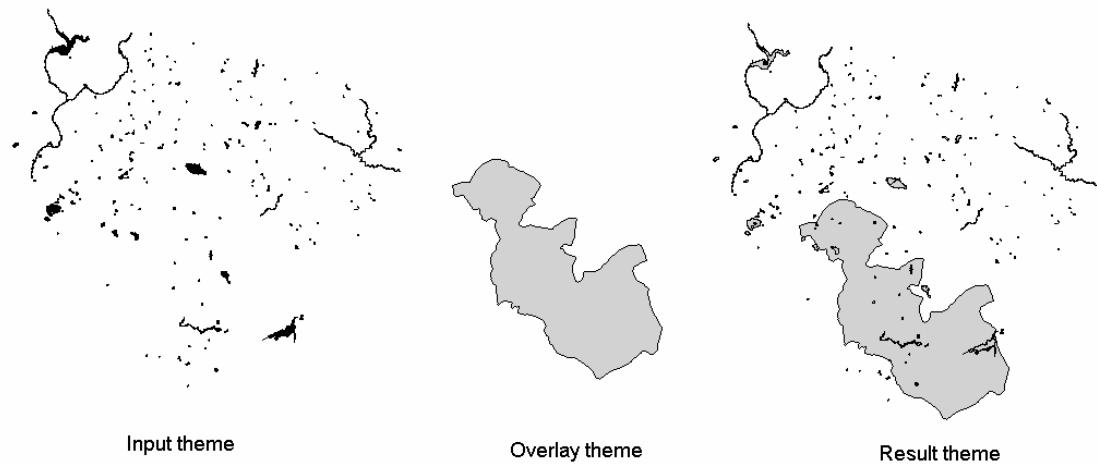


Figure i-10. Merge themes together. Merge appends the features of two or more themes into a single theme.

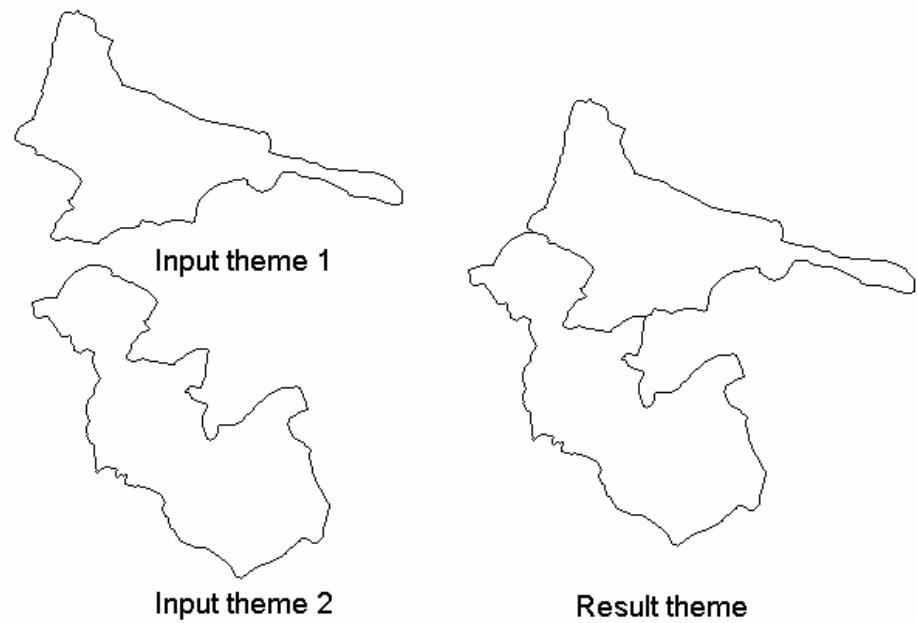


Figure i11. Assign data by location (spatial joint). Assigning data by location uses a spatial relationship to join data from one theme to another theme (ESRI, 1999).

	point	line	polygon
point	●	—	
line	●	nearest	nearest
polygon		n/a	n/a

CHPATER 1:

**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

Wei, A. and Chow-Fraser, P. 2005. Hydrobiologia 544: 1-9.

ABSTRACT

An approach based on a Digital Elevation Model (DEM) was used to untangle the confounding effects of long-term water-level fluctuations and increasing human population on the cover of emergent vegetation in Cootes Paradise Marsh, a degraded coastal wetland in Lake Ontario, Canada. Data for twenty observations between 1934 and 1993 were used in the analysis. First, we calculated the inundated area based on the DEM, a derived measurement that reflected the bathymetry of the marsh and the mean water level for a particular year. Then Mantel correlations and regression analyses were used to analyze the relationships between emergent cover and corresponding water level, inundated area, and human population, respectively. Results of the simple and partial correlations indicated that areal change of emergent plants was significantly correlated with inundated area after controlling for the effect of water level fluctuation; however, there was no significant correlation between emergent cover and water level after controlling for inundated area. This is an important consideration when multiple sites from the same Great Lake are compared since the same water level may correspond to vastly different inundated areas for different marshes. Changes in emergent cover were also significantly correlated with human population after controlling for water level effects. Altogether, inundated area explained 83.1 % of the variation, human population explained 4.2 %, and the interaction between population and inundated area explained an additional 4.3% of the remaining variation in areal emergent cover. This indicates that the synergistic effect of high water level (expressed as inundated area) and increased

human population induced greater detrimental impact on the emergent plants than did either stressor alone.

INTRODUCTION

Emergent marshes along the shores of the Laurentian Great Lakes play an important ecological role in stabilizing the substrate, providing habitat for fish, amphibians, invertebrates, birds, and mammals, replenishing dissolved oxygen concentration in the water column, and sequestering nutrients from the sediment. Their cyclical expansion and contraction are well documented, and many studies have confirmed the negative relationship between percentage cover of emergent vegetation and increased lake elevation (e.g. Lyon et al., 1986; Keddy & Reznicek, 1986; Williams, 1995; Williams & Lyon, 1997; Hudon, 1997; Chow-Fraser et al., 1998; Gottgens et al., 1998). In urbanized wetlands such as Cootes Paradise Marsh, the dramatic reduction in emergent cover has taken place over six decades of increasing anthropogenic disturbance (e.g. nutrient enrichment from sewage effluent, hydrologic modification, and increased runoff from agricultural and urban development in the watershed; Chow-Fraser et al., 1998; Chow-Fraser, 2004). Since both increased water level and nutrient enrichment can negatively affect the establishment of a large number of emergent taxa in experimental microcosms (Weiher et al., 1996), it is possible that factors other than high water level (e.g. cultural eutrophication) have been responsible for the loss of emergent vegetation in this and other wetlands in developed areas of the Great Lakes shoreline (e.g. Frenchman's Bay in Eyles et al., 2003).

A number of equations have been published that relate areal emergent cover to water level on a site-specific basis (e.g. Lyon et al., 1986; Hudon, 1997; Chow-Fraser et al.,

1998); however, these equations have rarely been applied to other wetlands because the effect of water level depends on marsh bathymetry that can differ greatly from site to site within one Great Lake. Thus, despite the accumulation of historic vegetation maps for several marshes (e.g. Williams, 1995; Chow-Frsaer et al., 1998; Wilcox et al., 2003), a regional model of vegetation response has not emerged, primarily because no study has yet been conducted that statistically relates emergent cover to the actual depth of inundation.

An effective way to quantify the amount of inundated area of a particular wetland is to apply water-level scenarios to a detailed bathymetric map of the wetland. With the advent of geographic information systems (GIS) and widespread availability of satellite data, digital elevation models (DEM) can now be used in conjunction with lake elevation data to calculate area of inundation. This approach has been used to simulate the effects of flooding in rivers (e.g. Vining, 2002), but has rarely been used in studies of water-level effects on wetland succession. In this paper, we propose to use this approach to investigate the response of emergent vegetation to simultaneous changes in water level and anthropogenic disturbance in the watershed of Cootes Paradise Marsh over sixty years of observations. First, we will use a DEM to quantify the amount of inundated area for twenty different water-level scenarios from 1934 to 1993, and then we will assess how the amount of inundated area and human population growth has contributed separately and synergistically to the corresponding decline in marsh vegetation. Our approach should be generally applicable to other wetlands and should eventually lead to

the generation of an aggregate response of emergent vegetation to water level fluctuations at a regional or basin-wide scale.

METHODS

Study site

Cootes Paradise is a 250-ha coastal wetland, located at the west end of Hamilton Harbour, a natural embayment at the west end of Lake Ontario (Fig. 1-1). Since the marsh is managed as a nature reserve, only non-motorized boats/canoes are permitted access, and there is no recreational fishing. Cootes Paradise is drained by three main tributaries, Spencer Creek, Borer's Creek and Chedoke Creek. It receives runoff from adjacent agricultural, residential, industrial, commercial, and recreational lands, as well as effluent from the Dundas Sewage Treatment Plant. It is surrounded by the city of Hamilton to the south, and the towns of Ancaster, Dundas and Flamborough to the north and west. The major natural disturbance is interannual and seasonal fluctuations in water level of Lake Ontario (Chow-Fraser, 2004). In general, water level rises in the spring, peaks in May or June, and recedes in the autumn, reaching its lowest level in December. Prior to the last century, almost 100% of Cootes Paradise was covered with emergent plants, but sustained high water depths from the 1930s to the 1980s have contributed to a dramatic decline of emergent cover approaching 10% by the 1990s (Chow-Fraser et al., 1998; Chow-Fraser, 2004). Historical changes in the percent cover of the major species of emergent plants in Cootes Paradise Marsh during the years of

observation are shown in Table 1-1. Under a high water level of 75.4 m, the marsh has a mean depth of 0.93 m (Fig. 1-2).

Data

The percentage cover of emergent vegetation (EM) between 1934 and 1993 were taken from Chow-Fraser et al. (1998). These data were obtained from several sources: historical aerial photographs and historical vegetation maps stored in the Royal Botanical Gardens. The historical vegetation maps for years of 1946, 1953, 1971, 1974, 1975, 1977, 1978, and 1979 were available in a GIS format. Detailed descriptions of the sources of data and data processing are documented in Chow-Fraser et al. (1998).

Since water quality data were not available for early years, historical census population data (POP) were used as a surrogate of potential cultural eutrophication. Data for the Hamilton-Wentworth region (including Hamilton, Dundas, Ancaster, and Flamborough shown in Fig. 1-1) were obtained from Statistics Canada and Ontario Ministry of Revenue. The data for several missing years were estimated from a regression equation ($\text{Population} = -30.647 \cdot \text{year}^2 + 8098.8 \cdot \text{year} - 31950$, $r^2 = 0.9931$).

Water-level (WL) data for Cootes Paradise were estimated by applying a formula developed by Chow-Fraser et al. (1998) to continuous water-level data measured at Station 13150 in Burlington, Ontario. The formula was : $\text{WL (C)} = 1.038 (\pm 0.029) \cdot \text{WL (B)} - 2.802$, $n = 38$, $r^2 = 0.88$, $P < 0.0001$, where WL (C) and WL (B) are the water levels (m, asl) for Cootes Paradise and Burlington, respectively.

Inundated areas (IA) for the corresponding mean summer (May to September

inclusive) water levels for different years were estimated using the DEM of Cootes Paradise in ESRI ArcGIS 8.1 (<http://www.esri.com>). The DEM was produced from Painter et al. (1989)'s contour map and a DEM provided by the Ontario Ministry of Natural Resources (OMNR; M. Robertson, Peterborough, Ontario, Canada). Since the contour map only provided depth information for 75.06 m above sea level, and site elevations greater than 75.06 m were not available, a seamless merge with OMNR's elevation points was required. We digitized the contour map using its scanned map, and then re-sampled 2300 water depth values from the contour map. The overlapping area was removed from OMNR's DEM before it was merged with the re-sampled points from the contour map. The final DEM was interpolated from the combined elevation points using Spatial Analyst in ESRI ArcGIS 8.1. The cell size of the DEM was 10 meters. Wise (2000) provides a comprehensive review of approaches to assess DEM quality and suggests that the best judge of the adequacy of a DEM is the level of accuracy for the intended purpose. We assessed the accuracy of the DEM using two sets of field-measured data. The first set consists of 21 points with known water depths and coordinates that were collected in summer of 2003. We found no significant differences in mean water depth between the DEM and the field data when difference in the means was set to 5 cm (paired t -test, $p = 0.15$, two tailed). The second dataset was the historical vegetation maps for years of 1946, 1953, 1971, 1974, 1975, 1977, 1978, and 1979. The DEM would be adequate for this study if the inundated areas derived from the DEM do not differ significantly from the measured open-water areas corresponding to the digitized

vegetation maps. Theoretically, in a wetland with abundant emergent plants, the inundated areas should be larger than the observed open-water areas. Based on this assumption, we set the difference in the mean areas to be 10%, and we found no significant difference between calculated inundated areas and measured open-water areas (paired t-test; $p = 0.14$, two tailed). Therefore, both analyses indicated that our DEM is hydrologically accurate for Cootes Paradise Marsh and is acceptable for the proposed use in this study (see Fig. 1-2 for a bathymetric map derived from the DEM).

Statistical analysis

We performed a simple Mantel test and a partial Mantel test using S-plus 2000 and zt (Bonnet & Peer, 2002) to assess the correlations among water level, inundated area, census population, and emergent cover. The Mantel test used here is a statistical test that has been widely applied in population genetics, ecology, anthropology, psychometry, and sociology (Legendre, 2000). The simple Mantel test is an examination of the relationship between two distance matrices. The partial Mantel test estimates the correlation between two matrices while controlling for the effect of a third matrix (i.e. one variable is held constant) (Legendre & Legendre, 1998) and it is used to remove potential spurious relationships. Regression analyses were performed in SAS JMP 5.1.

RESULTS

The hypsographic curve of Cootes Paradise marsh in Figure 1-3 describes the relationship between the surface area of marsh contours and lake elevation or depth.

According to this relationship, the range of water-level conditions that have been encountered in a high-water year (75.3 m) such as that encountered in 1993 would correspond to IA of 99%, while a low-water year such as that encountered in 1946 (74.4 m) would correspond to only 58% of the total marsh area. This confirms the great impact that water level fluctuations must have exerted on the emergent vegetation through the six decades of observations (Fig. 1-4).

EM was significantly regressed against WL, POP and IA (Fig. 1-4a, b, and c, respectively; Table 1-2). Of these, IA was the strongest predictor of EM, explaining 3.3% more of the unexplained variation than did WL, and 26.2% more than did POP. A simple Mantel correlation “r” measures the extent to which variations in distances of a variable (e.g., WL) correspond to variations in another variable (e.g., EM), while the partial Mantel test estimates the correlation between the two matrices while controlling for the effect of a third matrix (Fig. 1-5). Results of both tests are displayed in tabular form in Figure 1-5. Correlation coefficients for the Mantel are displayed above the diagonal, while those for the partial Mantel test are displayed below.

All coefficients of the Mantel tests for WL, IA and EM were significant (values in bold in Fig. 1-5a). However, the partial correlation coefficient of 0.1653 (Fig. 1-5a) indicates that after controlling for effects of IA, WL was no longer significantly correlated with EM. By contrast, the coefficient 0.3785 (Fig. 1-5a), indicates that IA was significantly correlated with EM, after controlling for variations in WL. Not surprisingly, the correlation between WL and IA was significant ($r = 0.9427$) and even

after accounting for variations in EM ($r = 0.7980$) (Fig. 1-5a). These results support the causal model in which WL causes changes in IA and which then causes changes in EM (flow diagram in Fig. 1-5a).

Next, we performed a similar analysis with IA, EM and POP (Fig. 1-5b). We omitted WL from the analysis because IA, which incorporates the effects of WL and bathymetric characteristics, was shown to have greater explanatory power. Both IA and POP were significantly correlated with emergent cover when the partial Mantel tests were performed (0.8071 and 0.4710, respectively; Fig. 1-5b). This is consistent with results of the regression analyses in Figure 1-4, and supports a causal model in which both IA and POP can affect the percent cover of EM in Cootes Paradise Marsh (flow diagram in Fig. 1-5b). Figure 1-5c is a combined causal model that reflects the results of all Mantel correlation coefficients. This model indicates that the effects of WL on EM are indirect, since IA, which takes into account the unique bathymetry of the wetland, has the most direct effect. On the other hand, POP appears to have a direct link to EM as well.

To determine the relative power of each explanatory variable for predicting variation in EM, we performed a stepwise regression analysis with IA, WL and POP. AI explained 83.1% of the total variation ($P = 0.0001$) while POP explained an additional 4.2 % ($P = 0.0308$), but addition of WL did not explain any more of the residual variation ($P = 0.4117$). We then performed a multiple regression analysis with IA, POP and IA*POP (interactive term between the two independent variables). This model was highly significant ($P < 0.0001$), and was accompanied by an improved R^2 -value of 0.916

compared with a model that does not include the interactive term. Each of the independent variables explained a significant amount of the variation in EM, as did the interactive term, POP * IA (Table 1-2). This analysis indicates that the synergistic effect of total inundated area and increased human population induced greater detrimental impact on emergent plants than did either stressor alone.

DISCUSSION

An important finding in this study is that WL showed no significant correlation with EM, after controlling for the confounding effects of IA; in contrast, when the effect of WL was controlled, IA still showed a significant correlation with EM (Fig. 1-5). This confirms that IA is a better predictor than WL for EM since the former reflects both site-specific bathymetric characteristics and lake level. Another important finding is that increased human population in the region had a significant negative effect on the cover of emergent vegetation in Cootes Paradise Marsh. Previous studies have confirmed the linkage between the levels of land-use alteration and the changes in aquatic ecosystems (e.g., Booth et al., 1997; Ellis et al., 1996; Wang et al., 2000). Ellis et al. examined the influence of landscape condition on water level fluctuations in wetlands in the Prairie Pothole Region of North and South Dakota, and concluded that water-level fluctuations are greater in wetlands located in areas of intensive agricultural activity relative to those in more natural grassland settings. Taylor (1993, cited in Schueler, T.R. 1994) suggested that additional storm water due to the effect of watershed development and increase in

imperviousness contribute to greater annual water level fluctuations on 19 freshwater wetlands in King County, Washington. In general, urbanization increases runoff due to increased impermeable surfaces such as rooftops, roadways, and sidewalks because the greater the population there is in a region, the greater the need there is for residential, commercial, industrial, institutional, agricultural, and transportation. The significant negative effect of increased human population on emergent plants in Cootes Paradise Marsh implied that increased impervious surfaces due to increased human population in the region also contribute to the greater water level fluctuations and eventually cause detrimental effect on the emergent vegetation in Cootes Paradise Marsh.

In addition to greater water level fluctuations, increased runoff often result in increased loadings of nutrients, toxic substances such as heavy metals, pesticides, oils, road salts, and detergents into waterways (Wang et al., 2000), which may have detrimental effects on aquatic vegetation (Owen, 1999). On a regional basis, Grosslink & Baumann (1980, cited in Mitsch & Grosslink, 2000) also found that loss rate of coastal wetlands was closely related to population density in the USA from 1954 to 1974.

In the case of Cootes Paradise Marsh, increase in the surrounding population also meant an increase in phosphorus loading since effluent from the Dundas Sewage Treatment facility has been discharged directly into the marsh for over 8 decades (Chow-Fraser et al., 1998). Even though sewage effluent is no longer an important source of external phosphorus load to the marsh, there is a great deal of accumulated phosphorus in the sediment that continues to be released during the summer, and thus

maintains the overall hypereutrophic state of the wetland (Kelton et al., 2004). The inundation of previously terrestrial soils by increasing water levels can also lead to an increase in available nitrate via increased denitrification. Weiher et al. (1996) has shown that emergent taxa such as *Typha* are unable to become established in enriched sediments even in drawdown conditions, although more can survive when they are exposed to slightly inundated but infertile sediments. Even though water depth and soil fertility are two dominant factors, there may be other important factors, such as wave actions and bioturbation (Chow-Fraser, 1998), and these should be investigated as appropriate.

CONCLUSION

The distribution of emergent plants is a function of many factors including water level, bathymetry, nutrients, competitive interactions among plants (Kellogg et al., 2003), and bioturbation (e.g. carp). We have shown that inundated area (a measure derived from DEM that reflects the bathymetry of the marsh and the mean water level for a particular year) as well as the population size of the region are significant predictors of the percent cover of emergent vegetation in Cootes Paradise Marsh, explaining close to 92 % of the variation. Mantel correlations indicate an overall causal model in which emergent plant cover is directly affected by both inundated area and population size, and where water level indirectly affects emergent vegetation by regulating the amount of inundated area. Cultural eutrophication, as a consequence of urbanization, contributes measurable negative impact on the aquatic habitat and can interact with natural

disturbances such as high water levels to accelerate marsh degradation.

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Table 1-1. Historical changes in the percent cover of the major species of emergent plants in Cootes Paradise Marsh during the years of observation.

Major species	1946	1953	1971	1974	1975	1977	1978	1979
<i>Typha latifolia</i>	24.0	4.0	3.6	0.3	0.4	0.6	1.7	1.2
<i>Glyceria maxima</i>	14.7	4.3	15.4	5.2	8.9	6.8	9.1	10.7
<i>Sparganium eurycarpum</i>	0.3	0.3	0.0	0.0	0.2	0.0	0.5	1.0
<i>Nymphaea tuberosa</i>	0.2	1.6	0.1	0.1	0.0	0.0	0.0	0.1
<i>Bidens cernua</i>	0.0	1.7	0.2	0.3	3.4	6.4	1.4	0.3
<i>Phragmites australis</i>	0.0	0.0	1.0	0.2	0.2	0.1	0.1	0.1

Table 1-2. Regression analyses (n = 20)

Dependent variable	Term	Coefficient	SE	t	p
EM	Intercept	2657.80	312.38	8.51	<.0001
	WL	-35.16	4.172	-8.43	<.0001
EM	Intercept	101.97	15.91	6.41	<.0001
	POP	-2.06	0.42	-4.87	0.0001
EM	Intercept	73.69	5.37	13.73	<.0001
	IA	-24.60	2.61	-9.41	<.0001
EM	Intercept	76.84	9.20	8.35	<.0001
	IA	-11.91	3.77	-3.16	0.0061
	POP	-0.81	0.26	-3.09	0.0070
	(IA-1.96)*(POP-37.11)	0.89	0.31	2.86	0.0113

Figure. 1-1 Location of Cootes Paradise Marsh, Hamilton Harbour and the surrounding city of Hamilton, and towns of Ancaster, Dundas and Flamborough.

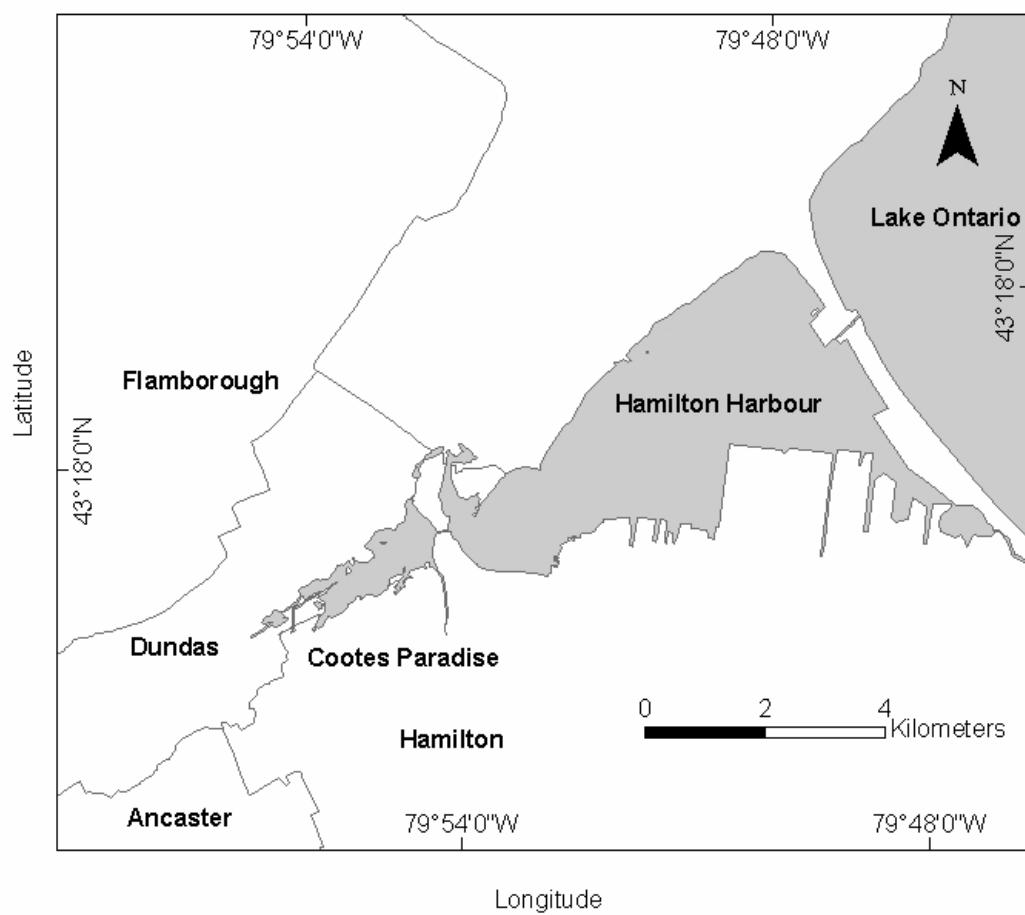


Figure. 1-2 A bathymetric map of Cootes Paradise Marsh. Units are in meters.

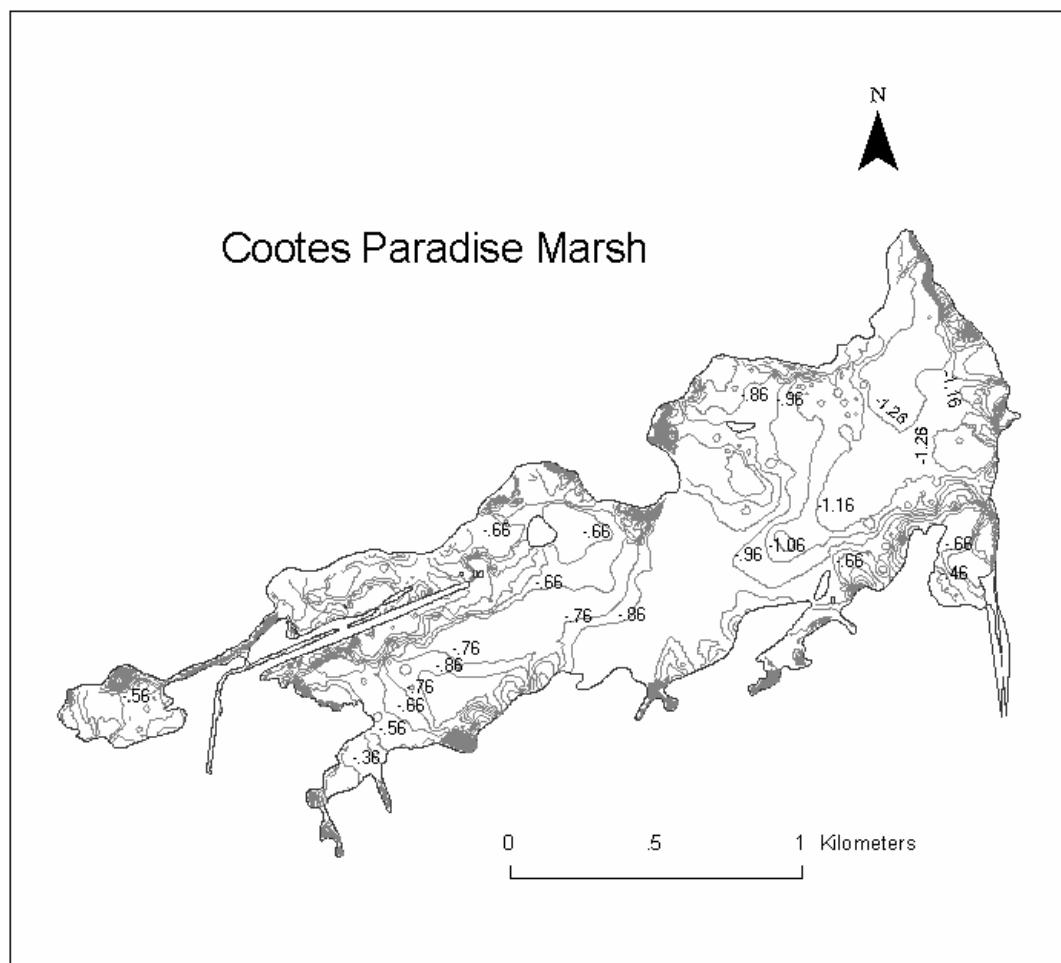


Figure 1-3. Hypsographic curve of Cootes Paradise marsh

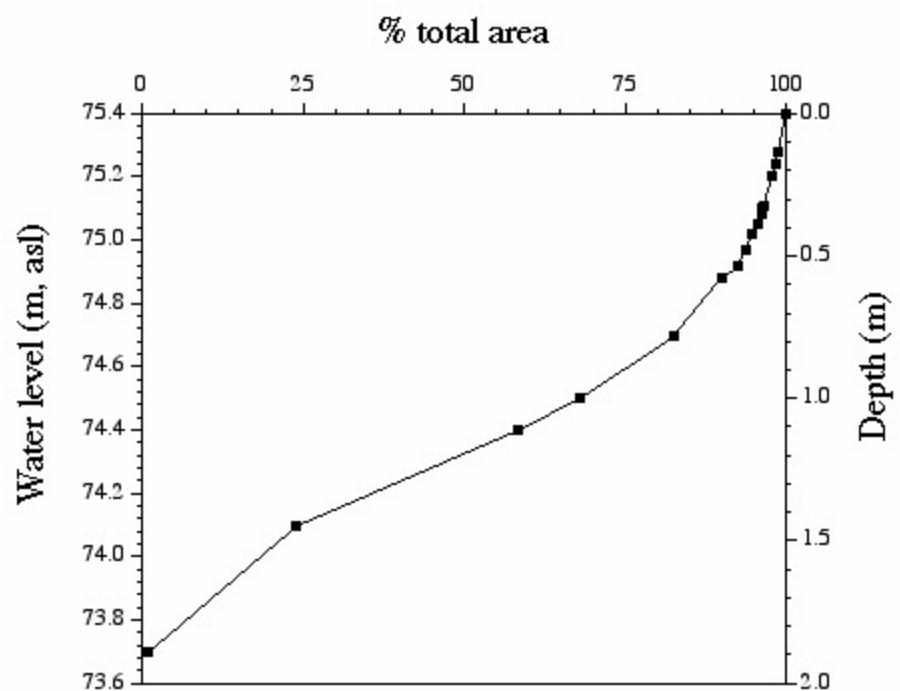


Figure 1-4. Linear regression between % emergent vs. a) water level (m, asl), b) population census and c) inundated area for Cootes Paradise from 1934 to 1993.

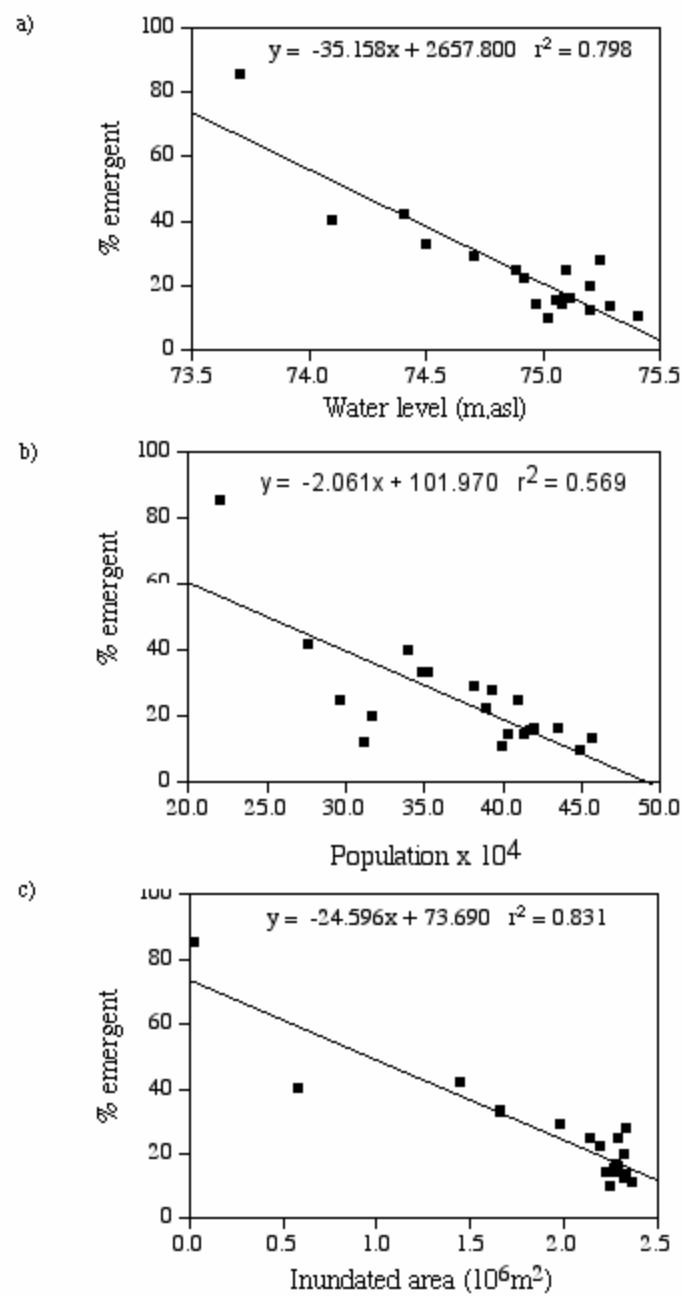


Figure 1-5. Mantel correlations between matrices for a) water level (WL), inundated area (IA), and emergent cover (EM), and b) inundated area, emergent cover and census population (POP). Causal models are presented to the right of the matrices. Correlations in bold indicate significance at $\alpha = 0.05$

a)

	WL	IA	EM
WL	---	0.9427	0.8379
IA	0.7980	---	0.8588
EM	0.1653	0.3785	---

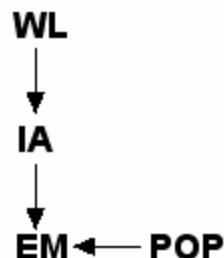


b)

	IA	POP	EM
IA	---	0.5072	0.8588
POP	-0.1160	---	0.6435
EM	0.8071	0.4710	---



c)



CHAPTER 2:

**Synergistic impact of water-level fluctuation and invasion of *Glyceria* on *Typha* in a
freshwater marsh of Lake Ontario**

Wei, A. and Chow-Fraser, P. 2006. Aquatic Botany 84: 63-69

ABSTRACT

The effects of multiple stressors on the native *Typha* marsh community (mainly *Typha latifolia*) were examined using historical records of water levels, human census population, and field vegetation maps. Percent cover of the major plant species was estimated in a GIS, and the percent cover of *Typha* was related to changes in water level, human population growth, and percent cover of exotic *Glyceria maxima* and invasive *Phragmites australis*. Water-level fluctuation was the major natural disturbance and it alone accounted for 88% of the variation in *Typha*. After partitioning out the effect of water level, both human population growth and the presence of exotic species were still significantly related to the decline of native *Typha*. We suggest that multiple stressors interact with each other to influence changes in native *Typha* community and cause greater detrimental impact. An important implication of our results is that projected water level decline due to climate change may not necessarily favor the restoration of a desirable native marsh because of the presence of other disturbances such as exotic and invasive species and altered nutrient regime.

INTRODUCTION

Coastal wetlands of large lakes are characterized by simultaneous variations in many environmental attributes, including interannual water-level fluctuations (Keddy and Reznicek 1986, Hudon 1997, Chow-Fraser 2005), eutrophication (Chow-Fraser et al. 1998; Wei and Chow-Fraser, 2005) and the presence of invasive and exotic plant species (Wilcox et al. 2003). Of these, water-level fluctuation is considered a natural disturbance, and the cyclical expansion and contraction of emergent marshes in response to Great Lakes water-level fluctuations have been well documented (e.g. Lyon et al., 1986; Williams and Lyon, 1997; Hudon, 1997; Chow-Fraser et al., 1998; Gottgens et al., 1998). Effects of the other two factors, however, are considered anthropogenic, since both degraded water quality and invasion of exotic species have been directly related to urbanization (e.g., Crosbie and Chow-Fraser, 1999; Davis and Froend, 1999; Eyles et al., 2003; Baldwin, 2004; Chow-Fraser, 2006).

Although water-level fluctuations is likely the largest single disturbance affecting the response of coastal marsh communities, it is necessary to study the effects of multiple factors simultaneously, since the response of ecosystems to one factor may be significantly influenced by others (Nilsen and Orcutt, 1996). Recent declines in lake levels of the Laurentian Great Lakes, especially in Lakes Huron and Michigan, have significantly affected the availability and diversity of coastal habitats for plants, fish, and wildlife. This situation is likely to worsen in the next several decades because climate-change models predict a further drop in lake levels by 0.23 to 2.48 m for all five

Great Lakes (Mortsch and Quinn, 1996). There is therefore a critical need for development of multi-stressor models that can predict the response of these coastal communities to the simultaneous effects of lower water level, increased eutrophication and invasion by exotic species since many of these coastal marshes are located in areas that are already stressed by these factors due to increased urban and recreational development (Green Bay, Lake Michigan; Brazner and Beals, 1997; Georgian Bay, Lake Huron; Chow-Fraser 2006).

There are currently several models that can be used to predict the response of wetland communities to climate change. Keddy (1986) proposed a simple model relating the type of wetland vegetation (i.e. emergent, wet meadow) to water level fluctuations alone, while van der Valk (1981) proposed a model based on water-level fluctuations and life-history traits of different plant species under consideration (from seedbank/propagule information). However, as van der Valk pointed out, his model only predicted which species would be present, but not the relative abundance of species and their potential interactions. The predictability of the model will likely improve if other abiotic factors such as topography of the site and water quality, as well as the magnitude and type of biotic factors (e.g. abundance of invasive exotic species) are included.

The purpose of this study is to examine the simultaneous effects of multiple stressors (i.e. water level, human population growth and invasion of exotic species) on the distribution of the native cattail population in a Great Lake coastal marsh. Historical records show that there were extended periods during the late 1800s when water levels

were relatively high in Lake Ontario (see Chow-Fraser, 2005), and yet, there was no evidence of permanent damage to the marsh vegetation in the ensuing decades. This implies that factors other than water level can contribute to the long-term reduction of the emergent vegetation. Wei and Chow-Fraser (2005) have already demonstrated that increased urbanization in the Hamilton region contributed to loss of the emergent community as a whole in Cootes Paradise Marsh, independently of sustained high water levels. In this paper, we have expanded our investigation to examine these simultaneous impacts at the species level. Our hypothesis is that both the total percent cover of all wetland vegetation as well as that of a single species will be negatively correlated with water-level fluctuations. We further hypothesize that increased nutrient availability and increased organic content of sediments due to urbanization should favor the growth of exotic species (e.g. *Glyceria maxima*) over that of the native species (e.g., *Typha*) and that the synergistic effect of multiple stressors will be greater than that of any stressor alone.

METHODS

Study site

Cootes Paradise is a 250-ha coastal wetland ($43^{\circ} 16'N$, $-79^{\circ} 55'W$), located at the extreme west end of Lake Ontario (see Fig. 1-1 in Wei and Chow-Fraser, 2005). It is a drowned river-mouth marsh that drains into the west end of Hamilton Harbour, and has three main tributaries: Spencer Creek, Borer's Creek and Chedoke Creek. The marsh

receives runoff from adjacent agricultural, residential, industrial, commercial, and recreational lands, as well as effluent from the Dundas Sewage Treatment Plant. It is surrounded by the city of Hamilton to the south, and the towns of Ancaster, Dundas and Flamborough to the north and west.

During the 1920s and early 1930s, the most abundant emergent plant was *Typha sp.* (mostly *T. latifolia* with some *T. angustifolia*) and it was associated with many other common native wetland species such as *Sparganium sp.* and *Polygonum punctatum*. Like many other marshes in eastern N. America, Cootes Paradise has been invaded by Eurasian species that became established following European settlement. The first record of *Glyceria maxima* (manna grass) a Eurasian invasive species, was first collected from Cootes Paradise in the 1940s (Mills et al., 1993). By the early 1950s, it overtook *Typha* as the dominant emergent plant. Over the next 30 years, its distribution expanded quickly. Currently, marsh restoration efforts that are being implemented as part of the Remedial Action Plan for Hamilton Harbour (see Lougheed et al., 2004) are being severely challenged by the invasive species, including *Lythrum salicaria* (purple loosestrife), *Phragmites australis* (common reed), as well as *Glyceria*.

Data

Percent cover of the major emergent vegetation between 1946 and 1979 was taken from Chow-Fraser et al. (1998). These data were available in a GIS format and assembled from field vegetation maps that had been archived by the Royal Botanical Gardens

(Burlington, Ontario, Canada). The percent cover of six major taxa was calculated in ArcView GIS and is shown in Table 2-1. Water-level (WL) data for Cootes Paradise were obtained from Chow-Fraser et al. (1998). The percent cover of open water was calculated from the same digitized historical vegetation maps of Cootes Paradise. Mean water depths (Fig.2-1) for the major species during the years of observation were estimated from the digital elevation model of the marsh. Detailed information on the digital elevation model was documented in Wei and Chow-Fraser (2005).

Open-water area in a marsh is usually positively correlated with changes in water level. However, linear changes in water levels may not correspond linearly with changes in open-water areas due to spatial variation in marsh bathymetry (Table 2-2; Wei and Chow-Fraser, 2005). Since changes in water level will not reflect spatial variability in marsh topography, the percentage of open water (OP) was used as a surrogate to reflect both changes in water levels and site-to-site differences in marsh depth. We used historical census population data (POP) as a surrogate of potential cultural eutrophication (Wei and Chow-Fraser 2005; Fig. 2-2) because water-quality data were not available prior to 1975 (see Chow-Fraser et al. 1998). Data for the Hamilton-Wentworth region (including Hamilton, Dundas, Ancaster, and Flamborough) were obtained from Statistics Canada and Ontario Ministry of Revenue. Detailed description of data processing is documented in Wei and Chow-Fraser (2005).

Statistical analysis

Correlation and regression analyses were performed to assess the relationships among water level (WL), census population (POP), and percent cover for *Typha sp* (TL), *Glyceria maxima* (GM), and *Phragmites australis* (PA). To assess the relationships among native species, exotic species, and POP, we partitioned out the effect of water-level fluctuation by regressing percent cover of the major species and POP against OP, and used the residuals of these in corresponding regression models: e.g. to estimate the effect of GM on TL using residuals regression, we regress the residuals of the regression on TL on OP on the residuals of the regression of GM on OP (see Freckleton, 2002). The statistical analyses were performed in SAS JMP 4.0.

The spatial displacement of major species between 1946 and 1979 was performed through the Union process in ArcView GIS. The 1979 vegetation map was used as the input theme and the 1946 map as the overlay theme to produce a union theme that indicated the total replacements of a species by any other species in space.

RESULTS

Long-term change of emergent plants

Percent cover of the major species of emergent plants in Cootes Paradise Marsh for 1946, 1953, 1971, 1974, 1975, 1977, 1978, and 1979 is given in Table 2-1. In 1946, *Typha latifolia* and *Glyceria maxima* were the two dominant emergent taxa, accounting for 24.0 and 14.7 % of the total area of the marsh, respectively. By 1979, 95% of the *Typha*, and 27% of the *Glyceria* had been eliminated. Although there had been a

consistent overall decline in the cover of all emergent taxa, the loss was disproportionately higher for cattails than for manna grass. In 1953, *Glyceria* overtook *Typha latifolia* as the dominant species, and thereafter retained this dominance. Distribution of the other major species fluctuated over the years of observation, and did not reveal any trends.

Effects of multiple stressors on Cootes Paradise marsh

Relatively high water levels in Cootes Paradise Marsh have adversely affected the wetland since the 1930s (Chow-Fraser et al. 1998; Chow-Fraser, 2005). During 1946, average depths for the major emergent taxa (*Typha*, *Glyceria* and *Sparganium*) ranged from 3 to 6 cm, while that for the floating water lily, *Nymphaea tuberosa*, was 24 cm. In all subsequent years, water depths were uniformly higher for all species, with plants occurring in water as deep as 77 cm during the historic high water levels experienced in 1974 (Fig.2-1). In general, *Typha* appeared to tolerate deeper water than did *Glyceria*, and both appeared to be more tolerant of inundation than *Phragmites*, which were found in water depths ranging from mud flat to 39 cm in 1974. *Typha* and *Glyceria* both responded negatively to high water levels (Table 2-2). Up to 66 and 79% of the variation in percent cover of *Typha* and *Glyceria*, respectively, could be explained by the percent of open water (OP) in the marsh (Fig. 2-3).

We found that apart from the effect of high water levels, human population growth correlated negatively with *Typha* ($r^2 = 0.644$, $P = 0.0165$), but positively with *Glyceria* ($r^2 = 0.507$, $P = 0.0477$) (Fig. 2-4a and b, respectively).

In addition to the effects of high water levels and urbanization, we further investigated the possibility that distribution of native marsh species such as *Typha* could be displaced by invasive species such as *Glyceria* and *Phragmites*. Mixed patches of these species often occurred side-by-side within the marsh, and in shallow areas, *Typha* appeared to have been outcompeted by *Glyceria*. This observation is supported by the significant relationship between *Typha* and *Glyceria* after the effect of water levels has been partitioned out (Fig. 2-5a; $r^2 = 0.55$, $P = 0.0354$). By excluding the 1977 data point, we obtained a stronger negative relationship between *Typha* and *Glyceria* ($r^2 = 0.88$, $P = 0.0020$); therefore, high water levels and invasion of *Glyceria* had a synergistic impact on *Typha*: the combination of these two factors caused greater detrimental impact than either factor alone (Table 2-3). By comparison, there was a weak but not significant effect of *Phragmites* on the recovery of *Typha* (Fig. 2-5b; $r^2 = 0.38$, $P = 0.1047$), perhaps because *Phragmites* had only been a minor component in the marsh (Table 2-1). Nevertheless, *Glyceria* and *Phragmites* were positively correlated and both may have contributed to the overall reduction in *Typha* (Fig. 2-5c).

Spatial displacement of Typha and Glyceria

A change detection analysis clearly demonstrated that the invasive *Glyceria* had spatially displaced the native *Typha* between 1946 and 1979 (Table 2-4). Over this period, almost 60% of the *Typha* had been damaged by high water levels (14.1/24); 24% had been replaced by *Glyceria* (5.8/24), 13.7% by other species, while only 1.6% (0.8/24) remained in their original habitat. Throughout the period of observation, *Typha* only accounted for 2% (0.4/24) of newly colonized area. By comparison, 67% of the *Glyceria* present in 1946 had been damaged by higher water levels, only 3.4% (0.5/14.7) of this had been replaced by other species (of which only 1.4% was *Typha*), and close to 30% of the original growth remained in the same location. Overall, 44.2% (6.5/14.7) *Glyceria* invaded new habitat, primarily areas that had previously been occupied by *Typha* (41.4%). This high rate of replacement of *Typha* by *Glyceria* was supported by the partial correlation (Table 2-2): a negative correlation was found between *Typha* and *Glyceria* after having controlled for the effects of water level and human population growth.

Correlations between exposed habitats and response of emergent cover for 1971-1979

Typha in Cootes Paradise Marsh was mainly distributed in the west end of the marsh. The year-to-year regrowth of emergent plants could have been attributed to either vegetative structures or propagules in the seed bank or both, but unpublished field notes from the Royal Botanical Gardens archives reveal that most of the growth had been through rhizome production. This is consistent with results of a 1999 survey conducted by the Royal Botanical Gardens that confirmed the paucity of viable *Typha* seed in the

seed bank at the west end of the marsh. In this study, the realized habitats were estimated from field vegetation maps that did not distinguish between growth by rhizome or seedlings, and any exposed habitats were estimated from the digital elevation model in combination with the historical water level information. Although all three species were positively correlated with the increase in exposed habitats (Table 2-5), *Glyceria maxima* appeared to have benefited most from exposed mudflat during 1970s.

DISCUSSION

In Cootes paradise, emergent cover was found to correlate negatively with water depth, and increased water levels during the past 6 decades was reflected in the steady decline in the emergent marsh (Fig 2-2a), a pattern that has been well documented for many other coastal marshes of the Great Lakes (Hudon, 1998; Lyon et al., 1986; Williams and Lyon, 1997; Wilcox et al., 2003). Recently, we showed that both increased urbanization (indicated by human population census data in the Hamilton region) and water level had independent and negative effects on the community of emergent plants in Cootes Paradise Marsh (Wei and Chow-Fraser, 2005).

Sustained high water levels is the major factor that contributed to the decline of the emergent marsh in Cootes Paradise. Water level alone explained 88% of the variation in the distribution of *Typha*. Addition of human population growth as a variable to the regression equation further accounted for 11.3% (99.6% - 88.3%) of the remaining variation (Table 2-3). There is a positive correlation between water level and human

population census data (Table 2-2), and we therefore partitioned out the effect of water level and found that *Typha* was still negatively related to the increased human population. This is consistent with the view that both urban and agricultural development in the watershed contributes to wetland degradation by increasing the loading of nutrients, road salts, pesticides and heavy metals, and by altering the hydrologic regime (Crosbie and Chow-Fraser, 1999; Davis and Froend, 1999). In contrast to *Typha*, *Glyceria* was positively related to human population growth, once water-level effects had been partitioned out (Fig. 2-4b). This observation is consistent with Lambert (1947) that soil samples associated with *Glyceria* have relatively high concentrations of iron, phosphorus and nitrogen and stands of *Glyceria* in Britain were favored by application of manure. We therefore suggest that part of the reason for the successful introduction of *Glyceria* in Cootes Paradise is the increased nutrient availability resulting from urbanization. We also suggest that *Glyceria* and *Typha* have a similar niche (Mounford et al., 1993), and will compete for any habitat that is released by reduced water levels.

High water levels limited the distribution of both *Typha* and *Glyceria* (Fig. 2-3); however, once water levels receded, *Glyceria* appeared to be the more successful colonizer, as indicated by its rapid replacement rate (Table 2-4) and correlations between exposed habitats and response of major emergent plants (Table 2-5). The replacement of *Typha* by *Glyceria* especially in the shallow areas could have been due to 1) increased nutrient availability in the sediments and 2) differences in life-history traits. Increased nutrient availability in the sediments favoring *Glyceria maxia* over *Typha* is consistent

with results from Lambert 1947 and Weiher et al 1996. Weiher et al. found that *Typha* were unable to become established in enriched sediments. *Glyceria* appears to favor shallow water and begins growth earlier in the year (Buttery and Lambert 1965), thus making it superior to *Typha* when competing for newly released habitat. *Typha* and *Glyceria* were often segregated according to water depths, with *Typha* in deeper water and *Glyceria* in shallower water (Fig. 2-1). Another invasive species, *Phragmites australis*, has also colonized the marsh, and may further impact *Typha*, a scenario that is consistent with Wilcox et al.'s (2003) observation that *Phragmites* frequently displaced *Typha* in a Lake Erie marsh during periods of high water level.

It is important to note that it has been 25 years since the last complete vegetation survey has been conducted in Cootes Paradise Marsh. Even though the marsh had been surveyed in 1993 and 1999, different sampling protocols and classification schemes precluded their inclusion in this study. The major problem with the 1993 survey was that many of the plant taxa had been lumped together during the survey, while the major problem with the 1999 survey was that only the west end of the marsh had been mapped. Therefore, we recommend that a future survey be undertaken of the entire marsh, following the protocol and classification scheme used in the historic surveys to validate the results of this study.

An important implication of our results is that projected water level decline due to climate change may not necessarily favor the restoration of a desirable native marsh because of the presence of other disturbances such as exotic and invasive species and

altered nutrient regime. Based on the synergistic effect of multiple stressors, we predict that native marsh restoration will be difficult due to the present availability of more efficient colonizer species such as *Glyceria* and possibly *Phragmites*.

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Table 2-1. Long-term changes in the percent cover of the major species of emergent plants in Cootes Paradise Marsh during the years of observation.

Major species	1946	1953	1971	1974	1975	1977	1978	1979
<i>Typha latifolia</i> (TL)	24.0	4.0	3.6	0.3	0.4	0.6	1.7	1.2
<i>Glyceria maxima</i> (GM)	14.7	4.3	15.4	5.2	8.9	6.8	9.1	10.7
<i>Sparganium eurycarpum</i> (SE)	0.3	0.3	0.0	0.0	0.2	0.0	0.5	1.0
<i>Nymphaea tuberosa</i> (NT)	0.2	1.6	0.1	0.1	0.0	0.0	0.0	0.1
<i>Bidens cernua</i> (BC)	0.0	1.7	0.2	0.3	3.4	6.4	1.4	0.3
<i>Phragmites australis</i> (PA)	0.0	0.0	1.0	0.2	0.2	0.1	0.1	0.1

Table 2-2. Summary of correlation analyses for this study. Values in brackets are partial correlation controlling for water level, open water, and human population.

Variables	r	p
<i>Glyceria maxima</i> vs. <i>Typha latifolia</i>	0.554 (-0.8768)	0.154 (0.051)
<i>Glyceria maxima</i> vs. human population	-0.188 (0.827)	0.656 (0.042)
<i>Glyceria maxima</i> vs. water level	-0.724	0.042
<i>Glyceria maxima</i> vs. open water	-0.772	0.025
<i>Typha latifolia</i> vs. human population	-0.842	0.009
<i>Typha latifolia</i> vs. water level	-0.809	0.015
<i>Typha latifolia</i> vs. open water	-0.935	0.001
Water level vs. open water	0.942	<0.001
Water level vs. human population	0.514	0.192
Open water vs. human population	0.672	0.068

Table 2-3. Summary of multiple regression ($n = 7$; 1977 data excluded). TL = *Typha latifolia*; GM = *Glyceria maxima*; OP = open water; POP = human population census data.

Dependent variable	Term	Estimate	SE	t	p
TL ($R^2=0.996$)	Intercept	40.875	4.168	9.81	0.0023
	GM	-0.706	0.143	-4.93	0.0160
	OP	-0.797	0.076	-10.51	0.0018
	(GM-9.388)*(OP-73.038)	-0.043	0.013	-3.21	0.0490
TL ($R^2=0.996$)	Intercept	41.482	4.108	10.10	0.0005
	OP	-0.250	0.054	-4.64	0.0098
	POP	-0.532	0.066	-8.03	0.0013
	(OP-73.038)*(POP-37.725)	0.039	0.006	6.26	0.0033

Table 2-4. Spatial displacement of major species from 1946 to 1979. Values in bold correspond to percentage area that had been replaced by *Glyceria maxima* in 1979.

Displacement from 1946 to 1979
48.1 % open water was occupied by open water
14.1% <i>Typha latifolia</i> was occupied by open water
9.8% <i>Glyceria maxima</i> was occupied by open water
8.2% other species was occupied by other species
5.8% <i>Typha latifolia</i> was occupied by <i>Glyceria maxima</i>
4.2% <i>Glyceria maxima</i> was occupied by <i>Glyceria maxima</i>
3.6% other species was occupied by open water
3.2% <i>Typha latifolia</i> was occupied by other species
0.8% <i>Typha latifolia</i> was occupied by <i>Typha latifolia</i>
0.5% open water was occupied by other species
0.5% <i>Glyceria maxima</i> was occupied by other species
0.5% open water was occupied by <i>Glyceria maxima</i>
0.2% other species was occupied by <i>Glyceria maxima</i>
0.2% <i>Glyceria maxima</i> was occupied by <i>Typha latifolia</i>
0.2% open water was occupied by <i>Typha latifolia</i>
0.0% other species was occupied by <i>Typha latifolia</i>

Table 2-5. Summary of correlation analyses between percent cover of major taxa and exposed habitats during the 1971-1979 surveys.

	<i>Typha</i>	<i>Glyceria</i>	<i>Phragmites</i>	Exposed
<i>Typha</i>	1.00	0.91	0.85	0.26
<i>Glyceria</i>		1.00	0.80	0.35
<i>Phragmites</i>			1.00	0.24
Exposed				1.00

Figure 2-1. Summer mean water level and water depth range for the major species of emergent plants in Cootes Paradise marsh: a) water level, b) water depth range. The dashed line indicates the grand mean water depth.

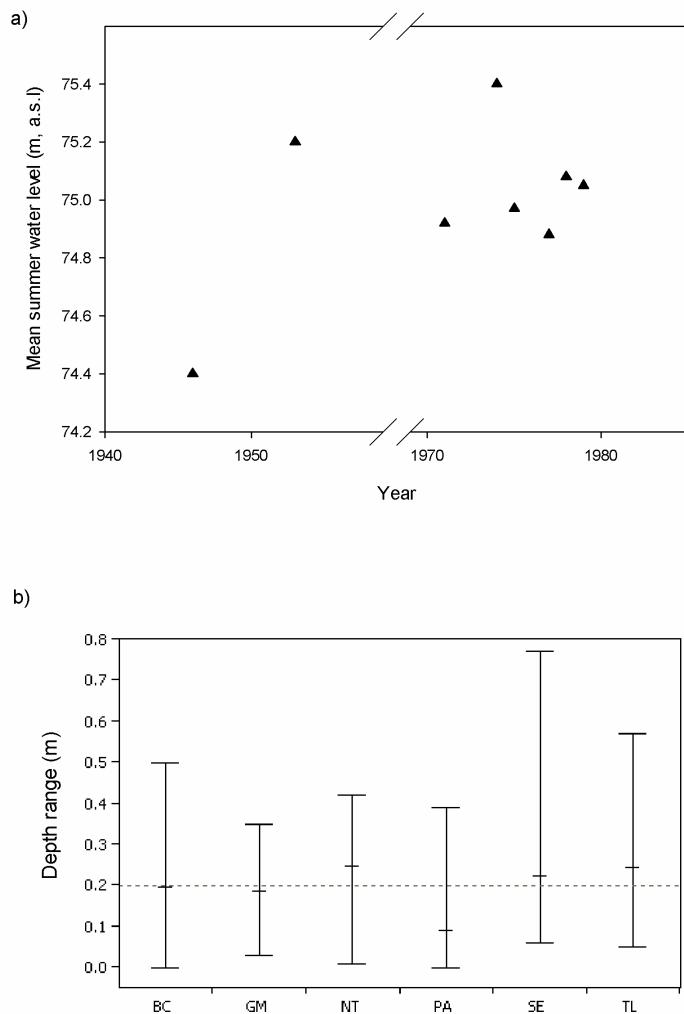


Figure 2-2. Relationship between percent total emergent cover vs. a) water levels in Cootes Paradise Marsh and b) human population census for the Hamilton-Wentworth Region from the 1930s to the 1990s

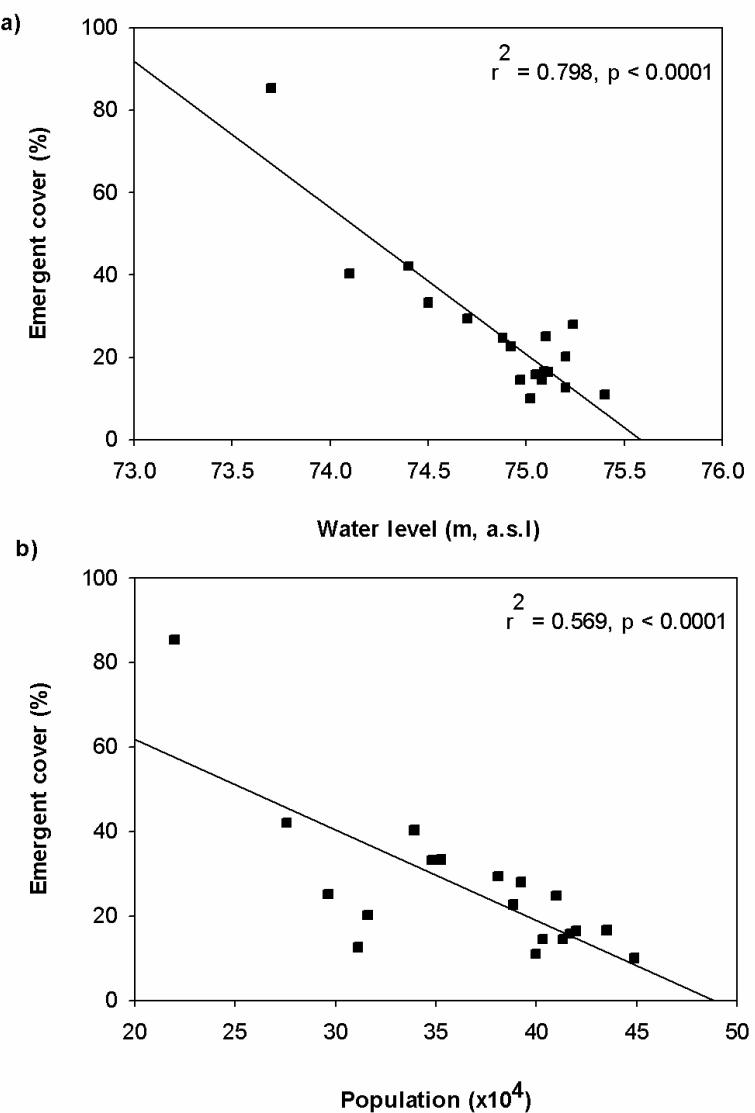


Figure 2-3. Relationship between a) *Typha* and b) *Glyceria* with percent open water for 1971-1979 (n = 6).

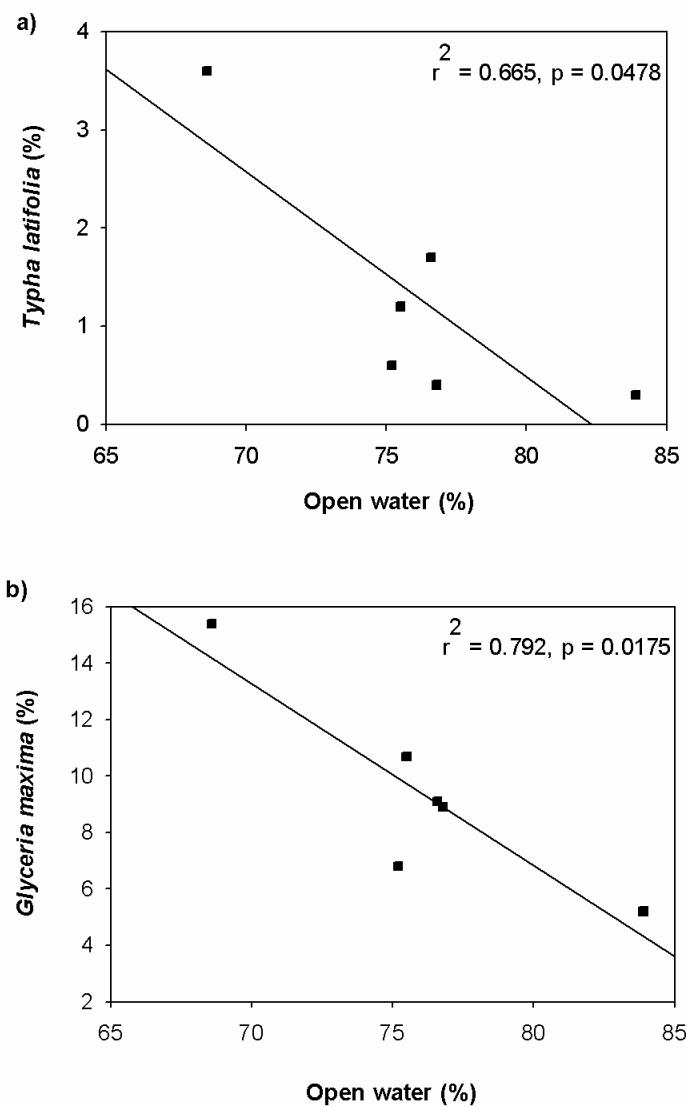


Figure 2-4. Relationship between residual of a) *Typha* and b) *Glyceria* with human population census (n = 8).

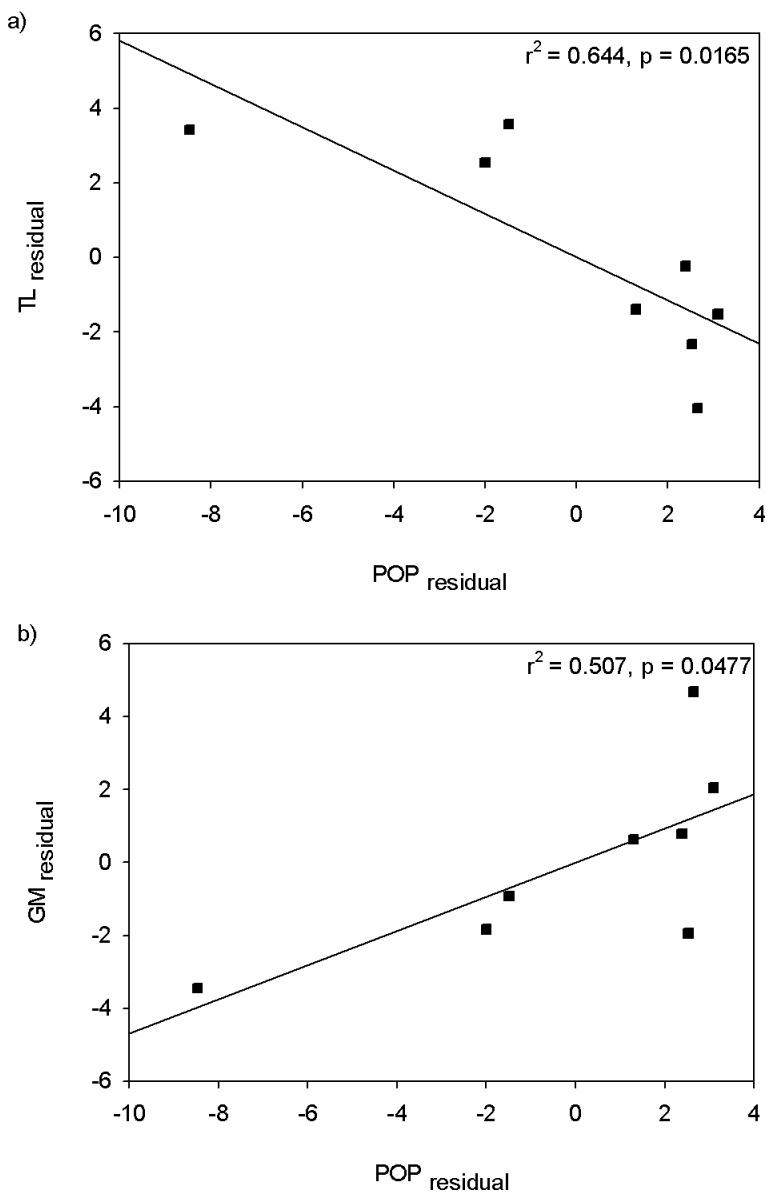
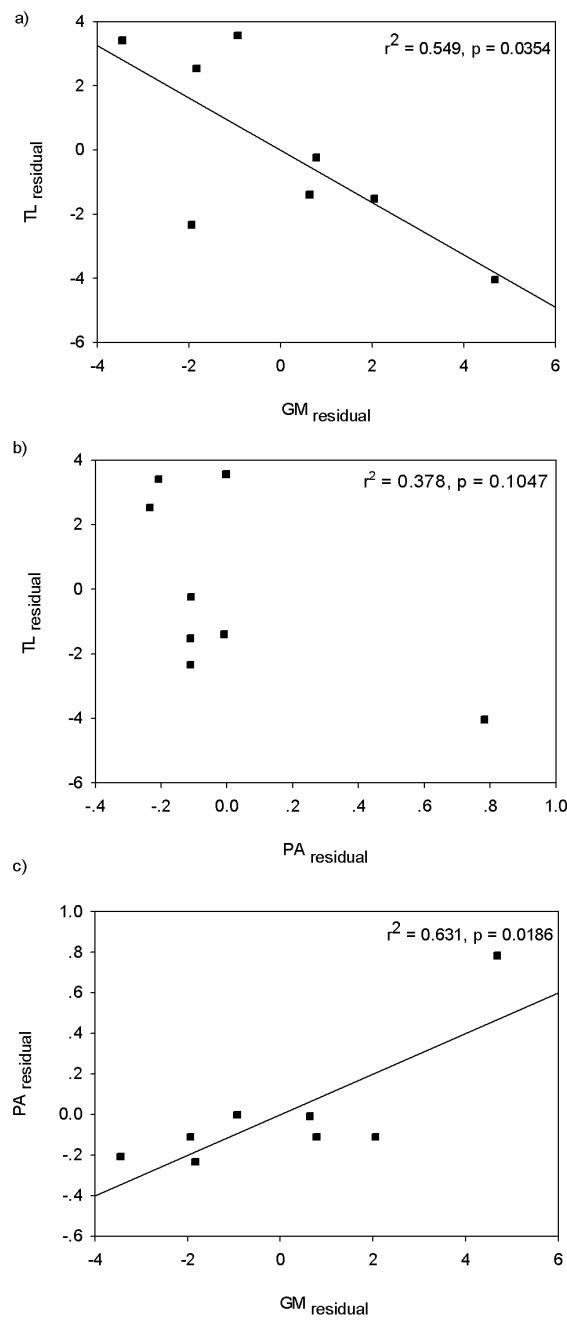


Figure 2-5. Relationship between a) residual of *Typha* and residual of *Glyceria maxima*, b) residual of *Typha* and residual of *Phragmites*, and c) *Phragmites* and residual of *Glyceria*. (n = 8).



CHAPTER 3:

A regional model of emergent vegetation in response to water level fluctuations in

Lake Ontario

Wei, A. and Chow-Fraser, P.

ABSTRACT

The effect of water-level fluctuations on both the structure and functioning of coastal marshes is well documented and, in the past, scientists have demonstrated this by relating historical changes in the areal cover of emergent vegetation of a particular site to corresponding water-level data. This approach of relating areal cover to water level cannot be applied to multiple sites from a region experiencing the same water level because, in that instance, water level would be a constant and can not be used as an explanatory variable for emergent cover. In a previous study of Cootes Paradise Marsh, we proposed the use of a Digital Elevation Model (DEM) to examine the effect of water level fluctuations on emergent plant cover over a 60-year period (1934-1993) and found that the inundated area was a better predictor of emergent cover than water level. However, this approach needs to be modified before it can be applied to multiple sites because a large wetland tends to have a large inundated area and the differences in wetland areas must be taken into account. In the present paper, we develop a regional model of vegetation response to validate the DEM-based method and to confirm the existence of a highly significant relationship between percent inundated area and percent emergent cover over a large spatial scale in eastern Lake Ontario. Additionally, we will show that this general relationship may be modified by the degree of urbanization in wetland watersheds. Our results suggest that DEM-based approach is useful for predicting the aggregate response of emergent vegetation to annual water-level fluctuations at both local and regional scales.

INTRODUCTION

Emergent marshes are a vital feature of Great Lakes coastal ecosystems because they are comprised of primary producers and provide food and shelter for diverse communities of waterfowl, fish, and invertebrates (Wei et al. 2004). Although coastal wetlands are known to be affected by multiple stressors, including, inter-annual changes in hydrology, nutrient and sediment enrichment, as well as bioturbation from introduced invasive species (Maynard & Wilcox 1996), the major natural disturbance affecting coastal wetlands is water-level fluctuation (Chow-Fraser et al. 1998; Wei & Chow-Fraser 2005, 2006). Indeed, these fluctuations are essential for maintaining the structure of wetlands through a well-established cycle of flooding and de-watering that results in contraction and expansion of emergent marshes, and thus promote the high biodiversity that characterizes coastal wetland ecosystems (e.g. Keddy & Reznicek 1986).

The response pattern of emergent marshes to water-level fluctuations has been extensively investigated throughout the Great Lakes System: St. Lawrence River (Hudon 1997), Lake Ontario (Chow-Fraser et al. 1998; Chow-Fraser 2005), Lake Erie (Gottgen et al. 1998) and Lake Michigan (Lyon & Drobney 1986). In the past, authors have tended to relate changes in areal cover of emergent vegetation (EM) to historical water levels (WL) to demonstrate the effect of WL fluctuations on the vegetation dynamics of a particular site. While this approach has been useful for examining effects of WL disturbance at a single site over multiple years, it cannot be applied to multiple sites from a region experiencing the same WL at a given time, because in the

latter case, WL is a constant, and cannot be used as an explanatory variable for EM. Another limitation of the traditional method is that it relied heavily on the use of long-term archives of aerial photographs or satellite data in order to statistically relate the historical changes in EM to WL fluctuations, and such historical resources are not always available or are difficult to assemble for a large geographical region.

Currently, there is a great need to develop predictive models to examine the effects of different water level regimes on emergent vegetation at a basin-wide scale (Wescoat et al. 2006). The recent drop in lake levels in Lakes Huron and Michigan to near-record-low levels has also increased the need to develop basin-wide forecasts of how wetlands in these middle lakes will respond to further declines in lake levels with anticipated climate change scenarios (Magnuson et al. 1997). However, predictive models based on WL would be inappropriate for such basin-wide forecasts. Williams (1995) provided a case study in which he proposed a basin-wide model to predict the response of EM to annual WL fluctuations. He hypothesized that the rates and magnitude of EM response to WL fluctuations would be similar throughout the Great Lakes. In other words, there should be a single model containing a basin-wide response component, and an unexplained component of local changes. He tested this hypothesis by running regressions of emergent area on long-term historical water levels for nine wetlands across the five Great Lakes basins. As Williams pointed out, although such a model was statistically significant, an unacceptably high unexplained error existed and he concluded that a better

approach needed to be developed to account for differences in response rate and magnitude for individual wetlands among the different Great Lakes.

In the present paper, we propose a solution that combines WL information with site-specific depth variation to investigate the response of EM to different water-level regimes at a regional scale. This solution stems from a previous study (Wei & Chow-Fraser 2005), in which we proposed the use of Digital Elevation Models (DEM) to examine the effect of long-term (1934-1993) WL fluctuations on the emergent vegetation of Cootes Paradise Marsh, a degraded freshwater marsh of Lake Ontario. We used the DEM to calculate a new term, called “Inundated Area” (IA), which approximated the amount of land in the wetland that is submerged under water. We found that % observed EM was significantly and negatively correlated with IA, after controlling for the effect of WL fluctuations. However, there was no significant correlation between % observed EM and WL after controlling for IA. We also found that urbanization surrounding the site negatively impacted Cootes Paradise Marsh (Wei & Chow-Fraser 2005). However, the approach described in Wei and Chow-Fraser (2005) needs to be modified before it can be applied to multiple sites because a large wetland tends to have a large inundated area and the differences in wetland areas must be taken into account. Here, we propose to a regional model to predict the aggregate response of emergent cover over a large spatial scale in eastern Lake Ontario. We tested this by examining unpublished data from ten sites in eastern Lake Ontario while taking into account the potential effect of urbanization. Due to its global relevance and

applicability to other geographical areas, we believe that the proposed approach will enhance efforts to assess the effect of water-level fluctuation on emergent marshes from a site-specific scale to that of an entire lake throughout the Great Lakes basin.

METHODS

Study area

Two independent data sets from western and eastern portions of Lake Ontario, corresponding to local and regional scales, respectively, are included in this study. Data for the study at the local scale came from Cootes Paradise Marsh, a 250-ha drowned river-mouth wetland located at the western end of Hamilton Harbour (Chow-Fraser et al. 1998; Wei & Chow-Fraser 2005) (Fig. 3-1). The marsh is surrounded by the city of Hamilton to the south, and the towns of Ancaster, Dundas and Flamborough to the north and west. Together, the amount of urbanized land accounted for 20% of the drainage basin, and is therefore regarded as a highly urbanized coastal marsh. Although it receives runoff from adjacent agricultural, residential, industrial, commercial, and recreational lands, as well as effluent from the Dundas Sewage Treatment Plant, the major disturbance is inter-annual and seasonal fluctuations in water level of Lake Ontario (Chow-Fraser, 2005; Wei & Chow-Fraser, 2005).

Data for the regional scale came from ten sites located within a 28 km-stretch of eastern Lake Ontario shoreline (see Fig. 3-1). The mean summer water levels along the shoreline of eastern Lake Ontario were nearly identical and the difference in mean WL recorded at the Oswego and Cape Vincent gauge stations was less than 1 cm (*t*-test based

on the daily summer WL for 1978: two-tailed $p = 0.79$, $n = 124$). This allowed us to test the effects of inundated area (% IA) and land-use alteration on the percent observed emergent cover (% observed EM) while ‘controlling’ for WL. These sites were also chosen because the amount of urbanized land in wetland watersheds were low (approximately 2%) relative to that of Cootes Paradise Marsh’s 20%.

Data Sources

(1) Cootes Paradise Marsh

The DEM was produced from Painter et al. (1989)’s contour map and a DEM provided by the Ontario Ministry of Natural Resources (OMNR; M. Robertson, Peterborough, Ontario, Canada). Since the contour map only provided depth information for 75.06 m above sea level, and site elevations greater than 75.06 m were not available, a seamless merge with OMNR’s elevation points was required. We digitized the contour map using its scanned map, and then re-sampled 2300 water depth values from the contour map. The overlapping area was removed from OMNR’s DEM before it was merged with the re-sampled points from the contour map. The final DEM was interpolated from the combined elevation points using Spatial Analyst in ESRI ArcGIS 8.1 (Wei & Chow-Fraser 2005). The observed percentage cover of emergent vegetation (EM) between 1934 and 1993 were taken from Chow-Fraser et al. (1998). These data were obtained from several sources: historical aerial photographs and

historical vegetation maps stored in the Royal Botanical Gardens. The historical vegetation maps for years of 1946, 1953, 1971, 1974, 1975, 1977, 1978, and 1979 were available in a GIS format. Historical census population data were obtained from Statistics Canada and Ontario Ministry of Revenue and used as a surrogate of urbanization (Wei and Chow-Fraser 2005).

(2) Coastal wetlands of Eastern Lake Ontario

Emergent-cover data for coastal wetlands of the ten eastern Lake Ontario were downloaded from the website (<http://www.nwi.fws.gov>), which is operated by the U.S. Fish and Wildlife Service. These data are part of the National Wetland Inventory (NWI) and contain digitized maps produced from aerial photos taken in 1978, corresponding to the U.S. shoreline of eastern Lake Ontario. These maps included vegetation zones, identified as shrubs, emergent vegetation, aquatic beds, etc without any detailed taxonomic information. In addition, open-water areas were also identified so that the total amount of wetland area ($TOTAL_{NWI}$) can be calculated by summing areal cover of vegetation types and open-water areas. Detailed information on the NWI wetland maps has been documented in Tiner (1999).

The DEM data were obtained from the Cornell University Geospatial Information Repository. Appropriate water levels were obtained from the National Oceanic & Atmospheric Administration (NOAA). Because human population data within each wetland watershed were not available, we used land-use type as surrogate of urbanization.

With the aid of land-use maps, we classified those wetlands adjacent to urbanized areas as “urbanized” wetlands, while the rest were classified as non-urbanized wetlands.

Digital Elevation Model and Inundated Area

The core of our new approach is a DEM. A DEM has been defined as a “digital model of landform data represented as point elevation values” (DeMers 2003). In a Geographical Information System (GIS), DEM is a digital map of elevation values over a two-dimensional grid (see Fig. 3-2). Fig. 3-2 is an illustration of how to calculate “Inundated Area” (IA) with a DEM. IA is the total number of grid cells or *pixels* in a wetland that are submerged under water (all shaded pixels in Fig. 3-2). Inundated area can be calculated from DEM in a GIS by selecting pixels that are below a specified water level and multiplying the number of selected pixels by unit area per pixel. It can be conceptualized as the amount of “open water”, as well as a small portion of the emergent cover since emergent plants often grow from the edge of upland vegetation into shallow water. Because we used the mean summer WL to calculate IA, the corresponding IA reflected the average amount of inundated areas for the sites.

Fig. 3-2 also shows how to use a DEM to derive the upper wetland boundary for the purpose of determining total wetland area. Conventionally, plant cover, soil type, and long-term hydrological information (including water marks, visual observations of inundation) have all been used to delineate wetlands from upland areas, and this is the

method used to delineate the boundary for the NWI wetlands (see Tiner (1999) for a comprehensive review). By contrast, the wetland boundary delineated from DEM (shown in Fig. 3-2; $TOTAL_{DEM}$) was determined by the highest historical water-level data point. It is therefore different from that based on field observations, and we caution against using DEM to delineate wetland boundary for anything other than to estimate % IA. In this study, all our wetlands were protected riverine wetlands (see Albert et al. 2005) that were connected to Lake Ontario through narrow openings so the outer boundaries can be placed naturally at the openings. Other DEM methods must be used to determine the outer boundary of exposed lacustrine wetlands that considers both the site geomorphology as well as the maximum depth of plant colonization by aquatic plants.

WL versus DEM-based approaches

The major difference between these two approaches is that the WL approach uses water level alone to predict emergent plants, and thus ignores site-specific spatial variation, while the DEM-based approach uses inundated areas to make a prediction. The following two scenarios further demonstrate the differences between these two approaches.

(1) Single site scenario

If wetland morphometry resembles that of a wash tub, with regular contours, and sides that slopes linearly with depth (see Fig. 3-3A), then there would be no difference in response of EM between these two approaches. This is because the amount of change in % IA will be perfectly correlated with WL. In the real world, however, a wetland seldom assumes this shape, but has slopes that change more or less non-linearly (Fig. 3-3B). The non-linear relationship between IA and WL for Cootes Paradise Marsh, and underscores the reason why use of WL alone to predict the response of % EM would ignore important spatial variation in depth within the wetland.

(2) Multiple site scenario

If the objective is to study the response of emergent cover to annual water level fluctuations at a regional scale, data from multiple sites have to be used. It is common that sites from the same sub-basin (e.g., east end of Lake Ontario or west end of Lake Ontario) have the same or similar mean water level. In this case, relating the same or similar WL regimes to different amounts of emergent cover is meaningless. The same change in WL may correspond to vastly different change in IA and hence, different potential habitat for emergent plants due to differences in wetland bathymetry (Fig. 3-4). Because IA reflects site-specific variation in depth, change in inundated area can still be used to detect differences in emergent cover among several sites that are experiencing the same WL.

Steps in DEM-based approach

- (1) Obtain digital elevation model(s) for study sites.

Some governmental agencies have published DEM data for public use (e.g., Ontario Ministry of Natural Resources and United States Geological Survey). If such data are not available for the target sites, water depths and geographical coordinates have to be measured in the field and then the measured point values can be interpolated in a GIS to produce DEM for the study sites (see Wei & Chow-Fraser 2005).

- (2) Collect water level data.

- (3) Calculate IA for the sites using mean summer water levels (see Fig. 3-2).

- (4) Derive total wetland area using the DEM-based approach outlined above ($TOTAL_{DEM}$; see Fig. 3-2). Note that this is independent of the method based on field observations as discussed earlier and will yield a value different from that of $TOTAL_{NWI}$.

- (5) Calculate the proportion of IA to $TOTAL_{DEM}$ (i.e. % IA). Note that we calculated % IA by dividing IA by the total wetland area derived by DEM rather than that by field observations (i.e. $TOTAL_{NWI}$), to avoid a potential spurious relationship introduced by a shared term (i.e., X/Z vs. Y/Z) (Jackson & Somers 1991; Brett 2004). The conversion of IA to % IA for the regional study is necessary because a large wetland tends to have a large inundated area and data standardization is required to adjust for differences in total

wetland area. Because there is a linear relationship between IA and % IA and the intercept is 0, conditions that are restrictive for a ratio to remove the size effect (Jackson & Somers 1991), use of % IA will not introduce a spurious relationship while removing the area effect.

- (6) Run a regression with permutation test to examine the relationship between % EM and % IA.

Some other calculations in this study are summarized as follows:

(a) $\text{EMERGENT}_{\text{potential}} = (\text{TOTAL}_{\text{DEM}} - \text{IA}_{\text{MIN}}) + \text{MOD}$

where $\text{EMERGENT}_{\text{potential}}$ is the maximum theoretical habitat occupied by emergent vegetation, $\text{TOTAL}_{\text{DEM}}$ is the total wetland area defined by DEM-based wetland delineation, IA_{MIN} is the inundated area corresponding to the minimum summer WL, and MOD is the amount of shallow-water habitat occupied by emergent vegetation, calculated differently for Cootes Paradise and eastern Lake Ontario wetlands. For Cootes Paradise Marsh, MOD included all areas shallower than 50 cm as a part of the potential EM habitat. For eastern Lake Ontario, we could only include areas shallower than 15 cm due to lack of elevation values for areas deeper than 15 cm in the DEM. In essence, this variable is a measure of habitat defined by the lowest and highest water mark.

(b) $\% \text{EMERGENT}_{\text{potential}} = (\text{EMERGENT}_{\text{potential}} / \text{TOTAL}_{\text{DEM}}) \times 100$

(c) $\% \text{EMERGENT}_{\text{observed}} = (\text{EMERGENT}_{\text{observed}} / \text{TOTAL}_{\text{field}}) \times 100$

where $\text{EMERGENT}_{\text{observed}}$ is the observed areal cover of emergent vegetation determined from vegetation maps and/or aerial photos, and $\text{TOTAL}_{\text{field}}$ is the total wetland area calculated from either the NWI or Cootes database.

Data Analysis

We used ESRI ArcGIS 8.1 to estimate IA by applying corresponding WL data to DEMs of Cootes Paradise and eastern Lake Ontario. Regression with permutation test was performed in Regressn (Program for multiple linear regression with permutation test by Pierre Legendre, Universite de Montreal). To control for the presence of spatial autocorrelation and the spatial distances among the sites, partial Mantel test was used and performed in PASSAGE v 1.1.

RESULTS

Effect of % Inundated Area (IA)

Wei & Chow-Fraser (2005) indicated that Inundated area (IA) alone explained 83.1% of variation in the percent observed emergent cover in Cootes Paradise Marsh. To make it directly comparable to % IA calculated for wetlands from eastern Lake Ontario, we re-analyzed the data from Cootes Paradise Marsh and converted IA to % IA by dividing IA by wetland boundary derived from the DEM (Fig. 3-5a, $r^2 = 0.831$). In the case of a single site study such as Cootes Paradise Marsh, conversion of IA to % IA did not change the response of observed EM ($r^2 = 0.831$).

As mentioned from the outset, the relationship between WL and % observed EM cannot be established for multiple sites in a large geographical area when they experience the same summer mean water level or similar water level with little variation. However, corresponding IA at these sites are likely to be very different due to variation in site topography, and could be used to predict the response of EM. To test the use of IA at a regional scale, we calculated % EM_{observed} using NWI digital maps, and % IA using the DEM-derived wetland area for ten coastal wetland complexes located on the shoreline of eastern Lake Ontario. Theoretically, % observed EM and % IA should be negatively correlated because as the proportion of IA in a wetland increases, proportionately greater hydrological stress will limit the colonization of emergent plants, and the % observed EM will be smaller. Percent observed EM in these marshes was regressed against corresponding % IA to test this assumption. We found a significant negative relationship between % IA and % observed EM ($r^2 = 0.538$, $n = 10$, $p = 0.018$), confirming that % IA is a good predictor of the proportion of emergent vegetation in coastal wetlands that share the same or similar WL.

For other wetland systems where wetland boundaries are difficult to delineate based on field observations, the potential emergent habitat based on DEM and WL information could be used to predict observed % EM. The potential habitat area, which we have called Emergent_{potential}, can be roughly estimated by determining the area of the marsh between the high and low watermarks (see Methods), and is conceptualized as the amount of habitat released by low water levels. The relationship between the % observed EM

(% Emergent_{observed}) and % potential habitat area for emergent plants (% Emergent_{potential}) is shown in Fig. 3-6. Theoretically, the % Emergent_{observed} should be equal to or less than % Emergent_{potential} because not all potential area can be occupied by emergent plants due to factors such as substrate quality, availability of seeds or reproductive propagules. Five values for Emergent_{potential} in Fig. 3-6b were underestimated because the DEM for eastern Lake Ontario did not include elevations below 75 m.a.s.l, and this resulted in underestimating some potential areas for EM. Since both % IA and % Emergent_{potential} are based on calculated inundated area.

Effect of urbanization

Table 3-1 shows that the effects of urbanization on emergent vegetation vary according to urbanization level. For Cootes Paradise Marsh, inclusion of human population as a surrogate for urbanization explained another 4% of variation in % Emergent_{observed} and the effect of urbanization was highly significant (Table 3-1a). For eastern Lake Ontario wetlands, we used land-use type (urban vs. other) as a surrogate for urbanization because human population data were not available, and found a significant relationship between % Emergent_{observed} and % IA, but no significant effect of urbanization (Table 3-1b).

Spatial autocorrelation

Many studies indicate that ecological data tend to display some extent of spatial autocorrelation. Because the ten sites were centered on the eastern portion of Lake Ontario, we tested for potential spatial autocorrelation to make sure that the significant relationships between % Emergent_{observed} and % IA was not due to spatial autocorrelation in the data. We performed a partial Mantel test to re-examine the relationship between the two variables with the geographical coordinates as a third matrix and found that the relationships between % Emergent_{observed} and % IA was still significant after controlling for spatial autocorrelation in the data (partial Mantel $r = 0.4098$, two-tailed $p = 0.021$, 999 iterations).

DISCUSSION

In an earlier study, Wei & Chow-Fraser (2005) demonstrated that inundated area, IA, was significantly better than water level for predicting the percent emergent cover of a single site, Cootes Paradise Marsh. In the present study, we have taken this approach and applied it successfully to the eastern Lake Ontario region. This method confers a substantial advantage over the conventional approach of relying on WL alone to predict % observed EM, because wetlands experiencing very similar water-level conditions may have different amounts of inundated area that reflect differences in bathymetry; thus, the same change in WL may correspond to different amounts of habitat for emergent plants (Fig. 3-4). In addition, the traditional method of estimating emergent vegetation cover from archived aerial photos and satellite images (e.g. Ozemi & Bauer 2002; Wilcox et al.

2003) requires access to such media that are not always available for the period of interest.

The general relationship between % observed EM and % IA appears to be modified by the degree of urbanization in wetland watersheds. We found that for a single wetland, high rate of urbanization in combination with inundation can induce a significantly greater negative impact on the emergent marsh over and above that caused by inundation alone (Table 3-1). Cootes Paradise Marsh is located in a highly urbanized region on the Canadian side of Lake Ontario. The urbanized area within its watershed has increased to 20% of the total watershed area since European settlement. Prior to the 1930s, almost 100% of Cootes Paradise Marsh was covered with emergent plants, but sustained high water depths from the 1930s to the 1980s, combined with the stress from increasing human population, have contributed to a dramatic decline of emergent cover to 10% by the 1990s. Many studies indicate that urbanization increases runoff and increased runoff often result in increased loadings of nutrients, toxic substances such as heavy metals, pesticides, oils, road salts, and detergents into waterways (e.g. Wang et al. 2000) and the altered environment as a result of urbanization may have detrimental impacts on aquatic vegetation (Owen, 1999). Because of the negative impacts of urbanization on emergent plants, the recovery rate of native emergent plants in a highly urbanized watershed (e.g. Cootes Paradise Marsh) will be lower than that in a primarily forested watershed once water level declines. In contrast to Cootes Paradise Marsh, the urbanization within the watersheds of the ten sites from eastern Lake Ontario is relatively

low, approximately 2%. Because of this, the impact of urbanization on the emergent plants of these wetlands was not significant (Table 3-1). This is consistent with the view that when urban land use increases beyond a small percentage of the total land cover (e.g. 5%), the impacts of urban land use dominate over other land use impacts (Baker 2003).

Our finding that there is a global pattern in the aggregate response of emergent vegetation to water level disturbance has important implications for wetland research and management. Whereas in the past, digital bathymetric data were more difficult to obtain than historic wetland maps, it will soon become the reverse as more and more government-sponsored digital libraries become available on line. This will make it a relatively simple task to calculate inundated area, the potential habitat for emergent plants in any wetland, along with the amount of urbanized area in its associated watershed. The generalized approach that we have provided here could become a very useful management tool to assess the basin-wide response of emergent marshes to water-level disturbances caused either by lake-level regulation or by global climate change scenarios.

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Table 3-1. Summary of multiple regressions relating % Emergent_{observed} to % IA, land-use change and human population census. Asterisks indicate significance level at p-permutation < 0.05 level.

(a) Highly urbanized Cootes Paradise Marsh ($R^2 = 0.873$, $n = 20$, $p = 0.001$)

Term	Estimate	Std Error	t Ratio	Prob> t
Intercept	91.90995	10.20492		0.00000 *
% inundated area	-0.47979	-6.40021	0.00100 *	0.00000 *
Human population	-0.74646	-2.38708	0.01500 *	0.01444 *

(b) Low urbanized wetlands in eastern Lake Ontario ($R^2 = 0.599$, $n = 10$, $p = 0.043$)

Term	Estimate	Std Error	t Ratio	Prob> t
Intercept	97.21887	4.96173		0.00082 *
% inundated area	-1.16925	-2.63502	0.01100 *	0.01683 *
Land use	24.45127	1.03678	0.16300	0.16716

Figure 3-1. Map of Lake Ontario, showing location of study sites in the eastern portion of the lake, as well as location of Cootes Paradise Marsh and the two water-level gauges at Oswego and Cape Vincent. Location of major cities in Canada and the U.S. are also indicated.

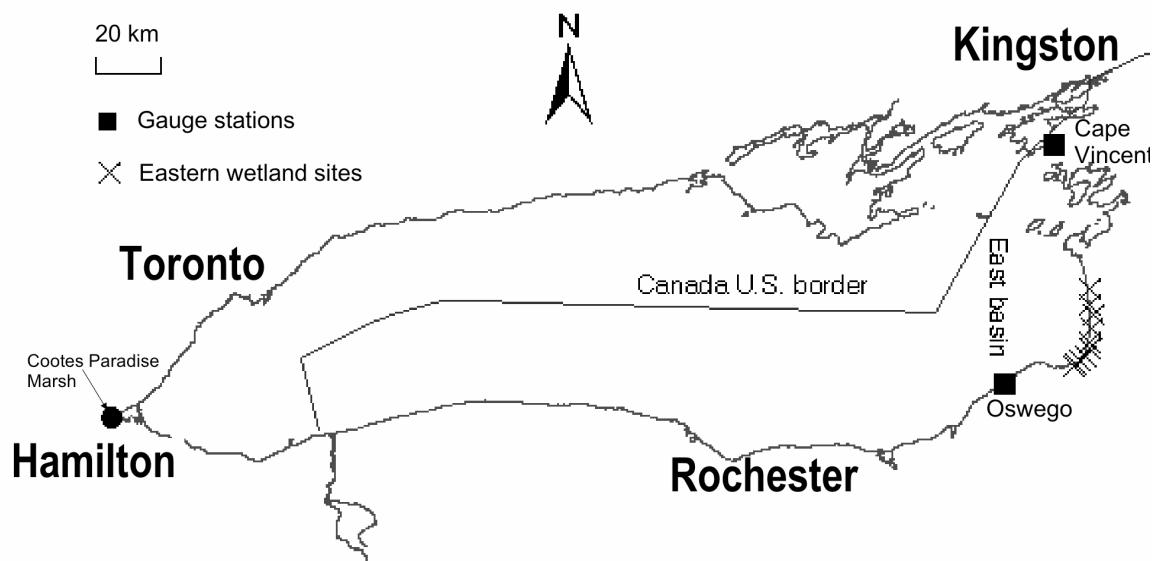


Figure 3-2. A simplified digital elevation model (DEM) of a wetland. In this model, the wetland is divided into regular grid cells or pixels. Values in the model represent elevations above sea level for the pixel. Shaded pixels in (A) and (B) represent inundated areas at different WL, with WL in Example A being lower than that in Example B. Inundated area (IA) can be calculated in a GIS by selecting pixels that are below a specified WL and multiplying the number of selected pixels by the unit area of the cell. The highest WL data point available is used to delineate the simplified wetland boundary for the model (shown with pixel borders in bold). The area enclosed by the wetland boundary is the total area of the wetland ($TOTAL_{DEM}$ in equations A and B; see Methods).

(A)

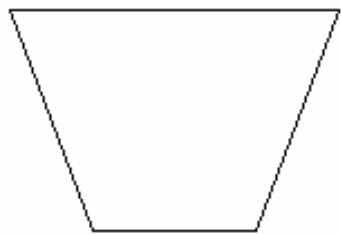
4	4	4	4	4	4	4	4	4
4	3	3	3	3	3	3	3	4
4	3	2	2	2	2	3	4	
4	3	2	1	1	2	3	4	
4	3	2	1	1	2	3	4	
4	3	2	1	1	2	3	4	
4	3	2	2	1	2	2	4	
4	4	3	2	1	1	2	4	
4	4	4	3	2	1	2	4	
4	4	4	4	4	1	4	4	

(B)

4	4	4	4	4	4	4	4	4
4	3	3	3	3	3	3	3	4
4	3	2	2	2	2	3	4	
4	3	2	1	1	2	3	4	
4	3	2	1	1	2	3	4	
4	3	2	2	1	2	2	4	
4	4	3	2	1	1	2	4	
4	4	4	3	2	1	2	4	
4	4	4	4	4	1	4	4	

Figure 3-3. Wetland cross sections showing (A) regular contours and (B) irregular contours.

(A)



(B)

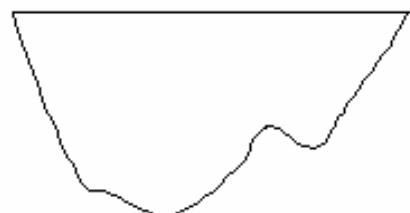


Figure 3-4. Comparison of inundated area (IA) and potential habitat area for emergent plants (EM) in two wetlands that are experiencing the same water level, showing that the same change in water level corresponds to different change in IA and EM.

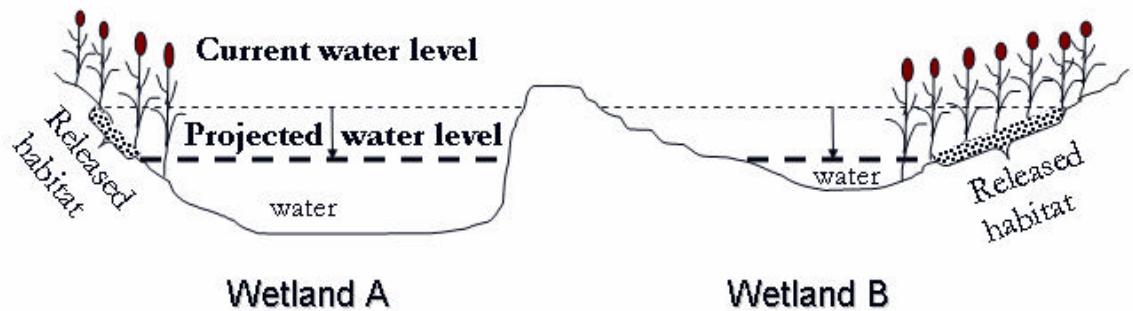


Figure 3-5. Regressions of % observed EM against % IA for (a) Cootes Paradise Marsh and (b) Eastern Lake Ontario wetlands

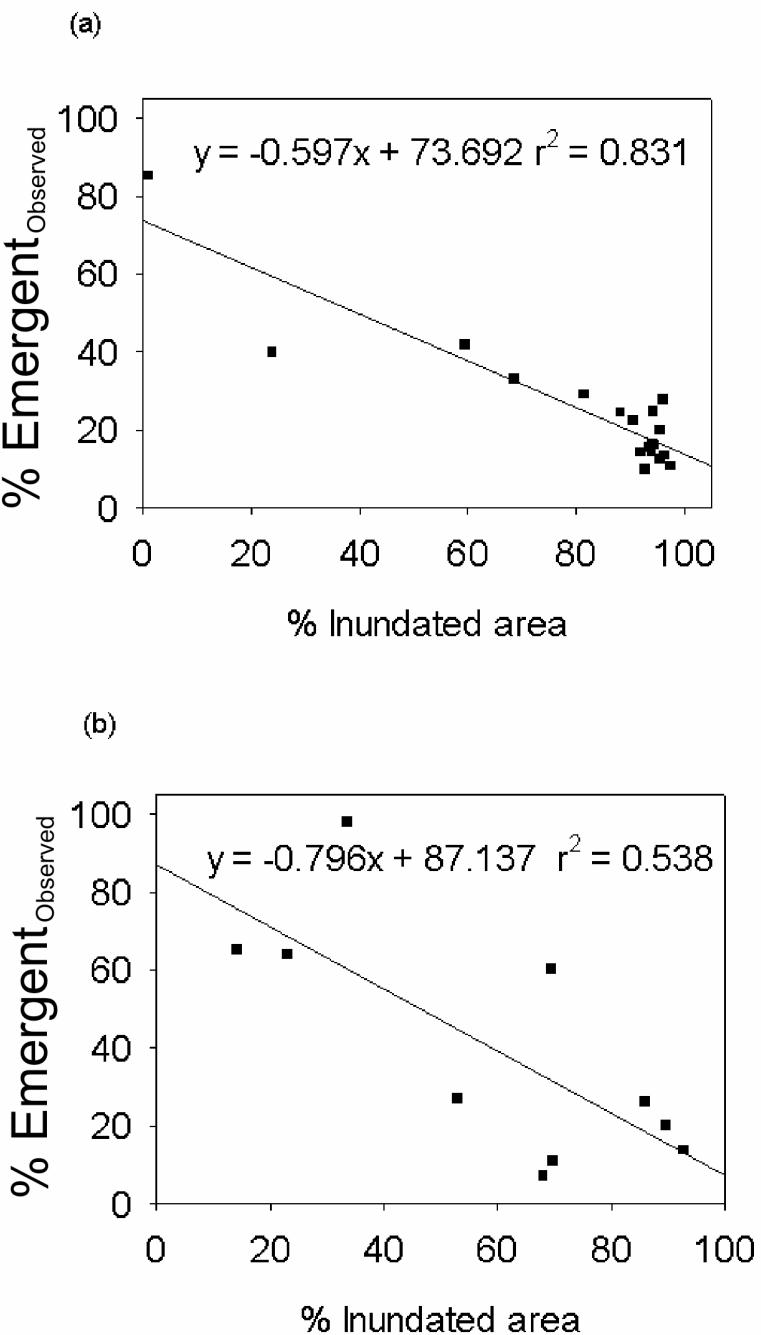
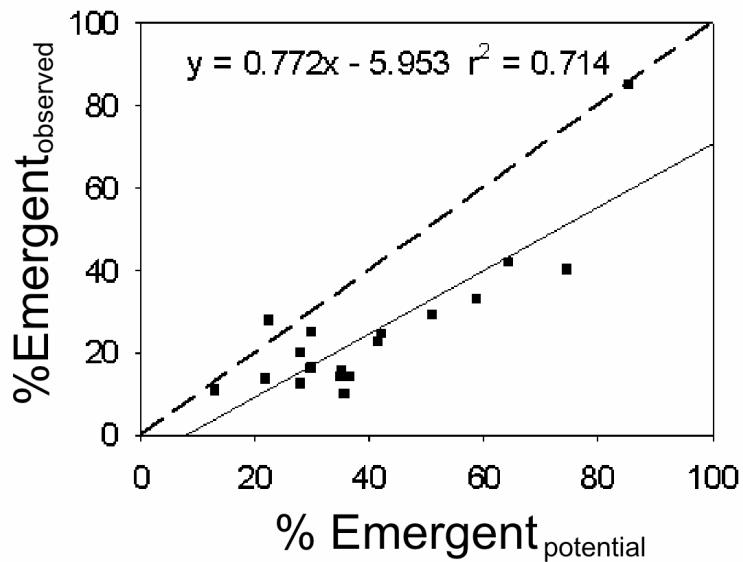
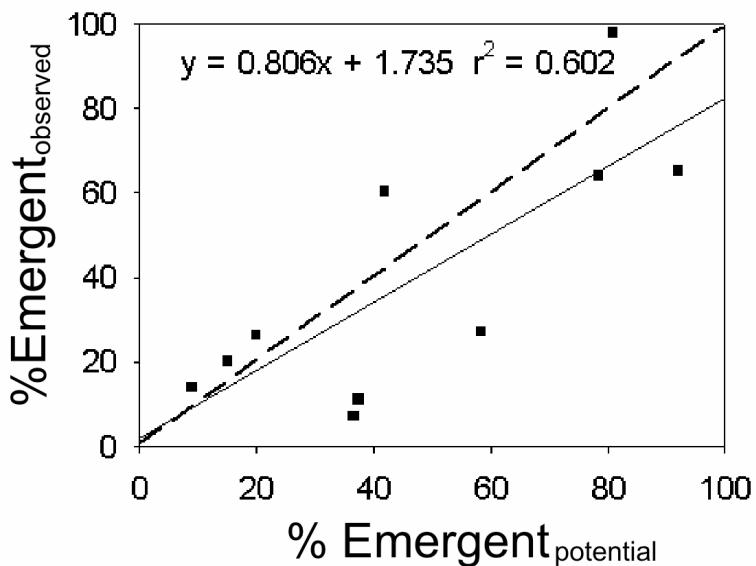


Figure 3-6. Relationship between % observed emergent vegetation (% Emergent_{Observed}) based on vegetation maps and % potential habitat occupied by emergent vegetation (% Emergent_{Potential}) (based on DEM information) for (a) Cootes Paradise and (b) Eastern Lake Ontario. The broken line shows $y = x$. The solid line is the regression line for the data points. In both cases, there is a general trend towards lower than expected observed EM cover based on water depth information (i.e. application of WL to DEM data).

(a)



(b)



CHAPTER 4:

Influence of shoreline features on fish distribution in the Laurentian Great Lakes

Wei, A., Chow-Fraser, P. and Albert, D. 2004.

Canadian Journal of Fisheries and Aquatic Sciences 61: 1113-1123.

ABSTRACT

In this paper, we used assembled fish distributions (over 9,500 field observations) and correlated them with 11 categories of the Great Lakes shoreline (i.e., bedrock, bluff, coarse beach, sandy beach-dune, sandy-silty bank, clay bank, low riverine-coastal plain, composite, wetland, and artificial, and unclassified) to validate the appropriateness of classifying Great Lakes fishes into three species-complexes (taxocenes) that account for differences in their dependence on shoreline features. A chi-square goodness-of-fit test with Bonferroni correction indicated a significant positive association between the presence of fish and three shoreline classes: wetland, sandy beach/dune, and bluff. The Dutilleul modified t-test was used to quantify the correlation between wetlands and distribution of the 25 most abundant species, and those of different functional groupings. Our results confirm that (a) the Great Lakes fish community utilize certain shoreline features (especially wetlands) disproportionately to their availability; (b) the distribution of wetland-associated taxa is influenced by wetland type (i.e., protected embayment versus open-shoreline wetland); and (c) the preferred utilization of coastal wetlands by a majority of the fish community is consistent across geographical scales, from the local site to the entire Great Lakes shoreline.

INTRODUCTION

Coastal wetlands of the Great Lakes provide important spawning and nursery habitat for wetland-dependent species that include a number of commercially and recreationally important taxa (Chubb and Liston 1986; Stephenson 1990; Jude and Pappas 1992). The U.S. Nature Conservancy estimated that about 80% of the approximately 200 fish species found in the Great Lakes use the near-shore areas for at least part of the year and directly depend on coastal wetlands for some part of their life cycles (Chow-Fraser and Albert 1999). Jude and Pappas (1992) have suggested that there are three species-complexes that vary according to their dependence on coastal wetland habitat. The “Great Lakes” taxocene, which is primarily associated with open water and only come into tributaries to spawn, do not depend on wetlands; the “transitional” taxocene, which utilizes open water and near-shore, will depend on wetland for spawning and/or nursery habitat; and the true “wetlands” taxocene, which includes species that are either permanent residents of wetlands or those that only migrate into wetlands for nursery, spawning or shelter, are heavily dependent on wetlands. The importance of this proposed classification scheme is that it provides an ecological rather than strictly taxonomic basis for organizing and studying the fish community. Once it has been properly validated, managers should be able to make inferences about availability of wetland fish habitat based on the relative abundance of fish that belong to the transitional and wetland taxocenes.

In this paper, we validate the appropriateness of the proposed classification scheme

of Jude and Pappas (1992). We used an independent dataset that includes information from all five Great Lakes and assign them to one of three taxocenes, and then related their distribution to 11 different habitat types. We determined whether or not fish of different species or taxocenes use wetlands in proportion to their availability or to the availability of other shoreline features. Finally, we examined the influence of wetland type (i.e., open shore-line versus protected embayment) on the distribution of wetland-dependent fish. By revealing the association between certain species or species-complexes and particular shoreline features within the Great Lakes, we attempt to provide a scientifically defensible method for managers and policy-makers to target the appropriate shoreline reaches for restoration, preservation or conservation.

METHODS

Data from various published and unpublished sources have been assembled for this study. Chow-Fraser and Albert (1999) used information from the Atlas of Spawning and Nursery Areas of Great Lakes Fishes (Goodyear et al. 1982) to determine the “biodiversity value” of different stretches of Great Lakes shoreline. This atlas remains the most comprehensive bi-national survey undertaken to date, containing information on all of the commercially and recreationally important species in the Great Lakes fishery. The fourteen-volume atlas documents the spawning and nursery location of 139 fish taxa along the entire shoreline of all five Great Lakes. A record in our fish database consisted of geographic coordinates of the site, fish taxonomic information, and other

classification information such as taxocene or thermal preferences. Fish occurrences were collected and imported into ArcView Geographical Information System. Close to 9,500 of the geo-referenced records, covering all five Great Lakes and connecting channels from the atlas were imported into the WIRE Net database (Wetland Inventory for Research and Education Network, McMaster University, <http://wirennet.mcmaster.ca>). In this study, fish occurrence means spawning and/or nursery use by a fish species at a particular site because we consider both spawning and nursery to be equally important.

Because of differences in gear type used, and the time of year when surveys were conducted, some species may have been missed, but we feel that this bias will affect a relatively small number of species because of the large number of records in the database. A more serious drawback is that common species may be over-represented because of multiple entries, since more than one data source (scientific surveys and published studies) may be included for any site. We filtered out this type of sample bias by only including one record for a given species per site (e.g., a marsh), regardless of how many times the given species had been reported at the given site. Goodyear et al. (1982) also admitted to a bias in their database in favour of major recreational or commercial fishes, or fish that were major components of the forage base.

The 96 of 139 fish taxa reported in the Goodyear et al. (1982) atlas were first grouped by *taxocenes* (Table 4-1). Jude and Pappas (1992) named their taxocenes, “Great Lakes”, “transitional” and “wetlands”. In this study, we have renamed the three taxocenes “open water”, “intermediate”, and “coastal”, respectively, because we felt that

the “Great Lake” taxocene could cause confusion. For comparison, we have also used Coker et al.’s (2001) classification system based on temperature preferenda to group fish into five categories: “cold” “cold-cool”, “cool”, “cool-warm”, and “warm” water species. The thermal criteria for “cold”, “cool”, and “warm” species are < 19.0 °C, 19-25 °C, and > 25 °C, respectively, while cold-cool species are those straddling the boundaries between “cold” and “cool”, and “cool-warm”, those straddling the boundaries between “cool” and “warm”. Coker et al. (2001) indicated that thermal preference was assigned in accordance with the published or reported preferred summer water temperature for a particular species. When data were available, thermal preferences were indicated by preference of adults for certain temperatures in laboratory experiments. If these were not available, then field temperatures at the time of capture reported by investigators were used. When more than one value or a range of values were available, Coker et al. (2001) used the midpoint.

Substrate type of shoreline

For substrate composite, we used medium resolution vector shoreline data provided by the National Oceanic and Atmospheric Administration (1999) for the Great Lakes-St. Lawrence River. The original 19 categories were regrouped into 11 to make the analysis more ecologically meaningful and manageable (see Table 4-2). Categories included bedrock, bluff, coarse beach, sandy beach/dune, sandy-silty bank, clay bank, low riverine-coastal plain, composite, wetland, artificial, and unclassified. We also

classified wetlands into two sub-classes according to geomorphology, “open-shoreline” and “protected embayments”, in a manner similar to that used by Keough et al. (1996).

Statistical analysis

The classical statistical techniques (e.g., ANOVA, chi-square, correlation analysis) assume independence of observations, but many ecological data are spatially autocorrelated and violation of assumption of independence may increase Type I error. One can remove spatial dependency among observations using pre-whitening transformation, trend-surface analysis, spatial variate differencing, or by correcting the number of degrees of freedom (Dutilleul, 1993).

Ease of calculation and interpretation has made categorical analysis (e.g., chi-square-based methods) a popular choice in habitat utilization studies (Brewer and McCann 1982; Alldredge and Ratti 1986, 1992; Thomas and Taylor 1990). For categorical analysis in the present study, the chi-square test is appropriate because of the large number of observations in the database (Jongman et al. 1995). To deal with the problem of non-independence, we followed the advice of Dale and Fortin (2002) who pointed out that a simple approach to the problem would be to acknowledge the existence of positive spatial autocorrelation in the data and then apply a conservative method, the Bonferroni adjustment (Boots 2002; Fortin et al. 2002), to reduce the Type I error rate (α) to a more conservative but arbitrarily chosen value (e.g. $\alpha' = \alpha/5$). In addition, we also used a correction that prevents inflated Type I error related to the number of multiple tests

conducted. Thus, the adjusted α was recalculated as: $\alpha_{\text{Bonferroni}} = \alpha/5 \cdot k$, where α is 0.05, k is the number of tests, and 5 is an arbitrarily chosen value. Therefore, although use of chi-square may not have been optimal in this study, the comparison of differences between expected and observed use of shoreline features was very useful in revealing the nature of the various associations.

We wanted to carry out detailed correlation analyses to determine the relationships between different species of fish and their use of wetlands. To ensure independence of the two variables, we applied Dutilleul's (1993) correction prior to carrying out the analyses (Fortin and Payette 2002; Legendre et al. 2002). To achieve this, we divided the shoreline into small segments and counted the number of fish that occurred in each. Then, we measured the length of shoreline associated with wetlands within each segment. In this way, we were able to obtain an independent dataset of fish and wetland information for each segment. This procedure is referred to as "re-sampling". Size of sections used for re-sampling must be appropriate and ecologically meaningful; too small a size would require too much computation and make this approach impractical, whereas too large a size would mask any substantial relationships. We used two methods to divide the shoreline segments for re-sampling. First, we used Chow-Fraser and Albert's (1999) eco-reach delineations to divide the shoreline segments into 44 segments (Fig. 4-1; the development of eco-reach concept is provided below). Secondly, we superimposed 40 km x 50 km grids over the entire Great Lakes shoreline (total of 198 grid units). Dimensions of the grid unit were chosen to encompass the maximum length

of shoreline feature in the database. For each eco-reach and grid, we calculated fish occurrences and the length of shoreline features. We then used a FORTRAN program, ‘modtest’ (unpublished; P. Legendre, Program Mod_t_test, Département de sciences biologiques, Université de Montreal, <http://www.fas.umontreal.ca/BIOL/legendre/>) to perform the Dutilleul modified t-test and to calculate a Pearson correlation coefficient corrected for spatial autocorrelation.

Description of “eco-reach”

In this study, we adopted the use of “eco-reaches” to represent stretches of the Great Lakes shoreline that support important concentrations of coastal wetlands, and which are characterized by distinctive conditions for coastal wetland development based on differences in climate, bedrock, geology, glacial geomorphology, shoreline configuration, and sills, as well as land use and disturbance factors (Minc 1997). Delineations of many of these eco-reaches do not match existing natural division maps of the Great Lakes area (Albert 1995) based on upland characteristics because coastline conditions reflect a combination of upland and near-shore characteristics. That is, the location of a reach relative to prevailing winds and persistent littoral currents, and to areas of erosion (sources of sediment moved along the coast) are of equal importance to the shoreline configuration as are topography and substrates of immediately adjacent uplands. Detailed descriptions of landform and shoreline characteristics, dominant site types, soil substrates and names of wetlands that are characteristic of each eco-reach are documented

in Chow-Fraser and Albert (1999), which includes a detailed list of sources that were consulted to determine the boundaries of eco-reaches.

RESULTS

Categorical analysis between fish and shoreline classes

The three most common shoreline classes where Great Lakes fish occurred were bedrock (includes both resistant and non-resistant; 21.9%), wetlands (includes open-shoreline, semi-protected and bay-mouth barrier beaches; 21.8%), and sandy beaches/dunes (18.4%) (Table 4-2). Infrequently used classes included a composite class of unknown composition (0.7%), clay bank (0.8%), sand-silty bank (2.4%), coastal plain (1.7%), coarse beach (3.7%), and artificial substrate (3.8%). Classes that had intermediate use included bluffs (11.2%) and an unclassified category (13.7%). The fish community used wetlands, sandy beaches/dunes, and bluffs more frequently than expected, and bedrock less frequently than expected (chi-square goodness-of-fit test, $P < 0.0001$, $\alpha_{\text{Bonferroni}} = 0.05/11 \cdot 5 = 0.0009$).

Analysis by taxocene

We found that 26.1% of the open-water taxocene were associated with bedrock. Use of this type of shoreline feature by the open-water taxocene was significantly greater than that for the other two taxocenes (17.3 and 17.7%, respectively for coastal and intermediate taxocenes; chi-square test, $P < 0.0001$, $\alpha_{\text{Boferroni}} = 0.05/33 \cdot 5 = 0.0003$).

Sandy beaches/dunes (24.7%), wetlands (17.1%), and bluffs (10.8%) were also widely used by the open-water taxocene (Table 4-3). These four shoreline classes together accounted for about 80% of the observed occurrences of the open-water taxa along the Great Lakes shoreline. Given that 16% of all shoreline classes were wetlands (Table 4-2) and 17% of the open-water taxocene were associated with wetlands, these fish were probably using wetlands in proportion to their availability. By comparison, 25% of all occurrences corresponding to the intermediate taxocene were associated with wetlands, indicating that fish in the intermediate taxocene significantly preferred coastal wetland habitat. Other shoreline habitats that were relatively well used by this taxocene included bedrock (17.7%), sandy beaches/dunes (14.3%), and bluffs (11.8%) (Table 4-3). These four shoreline classes together accounted for almost 70% of the observed distribution of the intermediate taxocene in the near-shore zone of the Great Lakes. Almost a third of the fish in the coastal taxocene were associated with coastal wetlands (31.1%; Table 4-3), indicating a very high preference for wetland habitat. Other shoreline habitats that were spatially associated with this group included bedrock (17.3%) and bluffs (11.3%).

Analysis by thermal preference

A breakdown of fish occurrence among the 11 shoreline classes for the five thermal groups (i.e., cold, cold-cool, cool-cold, cool-warm and warm) (Table 4-4, $P < 0.0001$; $\alpha_{\text{Bonferroni}} = 0.05/55 \cdot 5 = 0.00018$) showed that all five groups made extensive use of the

four shoreline classes: bedrock, wetlands, sandy beach/dunes, and bluffs. However, when all shoreline classes were considered, bedrock was used most frequently by the cold-water (25.4%) and cool/warm taxa (22.9%), indicating that these fish were using this habitat type in excess of its availability (21.9%). Wetlands, which co-occurred with 21.8% of the fish, were over-utilized by the warm- (31.0%), cool- (26.8%), and cold/cool-water (23.4%) taxa. By comparison, sandy beach/dunes, which accounted for 18.4% of the used habitat, were used more than expected by the cold-water group (24.9%), but were underutilized by the cold/cool- (16.2%), cool- (12.5%), cool/warm- (4.2%), and warm-water (7.0%) taxa.

The Dutilleul modified t-test of correlation between wetland and fish

Up to this point, we analyzed data using a Chi-square test on categorical data. We wanted to perform a more rigorous test to directly examine the correlation between wetland and fish. We pooled the data and found a highly significant correlation between wetlands and the presence of all species, regardless of taxocene or temperature preferences, and this was true when we re-sampled by eco-reach (Pearson $r = 0.60279$, $P < 0.00001$) or by shoreline grids (Pearson $r = 0.53534$, $P < 0.00001$).

Correlation by taxocene

When we sorted fish data by taxocene and performed the correlation analysis separately for each species-complex, using the eco-reach to resample, both the coastal and intermediate taxocenes exhibited a highly significant positive correlation with wetlands (Pearson $r = 0.49553$, $P = 0.00154$, and Pearson $r = 0.62993$, $P < 0.00001$, respectively). By comparison, when we used shoreline grids to resample, we still obtained a significant positive correlation for these two taxocenes, even though the correlation coefficients were lower (Pearson $r = 0.38185$, $P = 0.01657$ and Pearson $r = 0.53568$, $P < 0.00001$ for coastal and intermediate taxocenes, respectively). In contrast, the open-water species complex was not significantly correlated with wetlands when we used eco-reaches to resample (Pearson $r = 0.15955$, $P = 0.20691$) and was only weakly correlated with wetlands when we used shoreline grids to resample (Pearson $r = 0.21921$, $P = 0.01686$).

Correlation by thermal group

We then sorted the data by thermal group, and performed correlation analysis for each of the five groupings. Regardless of units we used to resample, the warm (Eco-reaches: Pearson $r = 0.62715$, $P < 0.00001$. Grids: Pearson $r = 0.48708$, $P = 0.00034$), cool (Eco-reaches: Pearson $r = 0.62260$, $P < 0.00001$. Grids: Pearson $r = 0.51405$, $P < 0.00001$), and cold-cool (Eco-reaches: Pearson $r = 0.45147$, $P=0.00072$; Grids: Pearson $r = 0.46065$, $P = 0.00002$) groups were significantly and positively correlated with wetlands. By comparison, there was either no significant correlation or only a weak correlation between wetlands and cold (Eco-reaches: Pearson $r = 0.14441$, P

= 0.25412; Grids: Pearson r = 0.20960, P = 0.02234) and cool-warm (Eco-reaches: Pearson r = 0.20244, P = 0.43364. Grids: Pearson r = 0.20903, P = 0.26760) taxa.

Correlation by species

We also calculated correlation coefficients for the top 25 species (by occurrence), which accounted for 80% of the data (Table 4-5). With eco-reach re-sampling, we found the distribution of 11 taxa to be significantly correlated with wetlands. Seven of these were consistent with the classification of Jude and Pappas (1992) as either coastal or intermediate (common carp, emerald shiner, largemouth bass, northern pike, smallmouth bass, walleye, and yellow perch, Table 4-1). Unexpectedly, there were four open-water taxa (lake herring, lake whitefish, round whitefish, and lake sturgeon, Table 4-1) that were also significantly correlated with wetlands. Consistent with expectations, there were eight open-water species that were not significantly correlated with wetlands (brook trout, Chinook salmon, Coho salmon, lake trout, pink salmon, rainbow trout, sea lamprey, and burbot, Table 4-5). Nevertheless, six species that were classified as being either coastal or intermediate were not significantly correlated with wetlands (alewife, gizzard shad, white bass, rainbow smelt, spottail shiner, and white sucker).

Re-sampling the shoreline with the grid increased the number of significant positive associations from seven to nine species within the coastal and intermediate designations (Table 4-5), and reduced the positive associations in the open-water taxocene by one.

Therefore, the correlation revealed by the grid re-sampling appeared to be more consistent with the classification of Jude and Pappas (1992). Regardless of re-sampling strategies, however, we found some wetland-dependent taxa identified by Jude and Pappas to be distributed independently of wetland. We speculated that these anomalies may be due to confounding effects of lumping wetland types together in the analysis, since some fish are known to prefer quiescent environments within protected embayments, while others can tolerate or even prefer exposed habitats of open-shoreline wetlands. Therefore, we recalculated correlation coefficients for the thirteen coastal and intermediate taxa after accounting for differences in site type (Table 4-6). This procedure increased the number of significant associations from seven to nine species when we used the eco-reaches to re-sample, and from nine to eleven when we used the shoreline grid to re-sample. Accounting for wetland type revealed that gizzard shad, rainbow smelt, and white bass were associated with protected embayments, even though we were unable to determine such a significant association when wetland types were lumped.

DISCUSSION

Coastal wetlands are important to fish because of the presence of emergent and submergent plants that provide shelter and a food-source for benthic invertebrates (Chow-Fraser et al. 1998; Lougheed and Chow-Fraser 1998) and epiphytic algae on which larval and juvenile fish feed during their first few months of life (McNair and Chow-Fraser 2003). In turn, these fish become prey for both resident and migratory

piscivores. Another reason for the preferred utilization of coastal wetlands by fishes is that coastal wetlands are warm and sheltered from the often-harsh wave conditions of the open water. It is therefore desirable to have a means to accurately predict which of the Great Lakes fishes depend on wetlands for nursery and spawning habitat so that managers can estimate the impact on the Great Lakes fishery when wetland habitats are lost or restored.

For the most part, we have validated that Jude and Pappas' (1992) classification can be applied to Great Lakes fishes to predict their dependence on wetland versus non-wetland habitat. Of the 25 most abundant species examined, we accurately predicted their preference for or against wetland habitat for 19 and 20 taxa with eco-reach and grid re-sampling, respectively, once we accounted for wetland site type (either as open shoreline or protected embayment). Furthermore, our results indicate that the fish community, as a whole, or when classified according to taxocene or temperature preferences, used wetlands in excess of their availability. The highly significant positive association between warm- and cool-water fish and wetlands was not surprising since most of these also belonged to the coastal and intermediate taxocenes. We also found a significant positive correlation between cold-cool species and wetlands, and this was somewhat unexpected for the burbot and brown trout, but not for the spottail shiner, which is a common species in coastal wetlands of the lower Great Lakes (Seilheimer and Chow-Fraser, pers. obs.). The lack of a significant correlation between cool/warm taxa and wetlands may have been due to their relatively small sample size (six of the

seven species having observations < 25). By comparison, we expected the distribution of cold-water taxa to vary independently with wetlands since these species correspond primarily with the open-water taxocene that associated more strongly with bedrock, coarse beaches, and sandy beach-dunes than with wetlands.

We also found that wetland type (i.e., open-shoreline wetland versus protected embayment) influenced the distribution of certain wetland-dependent taxa. The results from the correlation analysis with eco-reach re-sampling suggested that common carp, emerald shiner, gizzard shad, northern pike, rainbow smelt, walleye, white bass, and yellow perch favour protected embayments for spawning and/or nurseries. In contrast, alewife, largemouth bass, smallmouth bass, and spottail shiner appeared to favour open shoreline wetlands. The results from correlation analysis with grid re-sampling were consistent with those re-sampled by eco-reach except for largemouth bass and spottail shiner.

There were several departures from expected based on Jude and Pappas' (1992) classification, and knowledge of the temperature preferenda of the fish in question. The lack of a significant correlation between the distribution of white sucker and alewife with either wetland type could have been due to sampling bias related to the timing of surveys. These intermediate taxa are migratory species that swim into wetlands to spawn during early spring (Scott and Crossman 1998), and may have been under-sampled if surveys had been conducted only in mid-summer and fall. The positive association between lake sturgeon and wetlands is not surprising because of their presence in deeper

areas of Lake St. Clair and the connecting channels (M. Thomas, Michigan Department of Natural Resources, Mt. Clemens, 33135 South River Road Mt. Clemens, MI 48045, Pers. Comm.); hence, even though it is classified as a cold-water species, and an open-water taxocene, it is probably dependent on deeper areas of wetlands. We also found a significant positive association between wetlands and two other open-water, cold-water species, lake herring and lake whitefish. Mitsch and Gosselink (2000) have indicated that lake herring utilized coastal marshes or protected embayments for at least parts of their life cycle, and several investigators have observed a number of larval lake whitefish and lake herring in shallow areas along the St. Marys River (< 2 m) (Clady 1976; Liston et al. 1986; Jude et al. 1998) and in Grand Traverse Bay (< 3m) (Freeberg et al. 1990). Therefore, wetlands are probably very important habitat for at least these open-water taxa, and the extent to which others may depend on wetlands should be verified with future field surveys.

In addition to coastal wetlands, our study also indicated that near-shore waters associated with bedrock, sandy beach-dunes, and bluffs are widely used as reproductive habitats by the Great Lakes fish community. These three shoreline features are well represented along the Great Lakes shoreline, and account for 62% of the total shoreline length. Frequent occurrence of fish in these near-shore areas probably reflects use by both permanent residents as well as migratory fish (e.g., anadromous fishes) that use these for temporary feeding or nursery grounds (Edsall and Charlton 1997).

To our knowledge, this study is one of the most extensive examinations of fish

distribution patterns at the scale of the Great Lakes basin. We have confirmed that the Great Lakes fish community, especially species identified as wetland-dependent by Jude and Pappas (“transitional” and “wetlands” taxocenes; 1992) preferentially used coastal wetlands for spawning and nursery habitat. We also confirmed that the distribution of wetland-associated taxa is influenced by wetland type (i.e., protected embayment versus open-shoreline wetland) and that the Great Lakes fish community utilized certain shoreline features (especially wetlands) disproportionately to their availability. Future studies should determine the specific environmental attributes (water or substrate quality, food-web components, etc.) in wetlands that make these habitats important for the particular species-complexes.

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Figure 4-1. Map of 77 eco-reach for the Great Lakes (after Chow-Fraser and Albert 1999).

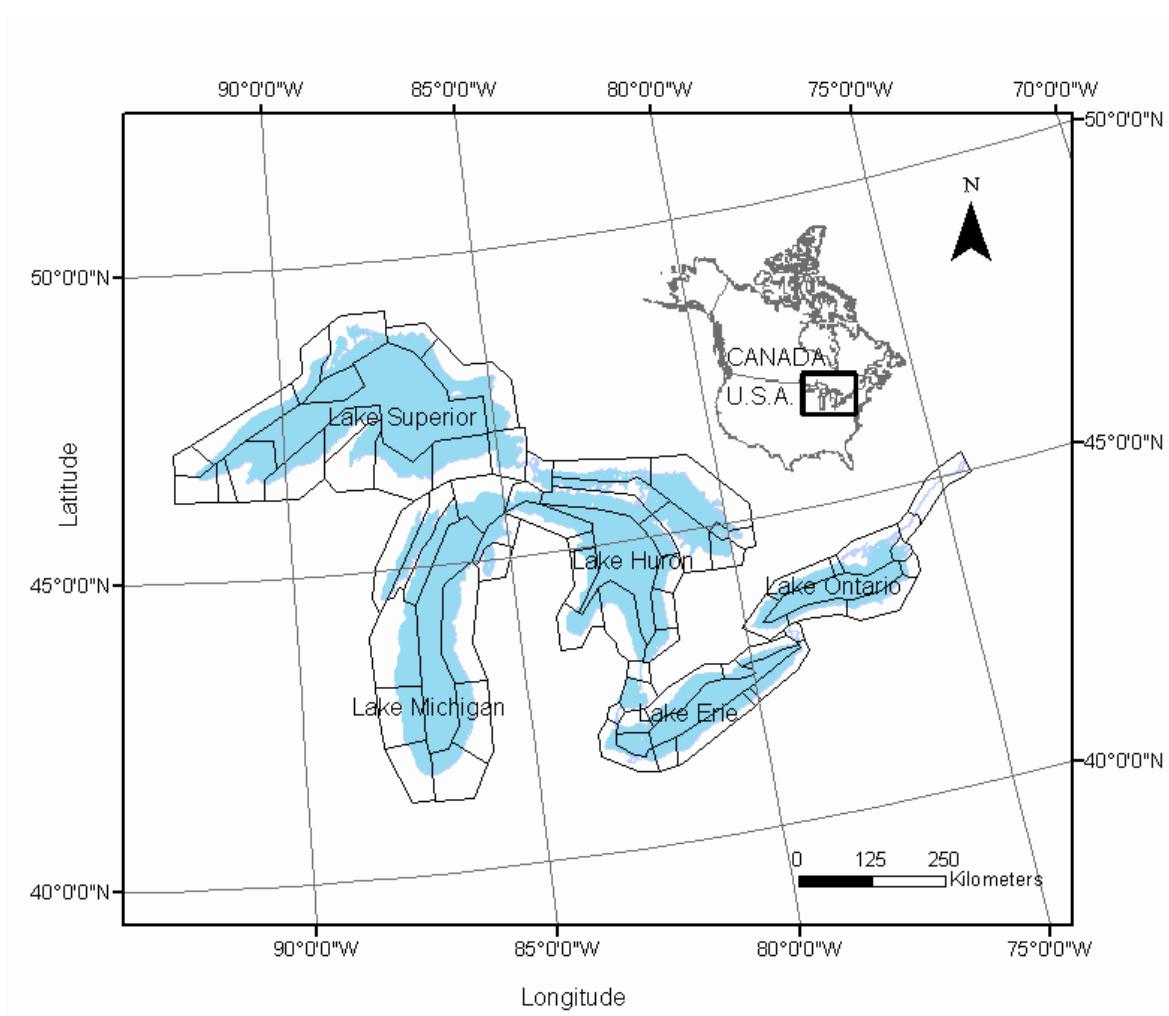


Table 41. Species in this study sorted by taxocene (after Jude and Pappas 1992) and temperature preferenda (after Coker et al. 2001). Numbers in brackets indicate the total number of occurrences for each species in the database.

Coastal taxocene		Intermediate taxocene		Open-water taxocene	
Species	Thermal group	Species	Thermal group	Species	Thermal group
Banded killifish (5)		Alewife (228)			
(<i>Fundulus diaphanus</i>)	Cool	(<i>Alosa pseudoharengus</i>)	Cold	(<i>Salmo salar</i>)	Cold
Blacknose dace (3)	Cool	Rainbow smelt (519)	Cold	Blackfin cisco (15)	Cold
(<i>Rhinichthys atratulus</i>)		(<i>Osmerus mordax</i>)		(<i>Coregonus nigripinnis</i>)	
Brook stickleback (7)	Cool	Trout-perch (93)	Cold	Bloater (27)	Cold
(<i>Culaea inconstans</i>)		(<i>Percopsis omiscomaycus</i>)		(<i>Coregonus hoyi</i>)	
Common shiner (31)	Cool	Spottail shiner (125)	Cold/cool	Brook trout (73)	Cold
(<i>Luxilus cornutus</i>)		(<i>Notropis hudsonius</i>)		(<i>Salvelinus fontinalis</i>)	
Fantail darter (4)	Cool	American eel (1)	Cool	Chinook salmon	Cold
(<i>Etheostoma flabellare</i>)		(<i>Anguilla rostrata</i>)		(<i>Oncorhynchus tshawytscha</i>)	

Gizzard shad (105)	Cool	Black crappie (62)	Cool	Coho salmon (300)	Cold
(<i>Dorosoma cepedianum</i>)		(<i>Pomoxis nigromaculatus</i>)		(<i>Onchorynchus kisutch</i>)	
Golden shiner (16)	Cool	Creek chub (17)	Cool	Kiwi (19)	Cold
(<i>Notemigonus crysoleucas</i>)		(<i>Semotilus atromaculatus</i>)		(<i>Coregonus kiyi</i>)	
Iowa darter (6)	Cool	Emerald shiner (113)	Cool	Lake chub (31)	Cold
(<i>Etheostoma exile</i>)		(<i>Notropis atherinoides</i>)		(<i>Couesius plumbeus</i>)	
Pugnose shiner (2)	Cool	Johnny darter (47)	Cool	Lake herring (320)	Cold
(<i>Notropis anogenus</i>)		(<i>Etheostoma nigrrum</i>)		(<i>Coregonus artedii</i>)	
Quillback (23)	Cool	Longnose dace (40)	Cool	Lake trout (1150)	Cold
(<i>Carpio cyprinus</i>)		(<i>Rhinichthys cataractae</i>)		(<i>Salvelinus namaycush</i>)	
Rock bass (83)	Cool	Northern pike (321)	Cool	Lake whitefish (675)	Cold
(<i>Ambloplites rupestris</i>)		(<i>Esox lucius</i>)		(<i>Coregonus clupeaformis</i>)	
White crappie (32)	Cool	Walleye (427)	Cool	Longnose sucker (70)	Cold
(<i>Pomoxis annularis</i>)		(<i>Stizostedion vitreum</i>)		(<i>Catostomus catostomus</i>)	
Blacknose shiner (2)	Cool/ warm	White sucker (229)	Cool	Mottled sculpin (15)	Cold
(<i>Notropis heterolepis</i>)		(<i>Catostomus commersoni</i>)		(<i>Cottus bairdi</i>)	

Central (20)	mudminnow <i>(Umbra limi)</i>	Cool/ warm	Yellow perch (410) <i>(Perca flavescens)</i>	Cool	Ninespine stickleback <i>(Pungitius pungitius)</i>	Cold
Logperch (74)	<i>(Percina caprodes)</i>	Cool/ warm	Black redhorse (3) <i>(Moxostoma duquesnei)</i>	Cool/ warm	Pink salmon (107) <i>(Oncorhynchus gorbuscha)</i>	Cold
Pugnose minnow (2)	<i>(Notropis emiliae)</i>	Cool/ warm	Brook silverside (25) <i>(Labidesthes sicculus)</i>	Cool/ warm	Pygmy whitefish (5) <i>(Prosopium coulteri)</i>	Cold
Big mouth buffalo (14)	<i>(Ictiobus cyprinellus)</i>	Warm	Mooneye (7) <i>(Hiodon tergisus)</i>	Cool/ warm	Rainbow trout (519) <i>(Oncorhynchus mykiss)</i>	Cold
Black bullhead (12)	<i>(Ameiurus melas)</i>	Warm	Common carp (247) <i>(Cyprinus carpio)</i>	Warm	Round whitefish (130) <i>(Prosopium cylindraceum)</i>	Cold
Bluegill (44)	<i>(Lepomis macrochirus)</i>	Warm	Channel catfish (97) <i>(Ictalurus punctatus)</i>	Warm	Sea lamprey (448) <i>(Petromyzon marinus)</i>	Cold
Bluntnose minnow (17)	<i>(Pimephales notatus)</i>	Warm	Flathead catfish (2) <i>(Pylodictus olivaris)</i>	Warm	Slimy sculpin (40) <i>(Cottus cognatus)</i>	Cold

Bowfin (41)	Warm	Golden redhorse (<i>Moxostoma erythrurum</i>)	Warm	Spoonhead sculpin (<i>Cottus ricei</i>)	Cold
Brindled madtom (3)	Warm	Mimic shiner (6) (<i>Notropis volucellus</i>)	Warm	Threespine stickleback (12) (<i>Gasterosteus aculeatus</i>)	Cold
Brown bullhead (72)	Warm	Muskellunge (79) (<i>Esox masquinongy</i>)	Warm	Brown trout (138) (<i>Salmo trutta</i>)	Cold/cool
Fathead minnow (7)	Warm	Northern hog sucker (2) (<i>Hypentelium nigricans</i>)	Warm	Burbot (125) (<i>Lota lota</i>)	Cold/cool
Freshwater drum (89)	Warm	Orangespotted sunfish (1) (<i>Lepomis humilis</i>)	Warm	Bridle shiner (3) (<i>Notropis bifrenatus</i>)	Cool
Goldfish (53)	Warm	Shorthead redhorse (30) (<i>Moxostoma macrolepidotum</i>)	Warm	Sauger (18) (<i>Stizostedion canadense</i>)	Cool
Grass pickerel (7)	Warm	Smallmouth bass (308) (<i>Micropterus dolomieu</i>)	Warm	Silver redhorse (17) (<i>Moxostoma anisurum</i>)	Cool

Green sunfish (8)	Warm	Stonecat (9)	Warm	Lake sturgeon (125)	Cool/ cold
(<i>Lepomis cyanellus</i>)		(<i>Noturus flavus</i>)		(<i>Acipenser fulvescens</i>)	
Lake chubsucker (6)	Warm			Channel darter (33)	Cool/ cold
(<i>Erimyzon sucetta</i>)				(<i>Percina copelandi</i>)	
Largemouth bass	Warm				
(149)					
(<i>Micropterus salmoides</i>)					
Longnose gar (37)	Warm				
(<i>Lepisosteus osseus</i>)					
Pumpkinseed (52)	Warm				
(<i>Lepomis gibbosus</i>)					
Sand shiner (11)	Warm				
(<i>Notropis stamineus</i>)					
Spotfin shiner (12)	Warm				
(<i>Cyprinella spiloptera</i>)					
Spotted gar (2)	Warm				
(<i>Lepisosteus oculatus</i>)					
Tadpole madtom (4)	Warm				
(<i>Noturus gyrinus</i>)					
White bass (123)	Warm				
(<i>Morone chrysops</i>)					

White perch (32) Warm

(Morone americana)

Yellow bullhead (7) Warm

(Ictalurus natalis)

Table 4-2. Occurrence of fish in different shoreline classes. Expected percent = [length of shoreline class/length in total]• 100. Observed counts = numbers of occurrences of fish associated with a shoreline class. Expected counts = [numbers of occurrences of fish • percent]. Numbers in bold indicate there is a significant departure between expected and observed after Bonferroni adjustment has been applied at $\alpha = 0.05/11\cdot5$.

Shoreline class	Expected	Expected	Observed	Observed
	percent (%)	counts	percent (%)	counts
Bedrock	38.9	3667.2	21.9	2062
Bluff	8.4	789.4	11.2	1057
Coarse beach	5.6	524.5	3.7	345
Sandy beaches-dune	15.0	1415.4	18.4	1734
Sandy-silty bank	1.4	128.6	2.4	228
Clay bank	0.7	67.1	0.8	75
Low riverine-coastal plain	5.3	497.5	1.7	162
Composite	0.4	36.1	0.7	63
Wetland	16.1	1520.1	21.8	2056
Artificial	2.6	242.4	3.8	359
Unclassified	5.7	541.7	13.7	1289
Total	100.0	9430.0	100.0	9430

Table 4-3. The number of fish associated with shoreline classes within three taxocenes.

Numbers below in brackets are percentages. Numbers in bold indicate that they are the highest for the taxocene.

Shoreline class	Expected %	Taxocene			
		Coastal		Intermediate water	All
					taxocenes
Bedrock	38.9	211 (17.3)	613 (17.7)	1238 (26.1)	2062 (21.9)
Bluff	8.4	138 (11.3)	407 (11.8)	512 (10.8)	1057 (11.2)
Coarse beaches	5.6	18 (1.5)	71 (2.1)	256 (5.4)	345 (3.7)
Sandy beaches-dunes	15.0	66 (5.4)	495 (14.3)	1173 (24.7)	1734 (18.4)
Sandy-silty banks	1.4	38 (3.1)	119 (3.4)	71 (1.5)	228 (2.4)
Clay banks	0.7	15 (1.2)	41 (1.2)	19 (0.4)	75 (0.8)
Low riverine-coastal plain	5.3	37 (3.0)	97 (2.8)	28 (0.6)	162 (1.7)
Composite	0.4	0 (0)	10 (0.3)	53 (1.1)	63 (0.7)
Wetlands	16.1	378 (31.1)	865 (25.0)	813 (17.1)	2056 (21.8)

Artificial	2.6	61	143	155	359
		(5.0)	(4.1)	(3.3)	(3.8)
Unclassified	5.7	255	600	434	1289
		(21.0)	(17.3)	(9.1)	(13.7)
Total	100.0		1217	3461	4752
					9430

Table 4-4. The number of fish associated with shoreline classes within thermal preference groups. Numbers below in brackets are percents. Numbers in bold indicate that they are the highest for the group.

Shoreline class	Thermal preference					
	Cold	Cold/cool	Cool	Cool/warm	Warm	Total
Bedrock	1303	71	351	38	299	2062
	(25.4)	(13.8)	(17.4)	(22.9)	(18.7)	(21.9)
Bluff	562	73	227	30	165	1057
	(10.9)	(14.2)	(11.2)	(18.1)	(10.3)	(11.2)
Coarse Beaches	262	12	38	3	30	345
	(5.1)	(2.3)	(1.9)	(1.8)	(1.9)	(3.7)
Sandy Beach-Dunes	1279	83	253	7	112	1734
	(24.9)	(16.2)	(12.5)	(4.2)	(7.0)	(18.4)
Sandy-Silty Banks	81	18	72	9	48	228
	(1.6)	(3.5)	(3.6)	(5.4)	(3.0)	(2.4)
Clay Banks	21	6	25	2	21	75
	(0.4)	(1.2)	(1.2)	(1.2)	(1.3)	(0.8)
Low Riverine-Coastal Plain	36	7	68	2	49	162
	(0.7)	(1.4)	(3.4)	(1.2)	(3.1)	(1.7)
Composite	52	5	6	0	0	63
	(1.0)	(1.0)	(0.3)	(0)	(0)	(0.7)
Wetland	868	120	542	31	495	2056
	(16.9)	(23.4)	(26.8)	(18.7)	(31.0)	(21.8)
Artificial	169	23	87	11	69	359
	(3.3)	(4.5)	(4.3)	(6.6)	(4.3)	(3.8)
Unclassified	500	95	353	33	308	1289
	(9.7)	(18.5)	(17.5)	(19.9)	(19.3)	(13.7)
Total	5133	513	2022	166	1596	9430

Table 4-5. Summary of correlation analyses for 25 fishes of the most abundant study species, which accounted for 80% of the fish occurrences. C = coastal; I = intermediate; O = open-water. P-values < 0.05 are in bold.

Species	Pearson <i>r</i>	P-value	Pearson <i>r</i>	P-value
	by eco-reach re-sampling		by grid re-sampling	
Largemouth bass (C)	0.62653	0.00263	0.48418	0.00404
Gizzard shad (C)	0.30925	0.17835	0.25355	0.26007
Common carp (I)	0.40187	0.00903	0.33029	0.02294
Emerald shiner (I)	0.63841	0.00764	0.39918	0.05702
Northern pike (I)	0.57417	0.00001	0.43713	0.00014
Smallmouth bass (I)	0.43833	0.01225	0.32917	0.01589
Walleye (I)	0.52120	0.00003	0.40030	0.00319
Yellow perch (I)	0.55336	0.00003	0.46045	0.00020
Rainbow smelt (I)	0.13365	0.29795	0.25732	0.01407
Alewife (I)	0.19428	0.22837	0.20325	0.07421
Spottail shiner (I)	0.30467	0.12953	0.43135	0.00591
White bass (I)	0.23201	0.35469	0.22884	0.23583
White sucker (I)	0.05236	0.75798	0.21220	0.09707
Brown trout (O)	0.04420	0.79251	0.18301	0.20457
Burbot (O)	0.13034	0.46117	0.16825	0.28619

Chinook salmon (O)	-0.12042	0.42289	-0.01826	0.88384
Coho Salmon (O)	-0.04538	0.75159	0.05423	0.62398
Lake trout (O)	0.08646	0.51231	-0.06632	0.54090
Pink salmon (O)	-0.09863	0.61812	-0.03304	0.86894
Rainbow trout (O)	-0.01937	0.88205	0.09872	0.31573
Sea lamprey (O)	0.12172	0.37750	0.19758	0.07251
Lake sturgeon (O)	0.61444	0.00025	0.56425	0.00010
Lake herring (O)	0.33458	0.01496	0.23469	0.04857
Lake whitefish (O)	0.33190	0.01272	0.30342	0.00480
Round whitefish (O)	0.36807	0.05337	0.07561	0.64704

Table 4-6. Summary of correlation analyses for 13 most abundant wetland-dependent fish species. PE = protected embayment; OS = open shoreline. C = coastal; I = intermediate; O = open-water. P-values < 0.05 are in bold.

Species	Wetland type	Pearson <i>r</i>	P-value	Pearson <i>r</i>	P-value
		by eco-reach		by grid	
		re-sampling	re-sampling	re-sampling	re-sampling
Largemouth bass (C)	PE	0.21498	0.39617	0.47089	0.01595
	OS	0.58273	0.01984	0.32502	0.16782
Gizzard shad (C)	PE	0.54499	0.00783	0.58024	0.01065
	OS	0.01273	0.97569	-0.07291	0.78411
Common carp (I)	PE	0.45868	0.00239	0.50280	0.00015
	OS	0.23486	0.34589	0.10343	0.59224
Emerald shiner (I)	PE	0.62559	0.00254	0.75615	0.00016
	OS	0.45634	0.18628	-0.03600	0.91231
Northern pike (I)	PE	0.60117	0.00002	0.43288	0.00043
	OS	0.26089	0.15036	0.27104	0.07507
Smallmouth bass (I)	PE	0.28233	0.13964	0.31385	0.03122
	OS	0.35034	0.09861	0.40666	0.01414
Walleye (I)	PE	0.52120	0.00003	0.61774	0.00001
	OS	0.39677	0.05199	0.12777	0.44339
Yellow perch (I)	PE	0.61386	0.00001	0.61546	0.00001
	OS	0.35468	0.06911	0.25465	0.10136

Rainbow smelt (I)	PE	0.23156	0.05551	0.36948	0.00027
	OS	0.10261	0.62064	0.14140	0.35058
Alewife (I)	PE	0.12035	0.42636	0.20946	0.09161
	OS	0.31244	0.13544	0.21207	0.21644
Spottail shiner (I)	PE	0.02532	0.88934	0.43148	0.00896
	OS	0.50803	0.12736	0.38045	0.03239
White bass (I)	PE	0.62533	0.00215	0.59268	0.00930
	OS	-0.27044	0.50746	-0.16921	0.48257
White sucker (I)	PE	0.02590	0.87767	0.17044	0.21383
	OS	0.08644	0.70362	0.29662	0.11370

CHAPTER 5:

Use of IKONOS imagery to map coastal wetlands of Georgian Bay

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ABSTRACT

Wetlands throughout North America have been diminished in quantity and quality because of human activities, and it is therefore important that fishery managers monitor changes in supply of this critical fish habitat. Use of traditional field-based methods to detect and record the change in aquatic vegetation in Great Lakes wetlands is a daunting task because wetlands are extensive and widely distributed along the Great Lakes shoreline. Mapping wetlands for such a large geographic area necessitates the use of remote sensing technology to obtain an accurate inventory of these ecosystems. The objective of this study was to explore the capabilities of using IKONOS satellite imagery to map different types of aquatic vegetation and habitat features in Great Lakes wetlands. We acquired imageries for Fathom Five National Marine Park in Lake Huron and an area of eastern Georgian Bay in 2002 and chose 11 wetlands for habitat mapping with remote sensing software. The comparison of results of the image analysis with reference data indicated that the overall accuracy of mapping was approximately 90%. This suggests that high resolution IKONOS imagery can be used effectively to monitor the change in aquatic vegetation and thus track alterations in fish habitat in Great Lakes coastal marshes.

INTRODUCTION

Coastal wetlands are known to be very important to the fisheries of the Laurentian Great Lakes because they provide spawning and nursery habitat for wetland-dependent species that include a large number of the commercially and recreationally important taxa (e.g., Jude and Pappas 1992; Wei et al. 2004). The U.S. Nature Conservancy estimated that about 80% of the approximately 200 fish species found in the Great Lakes use the near-shore areas for at least part of the year and directly depend on coastal wetlands for some part of their life cycles (Chow-Fraser and Albert 1999). Both government agencies and non-governmental organizations have now acknowledged the important ecological values and functions of these coastal ecosystems (Maynard and Wilcox 1997; Chow-Fraser and Albert 1999), and have devoted considerable effort over the past two decades towards developing strategies to protect and restore these habitats at a basin-wide scale.

An important first step in the management of coastal wetlands is the development of a basin-wide inventory that can be updated at regular intervals. The wide distribution of wetlands in the Great Lakes basin necessitates the use of remote sensing technology, such as aerial photographs or satellite images. With high-resolution color-infrared aerial photographs, detailed habitat features can be distinguished from each other, but the costs associated with this can be sufficiently high that updates can only be carried out at 10-year intervals (e.g., U.S. National Wetland Inventory; Wilen et al. 2002). By comparison, satellite data (e.g., Landsat 5 or 7) can be more cost-effective because of the

large spatial coverage captured in each satellite scene, but the resolution is often too coarse to discriminate habitat features such as type of aquatic plants at small spatial scales (e.g., Mumby and Edwards 2002). A third alternative, IKONOS (derived from the Greek word for "image"), is a high-resolution satellite capable of simultaneously collecting 1-m panchromatic (single band or monochrome imagery) and 4-m multispectral images (4 bands) over a relatively large geographic area. Suitable for mapping wetland habitat at much smaller spatial scales (e.g., < 10m) than has been possible with other satellite imagery such as Landsat satellite, IKONOS has been used successfully in several coastal projects in marine systems (e.g., Mumby and Edwards 2002; Andréfouët et al. 2003; Riegl and Purkis 2005).

To date, no study has detailed the use of IKONOS in freshwater coastal areas, such as the Laurentian Great Lakes. Our objective was to use IKONOS imagery to map detailed habitat features in freshwater wetlands in a small region of Lake Huron and Georgian Bay. We first conducted ground surveys of these wetlands, and then used the location of ground features (emergent vegetation, submergent vegetation, open water, etc) to guide the classification of aquatic vegetation cover in the IKONOS image. Finally we assessed the overall accuracy of this classification and evaluated the potential for using IKONOS imagery to map Great Lakes aquatic habitat at a basin-wide scale.

METHODS

Site description

Eleven wetland sites in the Georgian Bay region were examined (Table 1 and Figures 1 and 2). Ten of the 11 wetlands were found in Fathom Five National Marine Park (FFNMP), which is located at the boundary zone between Georgian Bay to the east and Lake Huron to the west. Two of the wetlands in this study are located on the mainland at the northern tip of the Bruce Peninsula: Hay Bay wetland complex and Ragged Bight wetland. In addition to natural stressors such as water level fluctuations, these wetlands are also affected by nutrient and sediment loading from their watersheds. The remaining FFNMP wetlands are located on two islands, the larger of which is Cove Island and the smaller is Russel Island. Most of these island wetlands are unaffected by human-induced stressors such as nutrient and sediment enrichment (Chow-Fraser, unpub. data). Herman's Bay is a very small (3-ha) pristine embayment, which is hydrologically attached to eastern Georgian Bay through Twelve Mile Bay (Figures 1 and 2). The shoreline is undeveloped and there is no obvious anthropogenic impact. Plant life in this marsh is extremely abundant and the distributional pattern of broad groups of wetland plants is distinct. These characteristics makes Herman's Bay an ideal site to explore the potential capability of IKONOS for detecting wetland plants at the level of species assemblages. Plant covers from Herman's Bay have been identified and classified into four types based on ground truth data collected in August 2004: (1) meadow, (2) emergent zone dominated by *Scirpus*, (3) emergent zone dominated by *Pontederia*, and (4) a mixed floating-emergent zone dominated by *Nuphar* and *Sparganium*.

Principles of mapping wetlands with remote sensing imagery

Satellite sensors can record reflectance from Earth surface features. Many of these features have distinctive spectral reflectance, which is referred to as spectral response pattern or spectral “signature.” Automated image classification uses the spectral information represented by the digital numbers in satellite imagery and attempts to assign all pixels (points) in the image to particular classes based on this spectral information (e.g., open water, submergent vegetation, or emergent vegetation). Figure 3 illustrates a stage in a typical procedure used to map wetland habitat with remote sensing techniques. Initially, geographic coordinates (i.e., latitude and longitude acquired with GPS units) must be collected in the field, which will serve as reference (ground truth) data to classify the major features being mapped (i.e., open water, submergent vegetation, emergent vegetation in this hypothetical wetland). Based on these field data, representative areas can then be selected by analysts on satellite imagery (Figure 3a). Supervised by analysts and trained by the representative areas, image pixels with similar reflectance patterns are grouped into the same habitat class (Figure 3 b-c).

Procedures used to map aquatic habitat in FFNMP wetlands

The classification procedures are similar to those for mapping terrestrial systems which can be found in most remote sensing textbooks. The procedures used to map FFNMP wetlands are summarized as follows:

- (a) *Acquiring IKONOS imagery*

The relevant imageries (Figure 2) were separately acquired by Parks Canada (for wetlands in FFNMP) and the Georgian Bay Association Foundation (GBA Foundation) (for Herman's Bay) from Space Imaging (Thornton, CO 80241) in 2002. In each case, both Parks Canada and GBA Foundation indicated the area of interest by providing Space Imaging with a set of geographic coordinates, as well as the preferred season.

(b) *Collecting ground-truth data*

First, we determined the number of habitat classes to be mapped. For FFNMP wetlands, we determined that five habitat features based on the dominant vegetation type and geological features would be suitable: (1) emergent vegetation, (2) submersed aquatic vegetation (SAV), (3) rock/shrubs, (4) rock, and (5) open water. However, for Herman's Bay we determined that five zones based on the distinct distribution pattern of plant assemblages would be more suitable: (1) sedge meadow, (2) *Scirpus validus* (tall emergent species that grew along the shoreline), (3) *Nuphar variegatum* and *Sparganium fluctuans* (both floating species growing in shallow to moderately deep water), (4) *Pontederia cordata* (short emergent species that grew in shallow water), and (5) open water without the presence of emergent or floating species. Note that we did not map the location of submergent species, because these were found growing below the water surface throughout the wetland, even where there were emergent and floating species. We verified that SAV was only absent in the vicinity of the

opening to Twelve Mile Bay, where water depth approached 1.0 m. The second step was to locate homogenous areas (minimum size of 4 x 4 m) of each habitat class within the wetlands. Thirdly, we obtained geographic coordinates within each homogeneous patch for each of the five classes using a GPS unit. The number of geographical coordinates to be recorded could vary according to the habitat complexity and size of the wetlands. For instance, we collected 17 pairs of coordinates for SAV in Hay Bat 1 (ML1) while only two pairs of coordinates for the same class in Cove Island North Pond (CNP). This is because SAV in CNP was highly homogenous (i.e., CNP was almost 100% covered by SAV) and two points would be sufficient for us to select representative areas for SAV on the imagery.

(c) *Working with field data and satellite imagery in a remote sensing platform*

We imported the ground-truth data, along with the satellite imagery into a remote sensing platform using software called ENVI 4.1 (ITT Visual Information Solutions, formerly Research Systems, Inc., Boulder, CO). Then, representative areas, also called training areas, were identified within homogeneous areas for each habitat class on the imagery. The selection of appropriate training areas is generally based on the analyst's familiarity with the geographical area and the availability of ground truth data (Figure 3). In remote sensing, it is not unusual to have field and satellite data collected at different times for a variety of reasons (e.g., use of existing archive images, limited project budgets, timing of funding

cycles, limited access to the field sites etc.). Since differences in vegetation cover between years may exist, the field data were not used directly in the classification procedure. Instead, field data were used to help the analyst to identify and choose representative areas of each habitat class on the imagery and then the representative areas were divided into a “training set” and a “testing set” to be used in a supervised classification procedure and to check for post-classification accuracy, respectively. For FFNMP wetlands, the training set was collected from Cover Island North (CN) and CNP while the testing set was independently chosen from ML1. The training and testing sets for Herman’s Bay were collected from the west and east portions of the wetland, respectively.

(d) Supervised classification procedure with maximum likelihood algorithm

Our “supervised classification” procedure is commonly used in remote sensing. This procedure is applied in two steps (Lillesand and Kiefer 2000): (1) in the training stage, representative sample sites of known ground features (training areas), are provided to the classification algorithm (e.g., Maximum Likelihood) and form the basis for image classification; and (2) in the classification stage, the computer algorithm (e.g., Maximum Likelihood) categorizes each pixel in the image into the representative class it most closely resembles (Figure 3). To reduce the complexity of classification and computational time, we used the wetland boundary to delineate the “region of interest” to avoid processing areas in the

satellite image that occurred outside the wetland.

(e) *Determining classification accuracy*

A classification error matrix is a common means of expressing classification accuracy. In such a matrix the accuracy values of each column indicate the percentages that are correctly classified. The overall accuracy reported in the classification error matrix is calculated by dividing the number of image pixels classified correctly by the total number of reference image pixels. Producer accuracy (Prod. Acc.) is calculated by dividing the number of correctly classified pixels for a class by the actual number of ground truth pixels for that class. User accuracy (User Acc.) is calculated by dividing the number of correctly classified pixels for a class by the total pixels assigned to that class.

RESULTS AND DISCUSSION

The 10 sites chosen from FFNMP for this study were all located within the same region of the satellite image. The goal of the Fathom Five study was to evaluate the capability of IKONOS imagery to accurately map aquatic habitat at a regional level. Results of the supervised classification for Fathom Five wetlands are shown in Table 2a and Figure 4. The classification error matrix based on the representative areas (testing set) indicated that the overall accuracy was 84.5% (Table 3).

During periods of high water, wetlands located on Cove Island were hydrologically connected to the rest of the lake, but during recent periods of low water levels (since

1999), some of these wetlands have become disconnected and “stranded.” We found that these stranded wetlands had almost 100 % cover of submergent plants, and this is unlike other wetland areas of FFNMP that are exposed to wave action, where submergent plants are scarce.

Unlike wetlands of FFNMP, Herman’s Bay is a highly protected marsh. It has a low-energy environment that allows organic matter to accumulate and thus supports a variety of aquatic plants in the marsh (Figure 2). The supervised classification estimated the following coverages for the five habitat features: 20.0% open water, 27.9% *Nuphar* and *Sparganium*, 16.9% *Pontederia*, 20.7% *Scirpus*, and 14.4% sedge meadow (Table 2b). The classification error matrix based on the representative areas (testing set) indicated that the overall accuracy was 90.82 % (Table 3).

Studies have shown that there is considerable improvement in the capabilities of IKONOS over Landsat and other satellite imagery that are more suitable for coarse habitat mapping (e.g., Andréfouët et al. 2005). Andréfouët et al. (2005) indicated that overall accuracy for Landsat was 15-20% lower than that for IKONOS when used to classify tropical coral reef environments, and that only IKONOS produced sufficiently high accuracy (> 80%) for four of the five classes. Our results indicate that IKONOS imagery can be used for wetland inventories, because of the large spatial coverage (over 100 km²) and the relatively high level of precision when carried out with the supervised classification, both of which are required when gathering synoptic information at regional or basin-wide scales. On an areal basis, the cost of IKONOS images is substantially lower

than that for aerial photographs, but still very expensive when compared with Landsat images (Table 4). If the primary objective of an investigation is to map the total wetland area for a large geographical area, Landsat will be more cost-effective. If habitat features need to be monitored at a small spatial scale (e.g., 100 m²), and the area to be mapped is <500 km², then IKONOS would be a cost-effective option (Mumby and Edwards 2002). Results from Herman's Bay also demonstrate that IKONOS imagery can be used to accurately identify plant form as well as species assemblages where training data are provided at the appropriate level of resolution (i.e., four broad groups with distinctive spectral properties). Our results indicate that use of IKONOS imagery to inventory wetlands has the advantage of wide spatial coverage and the precision of supervised classification, thus meeting the requirement for gathering synoptic information on wetlands at regional scales. The high water transparency and relatively undisturbed nature of the wetlands in eastern and northern Georgian Bay make them excellent candidates for use with IKONOS imagery for wetland classification.

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imagery of Herman's Bay and provided transportation to and from the wetland in 2003 to 2005 to conduct the sampling. We want to thank Scott Parker from FFNMP for providing transport to and from FFNMP islands, and for use of the IKONOS imagery for FFNMP sites. We also owe a debt of thanks to Mel Croft , Sheila McNair, and Maja Cvetkovic for assisting with the field sampling and conducting some of the plant surveys.

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<http://water.usgs.gov/nwsum/WSP2425/mapping.html>

Table 5-1. Summary of sites and a brief description of likely impact.

Site	Code	Area (ha)	Type of impact
1. Boat Passage	BG	16.7	Low human impact (Boat channel)
2. Cove Island Inner Harbour	HR1	5.7	No obvious human impact
3. Cove Island Outer Harbour	HR2	2.9	No obvious human impact
4. Cove Island North	CN	16.4	No obvious human impact No obvious human impact, declining
5. Cove Island North Pond	CNP	1.4	water level No obvious human impact, declining
6. Bass Bay	BB	39.0	water level High human impact (public beach,
7. Hay Bay One	ML1	7.7	high cottage density)
8. Ragged Bight	ML2	3.2	Moderate human impact No obvious human impact, declining
9. Russel Island East	RU-E	2.9	water level No obvious human impact, declining
10. Russel Island West	RU-W	3.8	water level
11. Herman's Bay	HM	3.0	No obvious human impact

Table 5-2. Results of the supervised classification for wetlands in FFNMP and Herman's Bay. Data shown are calculated areas occupied by the various habitat features. See Table 1 for explanations of site codes.

(a) Fathom Five Wetlands

Site code	Submergent (m ²)	Emergent (m ²)	Rock (m ²)	Rock-Shrub (m ²)	Open water (m ²)
BG	11,712	25,264	6,112	25,808	98,896
HR1	7,648	3,120	1,600	7,536	36,688
HR2	944	480	720	3,520	23,712
CN	5,200	4,544	14,064	28,704	111,904
CNP	6,160	3,600	816	3,920	0
BB	262,912	40,336	18,784	66,832	1,792
ML1	30,656	15,824	5,616	13,984	11,536
ML3	2,112	10,784	5,120	12,528	1,760
RU-E	5,920	5,408	1,296	9,072	7,920
RU-W	2,432	8,672	4,896	12,896	9,568

(b) Herman's Bay

Parameter	Wet meadow	Open water	<i>Scirpus</i>	<i>Pontederia</i>	<i>Nuphar & Sparganium</i>
Area (m ²)	4,709	6,540	6,768	5,527	9,124
% Total area	14.4 %	20.0 %	20.7 %	16.9 %	27.9 %

Table 5-3 (a). Error matrix for Fathom Five wetland classification

Class	Training set		Testing set	
	Prod. Acc. (%)	User Acc. (%)	Prod. Acc. (%)	User Acc. (%)
Submergent	98.57	98.57	100.00	86.17
Emergent	96.15	89.29	58.33	100.00
Rock	93.33	82.35	NA	NA
Rock-Shrub	78.57	91.67	NA	NA
Open water	100.00	100.00	NA	NA
Overall Accuracy = 97.28%		% Overall Accuracy = 84.50%		

Table 5-3 (b). Error matrix for Herman's Bay classification

Class	Training set		Testing set	
	Prod. Acc. (%)	User Acc. (%)	Prod. Acc. (%)	User Acc. (%)
Meadow	99.54	100.00	98.08	100.00
Open water	99.75	98.50	100.00	94.59
<i>Sedge</i>	95.24	96.62	84.09	88.10
<i>Nuphar</i>	95.12	95.71	76.67	67.65
<i>Pontederia</i>	99.44	99.44	91.30	97.67
Overall Accuracy = 98.19%		Overall Accuracy = 90.82%		

Table 5-4. Cost-benefits of IKONOS, aerial photo, and Landsat satellite imagery

	IKONOS	Aerial photo	Landsat
Resolution	1m, 4m	variable	15m, 30m, 60m
Pricing*	\$2000/ 100 km ²	\$54000 / 100 km ² **	\$425 per scene (31,110 km ²)
Accuracy of seagrass mapping***	89%	63%	59%

* Pricing for basic level of products

**Pricing for aerial photo is reported in Canadian dollars

***Mumby and Edwards (2002) and Mumby et al. (1997)

Figure 5-1. Map of study wetland sites in Georgian Bay. **BG** - Boat Passage, **HR1** - Cove Island Inner Harbor, **HR2** - Cove Island Outer Harbor, **CN** - Cove Island North, **CNP** - Cove Island North Pond, **BB** - Bass Bay, **ML1** - Hay Bay One, **ML3** - Ragged Bight, **RU-E** - Russel Island East, **RU-W** - Russel Island West, **HM** - Herman's Bay.

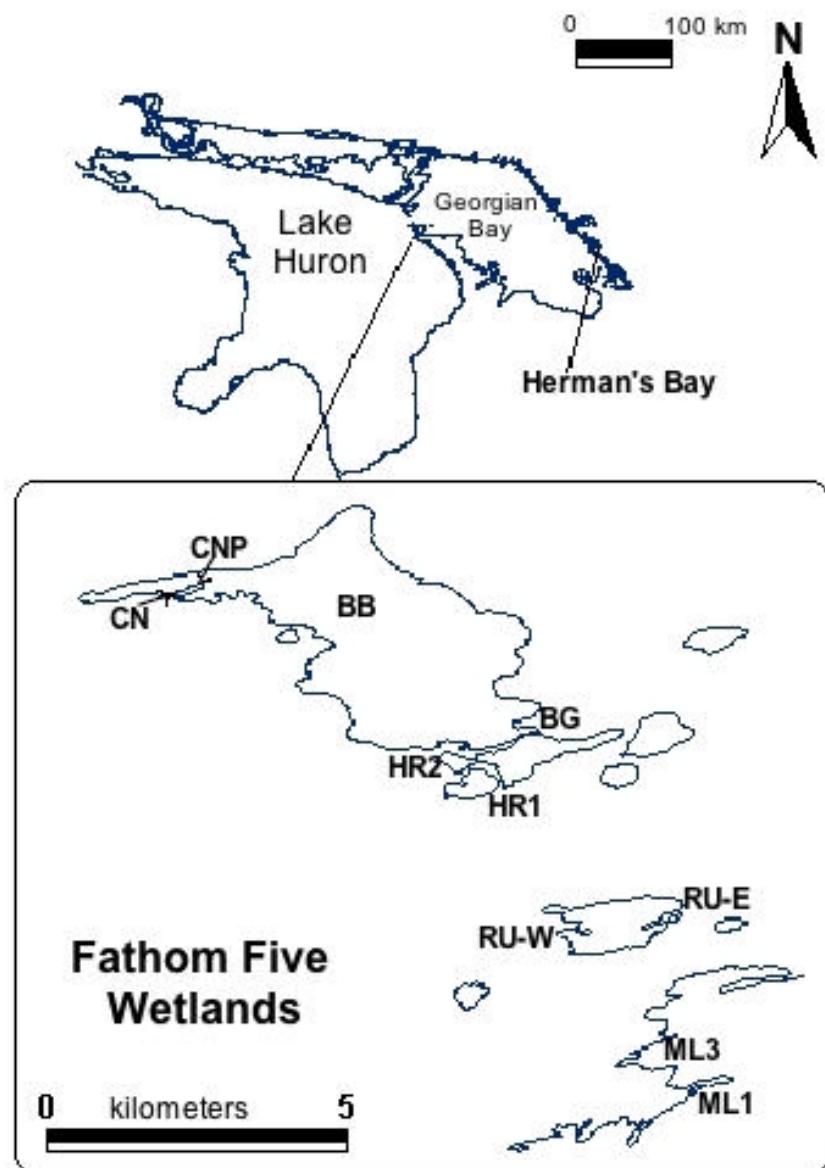


Figure 5-2. IKONOS images of Fathom Five wetlands and Herman's Bay. See Figure 5-1 caption for site code definitions.

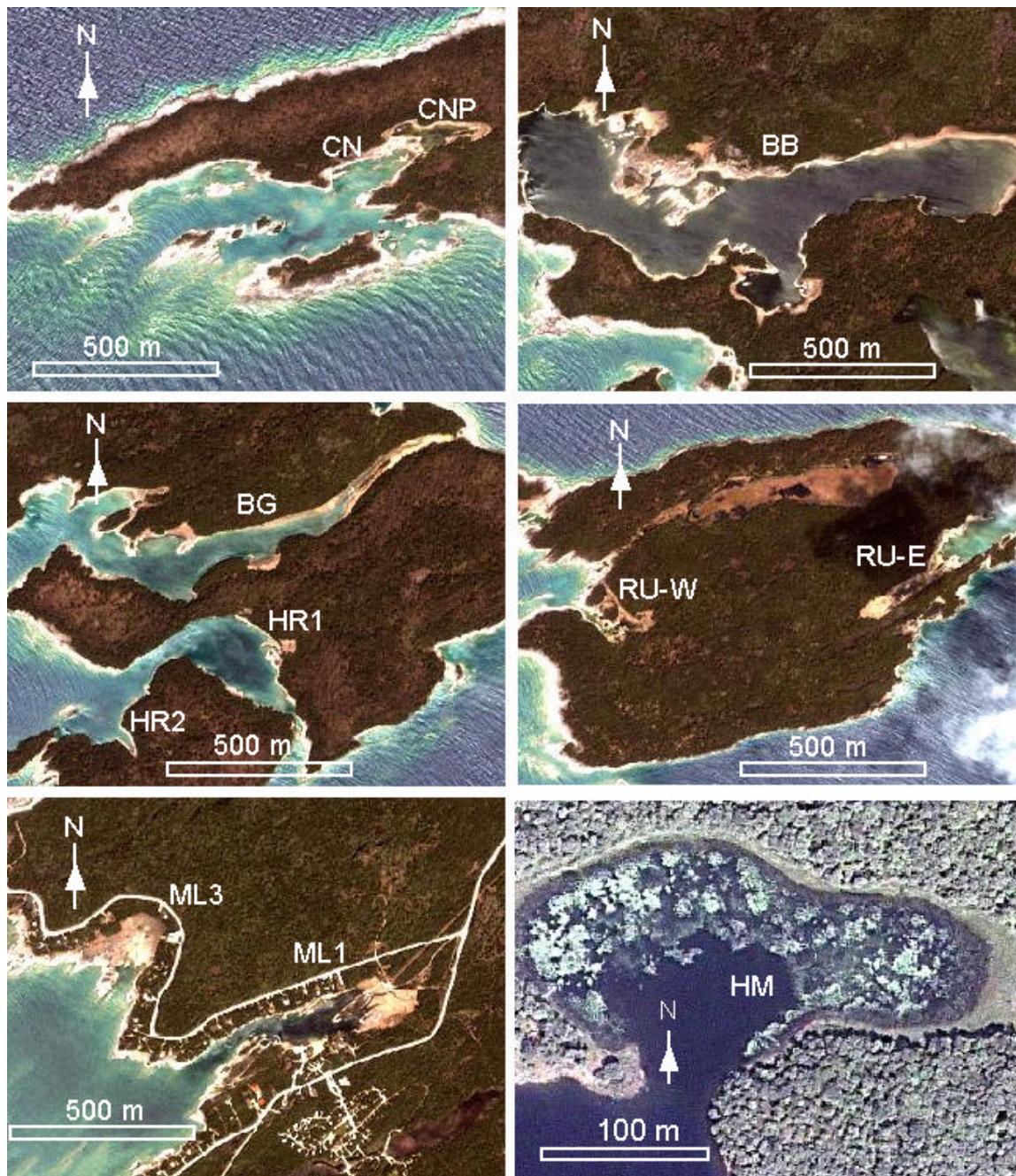


Figure 5-3. Supervised classification procedure for mapping wetland habitats. (a) Three habitat classes in a hypothetical wetland. OP – Open water, SAV – Submergent vegetation, EM – Emergent vegetation. Shaded areas – representative areas in the imagery identified by a human analyst with the aid of ground truth data. These areas are referred to as training areas. (b) A digital representation of the imagery. Values represent the numerical "signatures" for each habitat class. (c) Result of the supervised classification. Image pixels with similar numerical values will be grouped into the same habitat class.

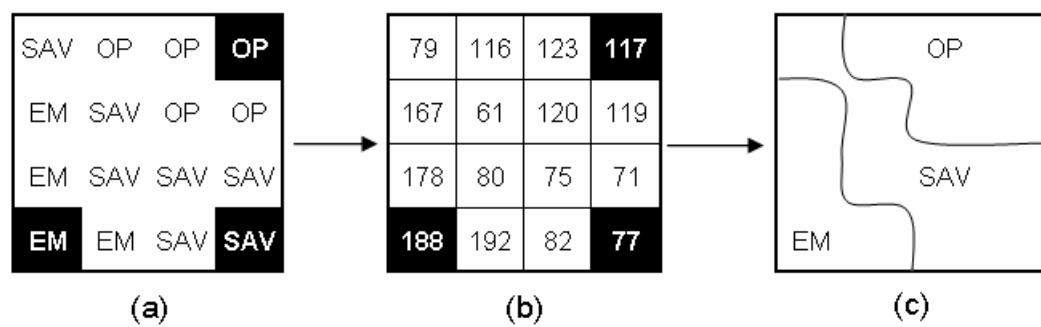
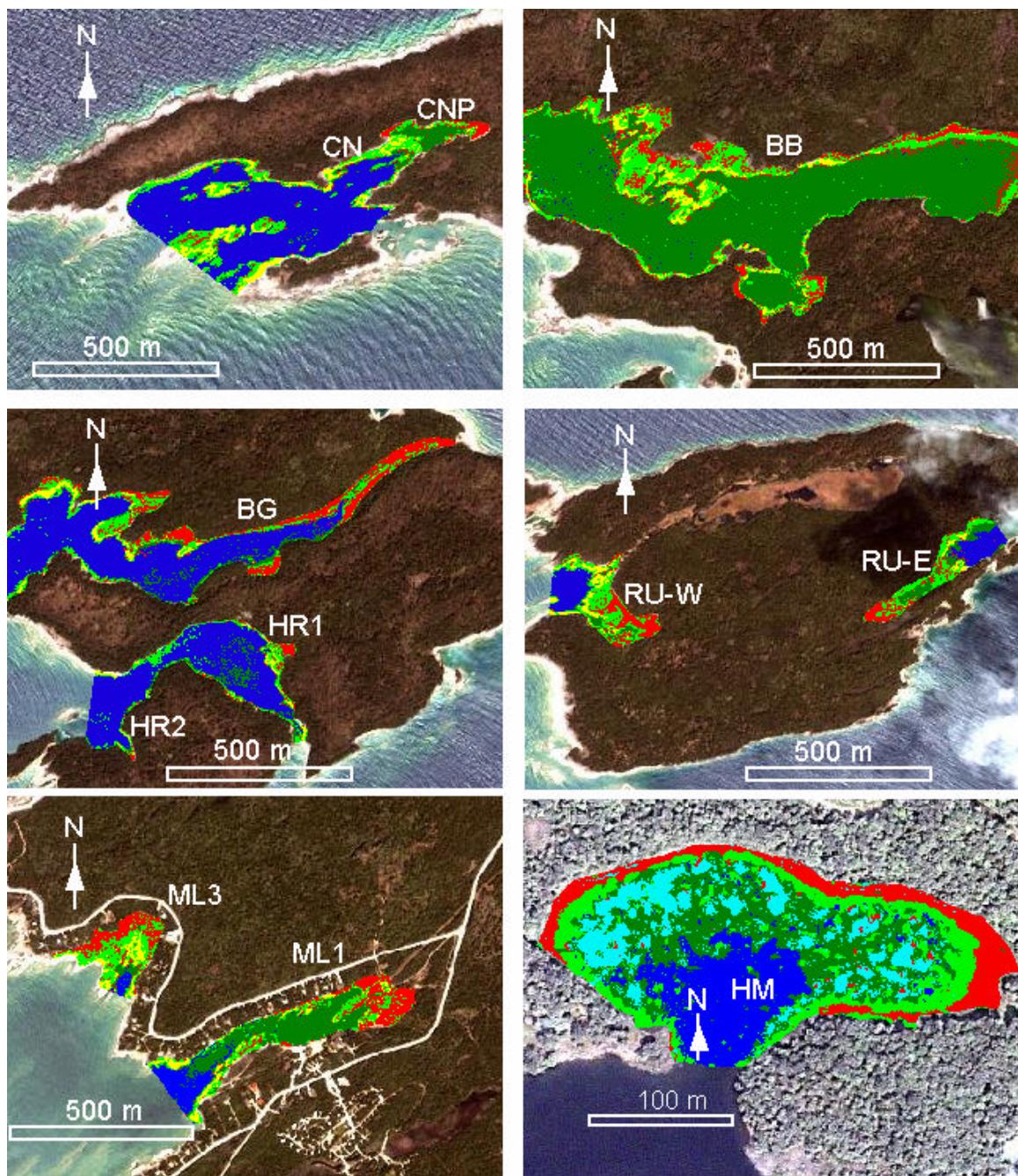


Figure 5-4. Results of supervised classification. Fathom Five wetlands (BG, HR1, HR2, CN, CNP, BB, ML1, ML3, RU-E, RU-W): Red – Emergent plants; Light green – Rock/Shrubs; Dark green – Submergent plants; Yellow – Rock; Blue – Open water. Herman’s Bay (HM): Red – Sedge Meadow; Light green – *Scirpus*; Dark green – *Nuphar* and *Sparganium*; Cyan (light blue) – *Pontederia*; Blue – Open Water.



CHAPTER 6:

**Development of an index to account for degree of exposure and wetland
geomorphology on submerged aquatic vegetation of Lake Huron coastal wetlands**

Wei, A. and Chow-Fraser, P.

ABSTRACT

Wave exposure is an important factor determining distribution and abundance of wetland organisms. In this paper, we develop a Wetland Exposure Index (WEI) to account for the effect of exposure on distribution of submerged aquatic vegetation (SAV) in coastal wetlands of large lakes. This index is based on wetland geomorphology and the Relative Exposure Index (REI), which has been widely applied to lake and marine ecosystems. Ten wetlands from Fathom Five National Marine Park, located at the boundary zone of Lake Huron and Georgian Bay, were evaluated for the effectiveness of this new index. We first extracted information on SAV from IKONOS satellite imagery, and then calculated corresponding exposure index values. Regressions of % SAV against exposure indices indicated that %SAV in Georgian Bay wetlands decreased with exposure but only the regression against WEI resulted in a statistically significant negative slope, and also explained more of the variation in %SAV. Since variation in %SAV among sites may be confounded by impact of human activities, we also performed a partial regression to verify the relationship between % SAV and the exposure indices. We found that the slope for WEI was still marginally significant after accounting for impact of human activities. Results of this study suggest that the Wetland Exposure Index can be used to effectively measure degree of exposure in coastal wetlands, and to predict the overall response of SAV to exposure disturbance.

INTRODUCTION

Submerged aquatic vegetation (SAV) is important to the health of wetland ecosystems such as those occurring along the Great Lakes shoreline. SAV produces oxygen, filters and traps sediment, absorbs nutrients like nitrogen and phosphorus, and provides food and shelter for diverse communities of waterfowl, shellfish, and invertebrates, and is critical habitat for the Great Lakes fish community (Wei et al. 2004). Nutrient levels, water depth, sediment characteristics, light quality, and wave action are all important determinants of SAV colonization in aquatic ecosystems (Chambers and Kalff 1985, 1987, Carr and Chambers 1998, Hudon et al. 2000). Various factors related to anthropogenic disturbance have been implicated in the decline of SAV in wetlands, but it has been difficult to attribute such a change in SAV to a specific anthropogenic disturbance because of confounding effects from natural factors. Therefore, in order to identify human-induced disturbances, we must be able to account for effects of natural factors such as wave exposure.

Submerged aquatic plants tend to occur in sheltered environments because they cannot usually tolerate high wind and wave action, and recent studies have shown that the occurrence of SAV is inversely related to degree of wave exposure (e.g., Fonseca et al. 2002, Riis and Hawes 2003). Keddy (1982, 1984) first introduced the idea that eventually became the basis of the Relative Exposure Index (REI) (Murphrey and Fonseca 1995), which is an estimate of wave energy within a site, and has been used successfully to predict the relative abundance of SAV in both lake and marine environments (e.g.,

Kelly et al. 2001, Short et al. 2002, Stoffels et al. 2005). Direct application of this approach to freshwater coastal wetlands is problematic, because coastal wetlands are hydrologically connected to large lakes that are themselves affected by wave exposure due to geomorphologic characteristics, and this can lead to a biased estimate of %SAV within wetlands. The goal of this paper is to develop a Wetland Exposure Index (WEI) that will account for both exposure disturbance from the attached Great Lake and the unique geomorphology of the wetland itself. We therefore hypothesize that this new WEI would be more useful for predicting the effects of exposure disturbance on SAV than the REI for coastal wetlands of large lakes such as Lake Huron.

METHODS

Study sites

The ten sites chosen for the study were all located within the same region of Fathom Five National Marine Park (FFNMP) and was covered by a single satellite IKONOS image. FFNMP is located at the boundary zone between Georgian Bay to the east and Lake Huron to the west, and includes one large (Cove Island) and one smaller island (Russel Island) (Figure 6-1). Six of our sites were located in Cove Island and these were: Cove Island North (CN), Cove Island North Pond (CNP), Bass Bay (BB), Boat Passage (BG), Cove Island Inner Harbor (HR1), and Cove Island Outer Harbor (HR2). Because Boat Passage had once been used as an east-west navigational channel during periods of high water levels, wetlands in this area may have experienced moderate boat

traffic then. By comparison, both CN and CNP have been free of human disturbance because of their remote location. There were two sites on Russel Island, one on the east (RU-E) and one on the west (RU-W). Like most of Cove Island, Russel island still has very pristine conditions, as there has been minimal human development along its shoreline.

In addition to human-induced disturbance, wetlands on these islands have also been negatively affected by low water level in recent years. During the late 1990s, water levels of Lake Huron dropped to such an extent that the large upland portion of RU-E became hydrologically disconnected from the main portion of Lake Huron because of presence of a rock ridge, and during the 2005 survey, only a small portion of the wetland remained. Low water levels have also separated CNP and BB from the rest of Lake Huron, causing portions of these to be hydrologically *stranded* as well. Water levels dropped from 176.59 m in 1978 to 176.24 m by 1999, and further dropped to 176.12 m by 2002. This decline of 47.3 cm over the 24 years has resulted in CNP being essentially inaccessible to fish from the outer bay (see Figure 6-1 inset), but unlike RU-E, the pond was sufficiently deep (approximately 50 cm) that a relatively diverse community of aquatic vegetation was still evident during the 2005 survey.

We also included two sites on the mainland portion of FFNMP (ML1 and ML3) located in Hay Bay (see Figure 6-1), which has experienced moderate cottage density in recent decades. ML1 is an embayment of the north-eastern arm, surrounded by cottages and a public beach (approximately 700 m²). ML 3 is located further west and

was hydrologically linked with the main portion of Hay Bay. Most sites in this study were sampled for water quality during 2005. All of the sites are considered in very good to excellent conditions with respect to their water-quality (Croft and Chow-Fraser 2007).

SAV data

Aquatic vegetation information (e.g., %SAV) in this study was extracted from IKONOS satellite imagery provided by Parks Canada (Scott Parker, FFNMP, Tobermory, ON). Field vegetation data and GPS records were collected in FFNMP in late August and September of 2004 and 2005. These field vegetation data were converted into GIS layers and then superimposed on an IKONOS image acquired in July, 2002 with the software called ENVI 4.1 (Research Systems, Inc., Boulder CO). Since field data and satellite data had not been collected simultaneously and difference in the aquatic vegetation cover between years may exist, the field data were not directly used as training data. With the aid of field information, representative training areas were identified and were divided into two sets, a training set for supervised classification, and a testing set for post-classification accuracy estimation. A supervised classification procedure (Lillesand and Kiefer 2002) was employed to classify the satellite imagery and then error matrices were calculated to assess the accuracy of classification results by comparing classification results with ground truth information in ENVI 4.1. The overall accuracy for SAV was approximately 85%. Detailed technical descriptions of mapping these wetlands with IKONOS imagery are documented in Wei and Chow-Fraser (2006).

The wetland boundary was determined by visual interpretation of the IKONOS

satellite image. The landward boundary was identified by tracing the edge of emergent plants. The lakeward boundary was mainly determined by the shoreline indentations and the dividing line was placed at the boundary zone between the lake and the wetlands (see Figure 6-1). The % SAV was calculated by dividing the amount of SAV by water surface area within the wetland boundary.

Although the species composition of the SAV was not an important consideration in this study, we identified all species of aquatic plants while collecting ground-truth information in 2005 and the common species encountered included *Myriophyllum spicatum*, *Potamogeton richardsonii*, *Chara sp.*, *Najas flexilis*, *Elodea canadensis*, and *Vallisneria americanum*.

Exposure indices

(a) *Relative Exposure Index (REI)*

The Relative Exposure Index is an estimate of wave exposure (Murphrey and Fonseca 1995), and the underlying concept was originally introduced by Keddy (1982, 1984):

$$REI = \sum_{i=1}^8 (V_i * P_i * F_i) \quad (1)$$

where i is i th compass heading in 45° increments, V is average summer monthly wind speed in km per hour, P is percentage of frequency at the i th direction which wind occurred, and F is fetch in meters. A mean REI was calculated for each site using

formula (1). Wind data for 2002 were obtained from a weather station on Cove Island managed by Environment Canada.

(b) *Wetland Exposure Index (WEI)*

Wetlands that develop along a straight, feature-less shoreline (e.g. fringing wetlands) are completely exposed to wave energy. If wetlands develop along indentations of the shoreline, then they become more sheltered from waves. Generally, the smaller the opening, the more protected the wetland, and the amount of exposure to the lake should therefore be a function of the size of the opening to the lake. To account for degree of exposure to a lake due to wetland morphology, we have developed a simplified geomorphology index (GI), which is calculated by dividing the width of the wetland opening by the perimeter of the wetland (Figure 6-2). Values of the GI should range between 0 and 1, although in practice, wetlands would seldom take on the extremes of zero or 1. A zero value would only be assigned to wetlands that are hydrologically disconnected from a lake because of low water-levels. At the other extreme, only fringing wetlands occurring along a perfectly straight shoreline would assume an index value of 1. All measurements pertaining to the GI (i.e. opening width and perimeter of the wetlands) were estimated from the IKONOS satellite image in GIS.

By definition, coastal wetlands refer only to wetlands that are hydrologically connected to a Great Lake (e.g. Lake Huron). From this definition we can infer that aquatic vegetation of coastal wetlands would be affected by exposure disturbance generated within these wetlands and from the Great lakes. To account for exposure

disturbance from both sources, we proposed the Wetland Exposure Index (WEI) as follows:

$$WEI = (GI + 1) * \log(REI) \quad (2)$$

The WEI combines aspects of REI and GI, and can be regarded as a modified REI for coastal wetlands. All of the data needed to calculate the WEI are shown in Table 6-1.

Statistical analysis

We used linear regression analysis with permutation test (Program for multiple linear regression with permutation test by Pierre Legendre, Universite de Montreal) to test the relationship between %SAV and exposure disposure, and compared ability of the two indices to predict %SAV. Data for %SAV were first arcsin-square-root transformed.

RESULTS AND DISCUSSION

%SAV and categorical exposure groupings

We first analyzed differences in %SAV among sites from islands, mainland, and stranded areas that represent different exposure groups (Table 6-1). In general, sites on islands of FFNMP are more exposed to large lake process than mainland sites. Sites in stranded areas have the least impact from the lake because they have been hydrologically disconnected from the lake. These differences in exposure were reflected in significant differences in mean %SAV among sites; Island sites had the lowest percentage of SAV (6.53 ± 5.91), while the stranded sites had the highest (mean of 99.65 ± 0.35), and mainland sites had intermediate values (63.60 ± 9.10) (ANOVA, $r^2 = 0.90$, $p = 0.0003$, $n = 10$).

Comparison of exposure indices

We then calculated REI and WEI to quantify degree of exposure for the study sites and related the indices to %SAV separately. We found a negative slope for REI and WEI, but only the slope for WEI was statistically significant when the P-value was calculated by permutation (Table 6-2). The overall model was not significant for either REI ($r^2 = 0.003$, $p = 0.871$) or WEI ($r^2 = 0.300$, $p = 0.101$) when all of the data were included. We reran the analysis after removing the data point for Bass Bay (BB, open symbol in Figure 6-3) because it was found to be an outlier for %SAV against log REI (Figure 6-3). In this instance, we found a significant overall regression for WEI (Figure 6-3, $r^2 = 0.494$, $p = 0.035$), which accounted for almost 50% of the variation. This contrasts the regression of %SAV against REI, where removal of the outlier did not produce a statistically significant regression coefficient.

It is clear that the percent cover of SAV in Fathom Five wetlands is negatively affected by exposure disturbance, and this agrees with Kelly et al's (2001) observation that wave exposure had a negative effect on establishment of seagrass. However, the Relative Exposure Index (REI), which has been successfully applied by Kelly et al. to marine systems, was not an appropriate measure of exposure for our coastal wetlands. Consistent with our hypothesis, the Wetland Exposure Index (WEI) proved to be better than the REI for predicting SAV distribution in coastal wetlands of large lakes.

In addition to exposure disturbance, water depth has been used as an independent

variable to predict percent cover of SAV (e.g., Fonseca et al. 2002). We were unable to integrate water depth into our models due to the lack of high-quality bathymetric data for these wetlands when examining the relationship between % SAV and exposure disturbance. Inclusion of water depth as an additional independent variable could improve the predictions of SAV.

Potential confounding effect of exposure and water quality on %SAV

To untangle the potential confounding effect of exposure and human impact on %SAV, we performed a partial regression with permutation test (Legendre and Legendre 1998) to determine the relationship between %SAV and WEI while the effect of human activities was taken into account. We partitioned out the effect of human impact by regressing ranking of human impact against WEI, and regressed SAV against the residuals of WEI. We found that the slope for WEI was still marginally significant ($p\text{-permutation} = 0.051$, $n = 10$) while the impact of human activities was taken into account. This suggests that exposure disturbance is a natural factor that is more likely to influence the establishment of SAV than anthropogenic disturbance in Fathom Five wetlands.

CONCLUSIONS

Results of this study support our original assumption that aquatic vegetation in coastal wetlands is affected by exposure disturbance that can be attributed to the lake and any disturbance originating from inside the wetland. Our results reinforce the general

expectation that SAV can only survive where there is moderate wave action and currents. Therefore, we recommend that effect of exposure and geomorphology be partitioned out when examining the effects of anthropogenic disturbance on submersed aquatic vegetation.

ACKNOWLEDGEMENTS

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Georgian Bay. *Fisheries* 32: 167-173.

Table 6-1. %SAV, exposure indices, and ranking of human impact

Site	Group	% SAV	REI	GI	WEI	Impact ranking*
BB	Stranded	99.3	861396	0	5.9352	1
BG	Island	10.6	534029	0.0328	5.9153	2
CN	Island	4.4	662401	0.1447	6.6632	1
CNP	Stranded	100	197755	0	5.2961	1
HR1	Island	17.3	387523	0.0298	5.7551	1
HR2	Island	3.8	288867	0.1593	6.3308	1
ML1	Mainland	72.7	342197	0.0738	5.9426	4
ML3	Mainland	54.5	113510	0.0464	5.2896	3
RU-E	Island	42.8	203620	0.0438	5.5412	1
RU-W	Island	20.3	182737	0.0805	5.6854	1

*The impact ranking was mainly determined by cottage densities around the sites and recreational activities. 4 is indicative of the most impacted conditions and 1 is indicative of the least impacted conditions.

Table 6-2. Summary of regression of %SAV against exposure indices. Data for %SAV and REI were first arcsine-square-root transformed and logarithmic transformed, respectively.

Variable	estimate	t	P-permutation	P-parametric	n
Intercept	4.44122	2.20776		0.02914 *	
WEI	-0.63670	-1.85146	0.04000 *	0.05062	10
Intercept	1.31612	0.37474		0.35880	
REI	-0.10735	-0.16827	0.43400	0.43527	10
Intercept	4.67191	3.01807		0.00972 *	
WEI	-0.69204	-2.61088	0.01200 *	0.01743 *	9
Intercept	5.71007	1.75074	0.06173		
REI	-0.93000	-1.55552	0.07100	0.08189	9

Asterisk indicates that the parameter estimate is significant at the 0.05 level. Regression coefficient: One-tailed test in direction of sign. Number of permutations of raw data: 999.

Figure 6-1. Fathom Five National Marine Park (45.3°N , 81.7°W), showing the locations of the study area.

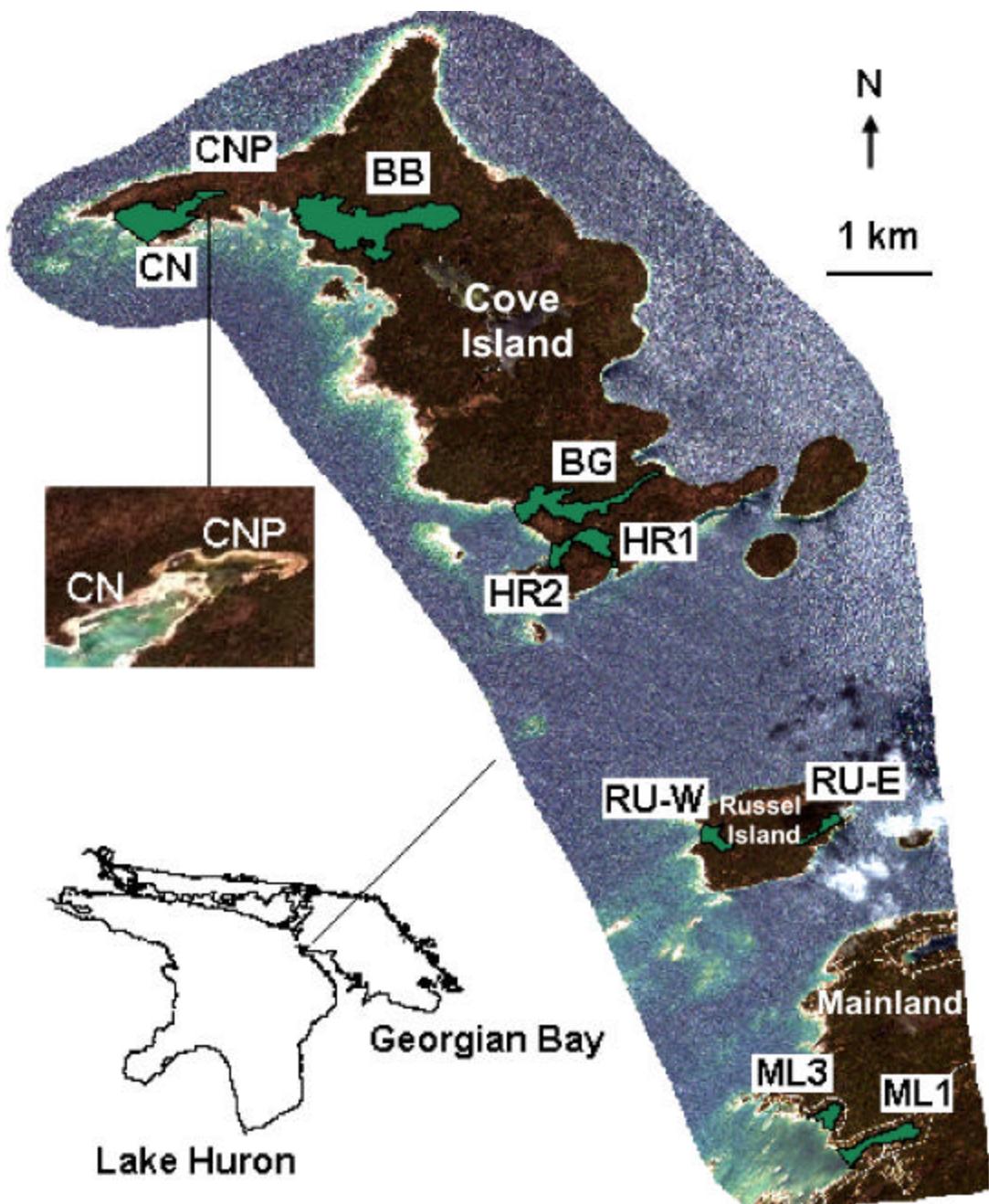


Figure 6-2. A schematic diagram showing the measurement of Geomorphology Index (GI)

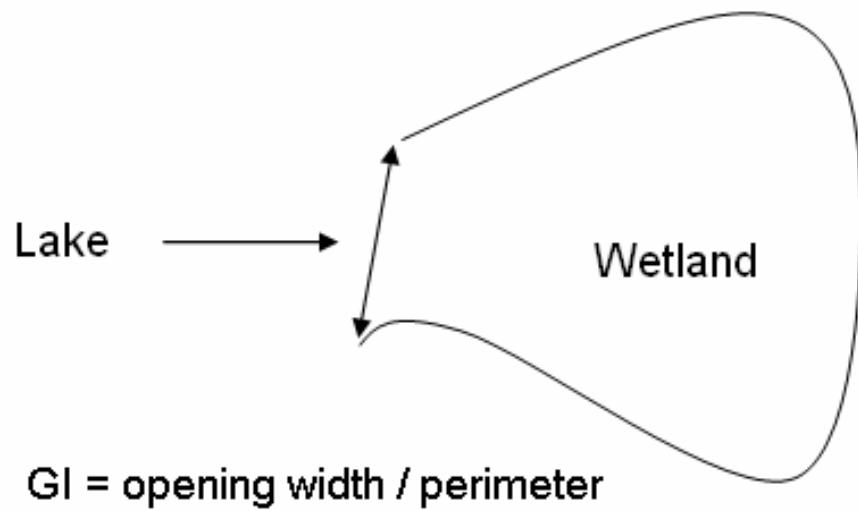
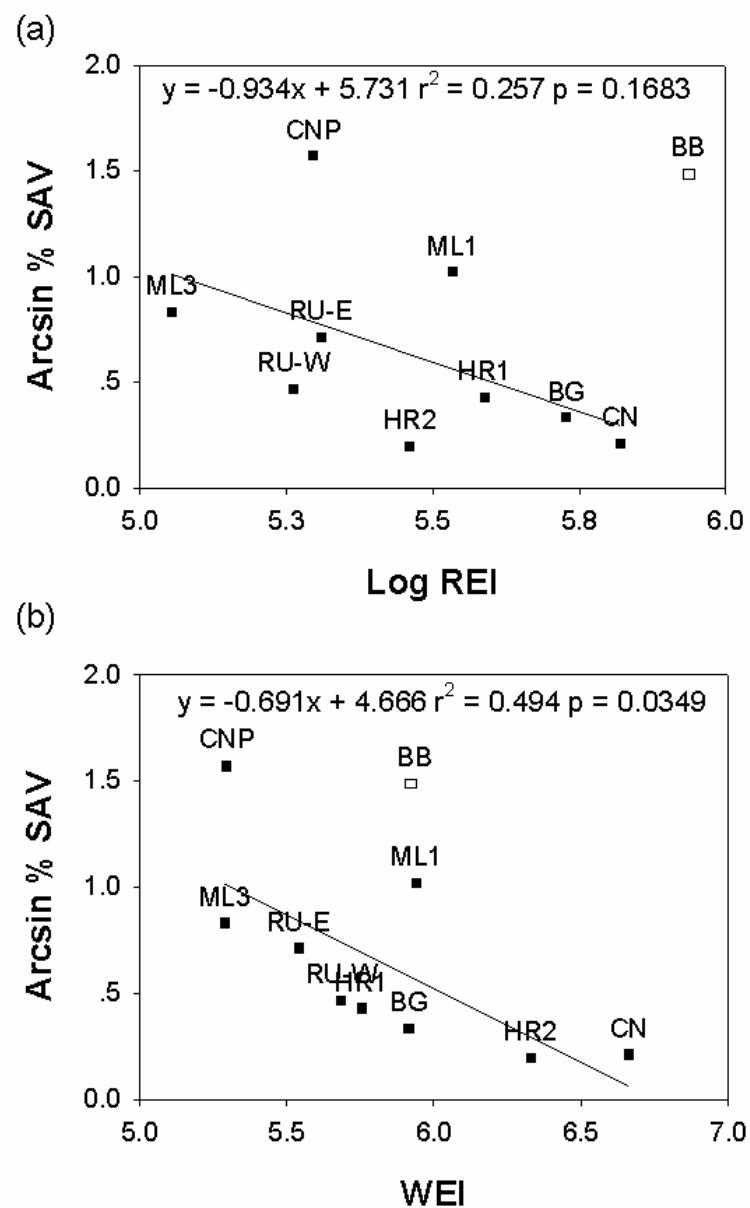


Figure 6-3. Relationship between %SAV and A) REI and (B) WEI (n = 9). Regression line through solid symbols only.



GENERAL DISCUSSION

Summary

The overall purpose of this study was to examine the response of coastal wetlands to declining water level and environmental disturbances in the Great Lakes.

In Chapter 1 we showed that areal change of emergent plants was significantly correlated with inundated area after controlling for the effect of water level fluctuation; however, there was no significant correlation between emergent cover and water level after controlling for inundated area. This is an important consideration when multiple sites from the same Great Lake are compared since the same water level may correspond to vastly different inundated areas for different marshes. We also showed that changes in emergent cover were significantly correlated with human population after controlling for water level effects. We predicted that the synergistic effect of high water level (expressed as inundated area) and increased human population would induce greater detrimental impact on the emergent plants than would either stressor alone. We also predicted that predictive models based on site bathymetry would be more powerful than those based on water-level alone.

In Chapter 2 we showed that sustained high water levels was the major natural disturbance and it alone accounted for 88% of the variation in *Typha*. After partitioning out the effect of water level, both human population growth and the presence of exotic species were still significantly related to the decline of native *Typha*. We also showed that

high water levels limited the distribution of both *Typha* and *Glyceria*; however, once water levels receded, *Glyceria* was the more successful colonizer. We predicted that multiple stressors (i.e. high water levels, human population growth as a surrogate of urbanization, and invasive species) would interact with each other to influence changes in the native *Typha* community and cause greater detrimental impact.

In Chapter 3 the approach introduced in Chapter 1 was verified at a regional scale. A predictive model based on water level alone could be problematic if multiple sites from a region experiencing the same water level are compared because in that instance, water level would be a constant and cannot be used as an explanatory variable for emergent cover. We predicted that corresponding inundated areas at these sites were likely to be very different due to variation in site topography, and could be used to predict the response of emergent vegetation. The approach proposed in Chapter 1 was originally designed for a single site study and it must be modified before it can be applied to multiple sites because a large wetland tends to have a large inundated area and the differences in wetland area must be taken into account. We were able to modify this approach and developed a regional model to examine the effect of water-level fluctuations on emergent plant cover over a large spatial scale. We showed that the inundated area (%) was a good predictor of emergent cover for multiple sites experiencing the same water level. This result confirmed that the new approach based on site bathymetry was effective for predicting the overall response of emergent marshes at both a single site and at multiple sites within a large region.

In Chapter 4 we showed that the Great Lakes fish community utilized certain shoreline features (especially wetlands) disproportionately to their availability. We also provided evidence that the distribution of wetland-associated taxa was influenced by wetland type (i.e., protected embayment versus open-shoreline wetland)

Chapter 5 and 6 outlined the use of high resolution satellite imagery to quantify the supply of fish habitat and wave disturbance in coastal wetlands. In Chapter 5, we explored the capabilities of using IKONOS satellite imagery to map different types of aquatic vegetation and habitat features in Great Lakes wetlands. The comparison of results of the image analysis with reference data indicated that the overall accuracy of mapping was approximately 90%. This suggests that high resolution IKONOS imagery can be used effectively to monitor the change in aquatic vegetation and thus track alterations in fish habitat in Great Lakes coastal marshes.

In Chapter 6 we developed a Wetland Exposure Index (WEI) to quantify exposure in coastal wetlands and applied this index to account for the effect of exposure on distribution of submerged aquatic vegetation (SAV) in coastal wetlands of large lakes. We compared the WEI to the Relative Exposure Index (REI), which has been widely applied to lake and marine ecosystems and showed that %SAV in Georgian Bay wetlands decreased with exposure but only the regression against WEI resulted in a statistically significant negative slope, and also explained more of the variation in %SAV. We also showed that the slope for WEI was still marginally significant after accounting for the impact of human activities. Results of this study suggest that the Wetland Exposure

Index can be used to effectively measure degree of exposure in coastal wetlands, and to predict the overall response of SAV to exposure disturbance.

Significance of the work

(1) Water levels

Water-level disturbance caused either by lake-level regulation or by global climate change has been identified as one of the most significant stressors affecting Great Lakes coastal wetlands. Traditional models based on water levels would be inappropriate for such basin-wide assessment. We propose a solution that combines water level information with site-specific depth variation to investigate the response of emergent vegetation to different water-level regimes at both local and regional scales. Our finding that there is a global pattern in the aggregate response of emergent vegetation to water level disturbance has important implications for wetland research and management. Whereas in the past, digital bathymetric data were more difficult to obtain than historic wetland maps, it will soon become the reverse as more and more government-sponsored digital libraries become available on line. This will make it a relatively simple task to calculate inundated area, the potential habitat for emergent plants in any wetland. The generalized approach that we have provided here could become a very useful management tool to assess the basin-wide response of emergent marshes to water-level disturbances caused either by lake-level regulation or by global climate change scenarios.

(2) Urbanization

This project has further confirmed that urbanization in wetland watersheds may exaggerate natural stressors such as water-level disturbance, and adversely affect wetland plant that are adapted to the natural range. The general relationship between emergent plants and water-level disturbance may be modified by the degree of urbanization. Because of the negative impacts of urbanization on emergent plants, the recovery rate of native emergent plants in a highly urbanized watershed will be slower than that in a primarily forested watershed once water level declines.

(3) Invasive species

The project has demonstrated that high water levels limited the distribution of both native plants (e.g. *Typha*) and invasive plants (e.g. *Glyceria*); however, once water levels receded, the invasive species became the more successful colonizer. The important implication of our results is that projected water level decline due to climate change may not necessarily favor the restoration of a desirable native marsh plants because of the presence of other disturbances such as exotic and invasive species and altered nutrient regime. Although water-level fluctuations is the largest single disturbance affecting the response of coastal marsh communities, it is necessary to study the effects of multiple factors simultaneously, since the response of ecosystems to one factor may be significantly influenced by others.

(4) Shoreline features

This study is one of the most extensive examinations of fish distribution patterns at the scale of the Great Lakes basin. By revealing the association between certain species or

species-complexes and particular shoreline features within the Great Lakes, we provide a scientifically defensible method for managers and policy-makers to target the appropriate shoreline reaches for restoration, preservation or conservation. The results of this study have been used by U.S. Nuclear Regulatory Commission to assess the environmental impact for license renewal of nuclear plants on the habitats most suitable for reproduction by the Great Lakes fish community (Generic Environmental Impact Statement for License Renewal of Nuclear Plants, Supplement 23, U.S. Nuclear Regulatory Commission, Office of Nuclear Reactor Regulation, Washington, DC 20555-0001. Final Report, August 2005). Relationships between fish and wetlands can be used to predict the response of fish to changes in fish habitat due to declining water levels.

(5) Wave exposure

Various factors related to anthropogenic disturbance have been implicated in the decline of wetland organisms, but it has been difficult to attribute such a change to a specific anthropogenic disturbance because of confounding effects from natural factors. Therefore, in order to identify human-induced disturbances, we must be able to account for effects of natural factors such as wave exposure. In addition, low water levels will change wetland areas and wave exposure in coastal wetlands will be altered by declining water levels. Wetland Exposure Index developed in this project can be used to effectively measure degree of exposure in coastal wetlands, and to predict the overall response of submerged aquatic vegetation to exposure disturbance. Wave exposure is also an important factor determining distribution and abundance of other wetland organisms.

Therefore, we anticipate that the Wetland Exposure Index developed in this project would generate a great deal of interest among wetland ecologists.

(6) Mapping wetland habitats

An important first step in the management of coastal wetlands is the development of a basin-wide inventory that can be updated at regular intervals. The wide distribution of wetlands in the Great Lakes basin necessitates the use of remote sensing technology, such as aerial photographs or satellite images. To date, no study has been published that details the use of IKONOS satellite imagery in freshwater coastal areas, such as the Laurentian Great Lakes. Our results indicate that use of IKONOS imagery to inventory wetlands has the advantage of wide spatial coverage and the precision of supervised classification, thus meeting the requirement for gathering synoptic information on wetlands at regional scales.

GLOSSARY

Automated classification: Process that groups data according to some criteria by automatic or semi-automatic as opposed to purely manual methods. During classification, each pixel is compared to each of the class signatures. The comparison is performed by computer using a predetermined classification algorithm.

Confusion matrix: also called error matrix. A matrix compares, on a category-by-category basis, the relationship between known reference data (ground truth) and the corresponding results of an automated classification.

Digital elevation model (DEM): Digital model of landform data represented as point elevation values.

Disturbance: any relatively discrete event in time that disrupts an ecosystem.

Emergent aquatic plants: Rooted plants growing in shallow water with a portion of their stems and leaves growing above the water surface.

Exotic species: Species that have been introduced to a region or continent, usually but not always through human activity.

Fetch: The distance wind can blow unobstructed over water. The greater the fetch, the greater the waves for a given wind speed.

Floating-leaved plants: a free-floating plant such as duckweeds; or one that is rooted to the bottom but has floating leaves such as the white water lily.

GIS: Geographic Information System, a computer system for collecting, checking, integrating and analyzing information related to the surface of the earth.

Global positioning system (GPS): a satellite-based device that records locational (X, Y, Z) and ancillary data for portions of the earth.

Grid: Raster GIS data model where each grid cell is referenced or addressed individually and organized in columnar fashion.

Ground truth data: known reference data.

Homogeneous region: If attributes are identical within a selected region, they are said to be homogeneous.

IKONOS: A satellite was developed by Space Imaging and launched in 1999. IKONOS collects data in four multi-spectral bands (i.e. blue, green, red, and near infrared) at a ground resolution of 4 m. IKONOS also incorporates a 1-m resolution panchromatic band (0.45 to 0.90 μm). The panchromatic band and the multi-spectral bands can be combined to produce multi-spectral imagery with a resolution of 1 m. A typical image is 11 x 11 km in size.

Interpolation: A process of predicting unknown elevational values by using known values occurring on multiple locations around the unknown value.

Marsh: A wetland dominated by herbaceous vegetation. A marsh does not have a canopy of large trees, is open to the sun, and is dominated by grasses, emergents, sedges, and reeds.

Native species: Species that inhabited in an area before European settlement.

Raster: A form of GIS graphic data structure that quantizes space into a series of uniform shaped cells.

Remote Sensing: The observation of objects or groups of objects, normally at a distance, most often with the use of some form of mechanical or electronic device. The data can either produce an image or be stored for later retrieval

Resolution: 1. The amount of earth surface represented by a single grid cell in a raster GIS. 2. The ability of a digitizer to record increments in space. The smaller the units it can handle the better its resolution.

Spatial: Anything dealing with the concept of space. In the geographic context, primarily dealing with the distribution of things on the surface of the earth.

Spatially correlated: Sets of objects that are spatially associated may also show statistically significant relationships that may indicate some connection between the processes acting on both objects or even the processes acting on one set of objects influencing those of another set of objects in the same area.

Stressor: Anything that changes the functioning of an ecosystem.

Submergent aquatic vegetation (SAV): Plants that have their stems and leaves below the water surface. They may have some flowing parts above.

Vector: A graphic data structure that represents the points, lines, and areas of geographical space by exact X and Y coordinates

Watershed: The entire surface drainage area that contributes water to a creek, stream, river, wetland, or lake.

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